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EVALUATION OF ECONOMIC FEASIBILITY AND ANIMAL PERFORMANCE FOR A NOVEL MANURE COLLECTION AND ANAEROBIC DIGESTION SYSTEM AT A COMMERCIAL SWINE FINISHER ENTERPRISE

A Thesis in

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by

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ABSTRACT

Anaerobic digestion of manure can provide many benefits at the farm-level. The following case-study at a commercial swine finishing operation near Danville, Pennsylvania, USA was conducted to evaluate the operational and economic feasibility of a novel manure management system in conjunction with an anaerobic digester. Investment capital for the system was provided in part by the producer and by public and private grants. The system utilized under-floor manure storage pits to collect manure for delivery to a digester, and then stored post-digested manure (digestate) in under-floor storage within the same swine houses. Positioning of manure collection pits under swine dunging areas in two large-pen 2200-head buildings allowed for the collection of 75% of total manure volume, which was moved to the digester. Digester-produced biogas content was approximately 28% carbon dioxide and 72% methane. No additional post-digestion manure storage construction was necessary at the farm. The cost-savings of electricity produced from combustion of biogas (monthly value of US $477.10) was nearly equal to the producer's debt service for capital investment required for the construction of the manure handling and digester system (monthly payment of US $478.54). Debt service did not include grant funds. Monitoring of air quality indicators both before and after the introduction of digestate to under-floor manure storage pits in swine housing resulted in no observations of hydrogen sulfide (H_2S) or methane (CH_4) concentrations above critical safety levels. No recorded concentrations of oxygen (O_2) were below critical entry guideline levels. Hourly mean ammonia (NH_3) concentrations at pig level (0.15 m above the floor) before digestate was present in the buildings were higher ($P<0.05$) compared to when digestate was present (24 ± 2.8 ppm vs. 17 ± 1.0 ppm). During steady-state digester operation the minimum ventilation system for the swine buildings was changed from manure pit ventilation to an end-wall fan on a timer. Hourly mean NH_3 concentrations at pig level were higher ($P<0.05$) after fan removal (37 ± 0.9) than when pit fans were present (17 ± 1.0 ppm). Swine group average daily gain, feed-to-gain ratio and culls and mortalities information from the case farm were compared to that of two other farms. Average daily gain of pigs on the case farm was lower ($P < 0.05$) than that of another farm receiving feeder pigs from the same sow units. Feed efficiency and a combination of culls and mortalities were statistically similar among three farms receiving pigs from the same sow units. We conclude that the novel manure collection system used on this farm can eliminate the need of a post-digester storage facility and reduce the cost for electricity for a commercial swine enterprise. Electric cost-savings made the combined digestion and manure collection system at this location were more affordable than that of a conventional digestion system. External funding and low interest financing were necessary in order for finance payments to be offset by electric cost-savings. Air quality measures did not indicate that the introduction of digestate into under-floor manure pits caused degradations of air quality at pig level. Because of the variations in management no clear effects could be determined from this manure treatment system on the growth performance of pigs in these buildings.

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Chapter 1. Introduction

Anaerobic digestion is a naturally occurring process in which organic material is decomposed by microorganisms in an oxygen-free environment. When used to treat manure several benefits can be realized. Benefits include stabilization of organic matter (Safely & Westerman 1994) and reduction of manure odors and pathogens (Salminen & Rintala 2002, Braber 1995, Shih 1987, 1993), while most manure nutrients remain in recoverable digested material (Salminen *et al.* 2001, Shih 1987, 1993). A product of anaerobic digestion is methane-rich biogas that can be used as an alternative energy source (Salminen & Rintala 2002).

Massé & Droste (2000) reported that anaerobic digestion is a four-step process involving (1) fermentative or acid forming bacteria; (2) hydrogen-producing acetogenic bacteria; (3) acetoclastic methanogens; and (4) carbon dioxide-reducing methonogenic bacteria that operate to perform hydrolysis, fermentation, and methane production in series and parallel steps. Biogas produced from manure degradation contains 60-80% methane (Roos *et al.* 2004) and can be combusted in an engine-generator system to produce electricity. The electricity can be used on the farm to offset electricity purchase.

Widespread adoption of anaerobic digestion technology has not occurred because of high capital investment and nominal economic return (Hill *et al.* 1985; Safley & Westerman 1994; Braber 1995). Traditionally there has been minimal incentive for livestock producers to seek alternative energy sources due to historically affordable fossil fuels and electricity. The competitiveness of biogas with other fuel used for heat or combined heat and power (CHP) is limited (Lantz *et al.* 2007). Many believe that anaerobic digestion will become more affordable as advances in technology, lower capital investment requirements, and rising costs of non-renewable fuels will make biogas systems more economical (Wiese & Haeck 2006). Rising social costs associated with environmental impacts, energy use, and manure odor generation make manure digestion attractive and may lead to economic subsidies for anaerobic system development.

A major component of the cost of constructing anaerobic manure digestion is that of manure storage facilities for post-digested (digestate) manure. If construction of digestate storage could be avoided the implementation of farm-level digestion may be more affordable. One method to avoid additional cost of digestate storage in a standard commercial swine housing unit would be to segregate the standard under-floor manure storage volume into compartments that store pre- and post-digested manure volumes and to collect a majority of manure from dunging areas in specified compartments. Pigs in swine housing commonly rest and excrete in different areas within the floor space of their living space. Relaxation is important for health and growth of pigs. In thermonuetral conditions of 16-18 ºC (defined by Petherick 1983) fattening pigs spend 88% (Huynh *et al.* 2005) to 90% (Ekkel *et al.* 2003) of the 24 hour day lying. Others have reported lying time as 78% (Taylor *et al.* 2006) and 80% (Haugse *et al.* 1965). Pigs are reported to space themselves near pen perimeters (Grandin 1980). It has been suggested that pigs like to dung in open areas (Fritschen 1975). Dunging areas arise in large-pen settings because pigs prefer to urinate and dung away from resting areas (Stolba & Wood-Gush 1989).

Weigand *et al.* (1994) found that swine used 32 and 27% of the pen floor area as a dunging area for small and large pens, respectively. The study concluded that pen size had no impact on inter-animal space and animal perception of pen space depended on the amount of pen wall available to the animals.

Storing digested manure under the swine living facility could eliminate the need for a separate post-digestate facility, but it may increase the concentration of dangerous gases in the swine living area. Methane (CH_4) is explosive at concentrations between 5-15% (NIOSH 1990). The US Occupational Safety and Health Administration's Permissible Exposure Limit for gaseous ammonia (NH_3) is 50 ppm. The 10-minute recommended exposure limit for hydrogen sulfide (H_2S) is 10 ppm. The minimum oxygen concentration level for safe human entry is 19.5% (NIOSH 1990). Monitoring of indoor air quality and pig growth efficiencies for potential negative impacts on growth or mortality in such housing would be warranted.

It is not clear whether an under-floor manure storage system designed to segregate raw manure and digested manure will provide sufficient amounts of raw manure for the practical operation of an anaerobic digester. Nor is it clear how this unique manure handling system may affect pig health and performance.

Therefore, the objectives of this study were to:

(1) Quantify the proportion of manure deposited into manure collection pits located under observed dunging areas in a commercial large-pen swine finishing operation.

(2) Evaluate manure constituents before and after digestion.

(3) Quantify manure loading, biogas production and quality, electricity production and engine run-time for a two-year period of steady-state operation of an anaerobic digester.

(4) Evaluate the economic viability of the digestion system based on producer capital costs and electric cost-savings.

(5) Characterize concentrations of CH_4, NH_3, H_2S, and O_2 in swine living space of the swine facility before and after the introduction of digestate to under-floor manure storage space, as well as after digestate introduction when manure pit ventilation was operational and when pit ventilation was not operational.

(6) Evaluate growth performance and mortality and culling with digestate introduction to manure storage located beneath swine living areas, in comparison to pigs at other barns where no manure treatment occurred.

References

BRABER, K. (1995). Anaerobic digestion of municipal solid waste: a modern waste disposal option on the verge of breakthrough. *Biomass Bioenergy* 9, 365-376.

EKKEL, E. D., SPOOLDER, H. A. M., HULSEGGE, I. & HOPSTER, H. (2003). Lying characteristics as determinants for space requirements in pigs. *Applied Animal Behaviour Science* 80, 19-30.

FRITSCHEN, R. D. (1975). *Toilet Training of Pigs On Partly Slatted Floors*. University of Nebraska, Lincoln, NebGuide G 74-40.

GRANDIN, T. (1980). Livestock behavior as related to handling facilities design. *International Journal for the Study of Animal Problems* 1, 33-52.

HAUGSE, C. N., DINUSSON, W. E., ERICKSON, D. O., JOHNSON, J. N. & BUCHANAN, M. L. (1965). A day in the life of a pig. *Feedstuffs* 37, pp. 18-23.

HILL, D. T., HOLMBERG, R. D. & BOLTE, J. P. (1985). Operating and performance characteristics of scraped swine manure as a thermophilic anaerobic digestion substrate. *Agricultural Wastes* 14, 37-49.

HUYNH, T. T. T., AARNINK, A. J. A., GERRITS, W. J. J., HEETKAMP, M. J. H., CANH, T. T., SPOOLDER, H. A. M., VERSTEGEN, M. W. A. & KEMP, B. (2005). Thermal behavior adaptation of growing pigs as affected by temperature and humidity. *Applied Animal Behaviour Science* 91, 1-16.

LANTZ, M., SVENSSON, M., BJÖRNSSON, L. & BÖRJESSON, P. (2007). The prospects for an expansion of biogas systems in Sweden – incentives, barriers and potentials. *Energy Policy* 35, 1830-1843.

MASSÉ, D. I. & DROSTE, R. L. (2000). Comprehensive model of anaerobic digestion of swine manure slurry in a sequencing batch reactor. *Water Resources* 34(12), 3087-3106.

NIOSH (1990). *NIOSH Publication No. 90-103: Preventing deaths of farm workers in manure pits*. US National Institute for Occupational Safety and Health, May 1990.

PETHERICK, J. C. (1983). A biological basis for the design of space in livestock housing. In: *Farm Animal Housing and Welfare* (Eds S. H. Baxter, M. R. Baxter & J. A. S. C. MacCormack), pp 103-120, Martinus Nijoff Publisher, Boston, MA.

ROOS, K. F., MARTIN, JR, J.H. & MOSER, M. A., 2004. *AgSTAR Handbook; A manual for developing biogas systems at commercial farms in the United States; Second Edition.* U.S. Environmental Protection Agency. EPA-430-B-97-015.

SAFLEY, JR., L.M., & WESTERMAN, P. W. (1994). Low-temperature digestion of dairy and swine manure. *Bioresource Technology* 47, 165-171.

SALMINEN, E. & RINTALA, J. (2002). Anaerobic digestion of organic solid poultry slaughterhouse waste – a review. *Biosource Technology* 83, 13-26.

SALMINEN, E., RINTALA, J., HÄRKÖNEN, J., KUITUNEN, M., HÖGMANDER, H. & OIKARI, A. (2001). Anaerobically digested solid poultry slaughterhouse wastes to be used as fertilizer on agricultural soil. *Bioresource Technology* 78, 81-88.

SHIH, J. C. H. (1987). Ecological benefits of anaerobic digestion. *Poultry Science* 66, 946-950.

SHIH, J. C. H. (1993).Recent development in poultry waste digestion and feather utilization – a review. *Poultry Science* 72; 1617-1620.

STOLBA, A. & WOOD-GUSH, D. G. M. (1989). The behavior of pigs in a seminatural environment. *Animal Production* 48, 419-425.

WEIGAND, R. M., GONYOU, H. W. & CURTIS, S. E. (1994). Pen shape and size: effects on pig behavior and performance. *Applied Animal Behaviour Science* 39, 49-61.

WIESE, J. & HAECK, M. (2006). Instrumentation, control and automation for full-scale manure-based biogas systems. *Water Science and Technology* 54(9), 1-8.

Chapter 2. Overview of Anaerobic Digestion

An important benefit of the anaerobic digestion process is the production of biogas. Biogas contains methane (CH_4) and is produced from the degradation of carbon-based molecules which are abundant in manure. The CH_4 content of biogas generally ranges from 60 to 80% and the biogas has a heating value of approximately 600 – 800 Btu/ft^3 (Roos *et al.* 2004). The remaining gas content is largely composed of carbon dioxide (CO_2), although other components, including hydrogen sulfide (H_2S) are usually present in small quantities. Anaerobic digestion by naturally occurring microorganisms can occur over a wide temperature range. Safley & Westerman (1988, 1989) and Cullimore *et al.* (1985) reported successful initiation of biogas production at temperatures between 3 and 9 ºC, however Sommer *et al.* (2007) report that below 15 °C, CO_2 is the main product of decomposition and the production of CH_4 is not significant. Three practical temperature ranges are generally considered for anaerobic biogas systems: psychrophilic (15 – 25 ºC); mesophilic (30 – 38 ºC); and thermophilic (50 - 60 ºC). These respective temperature ranges facilitate the growth of specific microbes.

Psychrophilic anaerobic digestion systems operate at ambient temperatures (15 to 25°C). Psychrophilic systems contain bacteria that are not easily inhibited by environmental and chemical fluctuations. These stable microorganisms produce high quality biogas (Massé *et al.* 1996, 1997) at low temperatures (20-25 ºC). With proper retention time the volume of gas produced at this temperature range approaches the volume produced at mesophilic biogas plants (Stevens & Schulte 1979). To be effective psychrophilic digestion requires sufficient microbial inoculation, takes longer to initiate (Nohra *et al.* 2003), requiring a retention time which is about twice as long as that of a mesophilic treatment system (Van Lier *et al.* 1997). Some psychrophilic systems retain manure for 250-300 days (Nohra *et al.* 2003). While many digestion systems are constructed to harvest biogas, the main incentive behind development of psychrophilic digestion is often to deodorize manure, conserve nitrogen, and decrease the degree of pathogenic bacterial contamination (Nohra *et al.* 2003).

Mesophilic anaerobic digesters operate in the temperate range of 30 to 38 ºC. Farm level digestion systems designed to capitalize on the use of biogas commonly function within a mesophilic temperature range. Supplemental heat is needed to keep manure within a desired temperature range, and this heat is typically derived from the combustion of CH_4 in a boiler or from engine heat when biogas is used to run an engine. The need for heat transfer equipment increases the capital cost for mesophilic digestion compared to that of psychrophilic digestion. Gas yields can be of high quality and quantity. Under most operational conditions mesophilic anaerobic digesters have faster start up phases than that of psychrophilic systems and produce more biogas (Chynoweth *et al.* 1999).

For thermophilic anaerobic digestion, temperatures must be carefully maintained in the range of 50 to 60 ºC, which requires specialized handling and heating equipment. When compared to other systems the higher temperatures of a thermophilic system enhance chemical reactions and promote bacterial growth leading to faster reaction times, lower retention times, higher gas production, and higher rates of pathogen and weed seed destruction (Kim *et al.* 2002; Kim *et al.* 2006). Advantages of higher loading rates and lower retention times allow thermophilic reactors to be smaller than mesophilic reactors in volume by an order of four times (Hill *et al.* 1985). The net energy produced per unit of mass of waste for both thermophilic and mesophilic digesters is approximately the same, while energy usage by the system as a percentage of energy produced is less for thermophilic digesters (Hill 1983) because lower manure volumes are heated for shorter periods of time. However, thermophilic systems are more sensitive to environmental changes such as temperature fluctuations and chemical concentrations produced during the digestion process (Ahn & Forster 2002; El-Mashad *et al.* 2003; Kim *et al.* 2002) because the number of functional microorganism species that thrive at this temperature range is considerably less than those that survive at lower temperatures (Smith 1980; Wolfe 1979; Ziekus 1977). Only a few specialized microorganism species are available to perform the conversion of organic matter to biogas in this environment which makes these systems acutely susceptible to stress and 'upset' due to high loading rates, loading rate fluctuations, and temperature changes (Hill *et al.* 1985).

References

AHN, J. H. & FORSTER, C. F. (2002). A comparison of mesophilic and thermophilic anaerobic digestion upflow filters treating paper-pulp-liquors. *Process Biochemistry* 38, 256-261.

CHENOWYTH, D. P., WILKIE A. C. & OWENS J. M. (1999). Anaerobic treatment of piggery slurry – Review. *Asian-Australian Journal of Animal Science* 12(4), 607-628.

CULLIMORE, D. R., MAULE, A. & MANSUY, N. (1985). Ambient temperature methanogenesis for pig manure waste lagoons; thermal gradient incubator studies. *Agriculture Wastes* 12(2), 147-157.

EL-MASHAD, H. M., VAN LOON, W. K. P. & ZEEMAN, G. (2003). A model of solar energy utilization in the anaerobic digestion of cattle manure. *Biosystems Engineering* 84, 231-238.

HILL, D. T. (1983). Energy consumption relationships for mesophilic and thermophilic digestion using a swine waste substrate. *Transcripts of the ASAE* 26(13), 841-852.

HILL, D. T., HOLMBERG, R. D. & BOLTE, J. P. (1985). Operating and performance characteristics of scraped swine manure as a thermophilic anaerobic digestion substrate. *Agricultural Wastes* 14, 37-49.

KIM, J. K., OH, B. K., CHUN, Y. N. & KIM, S. W. (2006). Effects of temperature and hydraulic retention time on anaerobic digestion of food waste. *Journal of Bioscience and Bioengineering* 102(4), 328-332.

KIM, M., AHN, Y. H. & SPEECE, R. E. (2002). Comparative process stability and efficiency of anaerobic digestion; mesophilic vs. thermophilic. *Water Resource* 36, 4369-4385.

MASSÉ, D. I., DROSTE, R. L., KENNEDY, K. J., PATNI, N. K. & MUNROE, J. A. (1997). Potential for psychrophilic anaerobic treatment of swine manure in using a sequencing batch reactor. *Canadian Journal of Agricultural Engineering* 39(1), 25-33.

MASSÉ, D. I., PATNI, N. K., DROSTE, R. L. & KENNEDY K. J. (1996). Operation strategies for psychrophilic anaerobic digestion of swine manure slurry in sequencing batch reactors. *Canadian Journal of Civil Engineering* 23, 1285-1294.

NOHRA, J. A., BARRINGTON, S., FRIGON, J. C. & GUIOT, S. R. (2003). In storage psychrophilic digestion of swine slurry. *Resources, Conservation and Recycling* 38, 23-37.

ROOS, K. F., MARTIN, JR, J. H. & MOSER, M. A. (2004). *AgSTAR Handbook; A manual for developing biogas systems at commercial farms in the United States; Second Edition.* U.S. Environmental Protection Agency. EPA-430-B-97-015.

SAFLEY, JR, L. M. & WESTERMAN, P. W. (1988). Biogas production from anaerobic lagoons. *Biological Wastes* 23, 181-193.

SAFLEY, JR, L. M. & WESTERMAN, P. W. (1989). Anaerobic lagoon biogas recovery system. *Biological Wastes* 27, 43-62.

SOMMER, S. G., PETERSEN, S. O., SØRENSEN, P., POULSEN H. D. & MØLLER, H. B. (2007). Methane and carbon dioxide emissions and nitrogen turnover during liquid manure storage. *Nutrient Cycling in Agroecosystems* 78(1), 27-36.

SMITH, P. H. (1980). Studies of methanogenic bacteria in sludge. *Completion report No. EPA-600/2-80-093.* NTIS, Springfield, VA 22161.

STEVENS, M. A. & SCHULTE, D. D. (1979). Low temperature digestion of swine manure. *Journal of the Environmental Engineering Division, ASCE 105(EEI)*, 33-42.

WOLFE, R. S. (1979). Microbial biochemistry of methane—A study in contrasts. *Microbial Biochemistry* 21, 335.

VAN LIER, J. B., REBAC, S. & LETTINGA, G. (1997). High-rate anaerobic wastewater treatment under psychrophilic and Thermophilic conditions. *Water Science and Technology* 35(10), 199-206.

ZIEKUS, J. G. (1977). The biology of methanogenic bacteria. *Bacteriological Reviews* 4(1), 514-541.

Chapter 3. Detailed Description of Anaerobic Digestion

Overview of Anaerobic Degradation Pathways

The speed of degradation of organic material is dependent on the type and composition of organic substrate (Murto *et al.* 2004). Because anaerobic digestion systems commonly receive periodic influxes of organic material a wide range of chemical reactions occur simultaneously. The reactions are dependent on a diversity of microorganisms found within the system. Although much has been written describing anaerobic digestion, the process is not completely understood due to its dynamic nature.

Anaerobic decomposition is sometimes described as a two step process (Figure 3.1) – an acid-production phase followed by an acid-consumption phase (Munch *et al.* 1999). The first phase involves the degradation of organic matter such as proteins, carbohydrates, cellulose, hemicellulose, and lipids. These manure substrates are metabolized by fast growing acidogenic (acid-forming) bacteria, forming short-chained fatty acids, such as acetic, propionic and butyric acids. Other products of this stage are CO_2 and hydrogen (H_2) gases. The second phase of anaerobic digestion involves slower-growing methanogenic (methane-forming) bacteria that utilize fatty acids and hydrogen to form biogas.

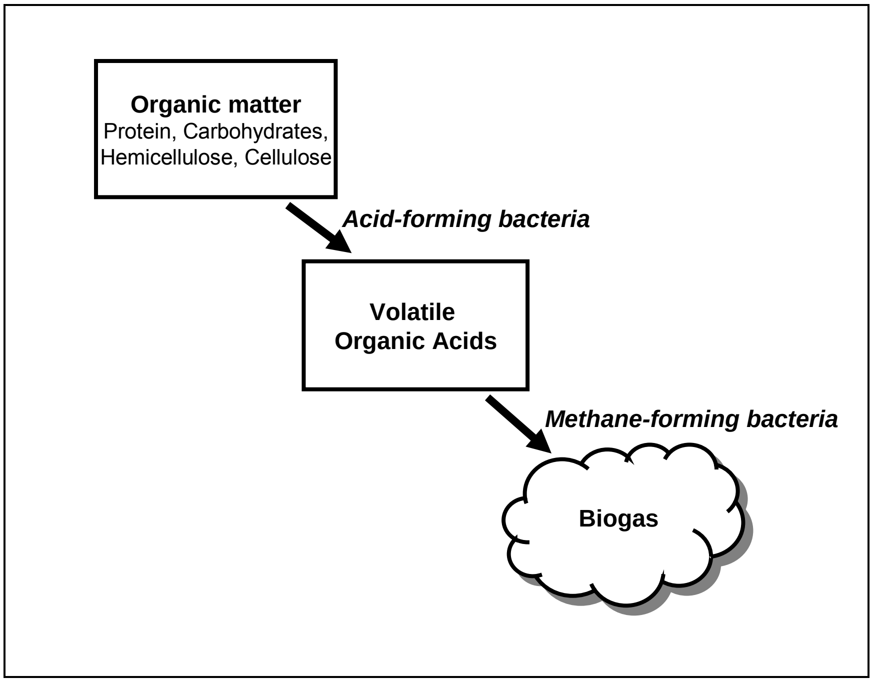


Figure 3.1. Simplified anaerobic digestion reaction diagram.

Massé & Droste (2000) reported that anaerobic digestion is a four-step process involving (1) fermentative or acid forming bacteria; (2) hydrogen-producing acetogenic bacteria; (3) acetoclastic methanogens; and (4) carbon dioxide-reducing methonogenic bacteria that operate to perform hydrolysis, fermentation, and methane production in a series and parallel steps. A reaction diagram is presented in Figure 3.2 (adapted from Pavlostathis & Giraldo-Gomez 1991 and Pavlostathis & Gosset 1986).

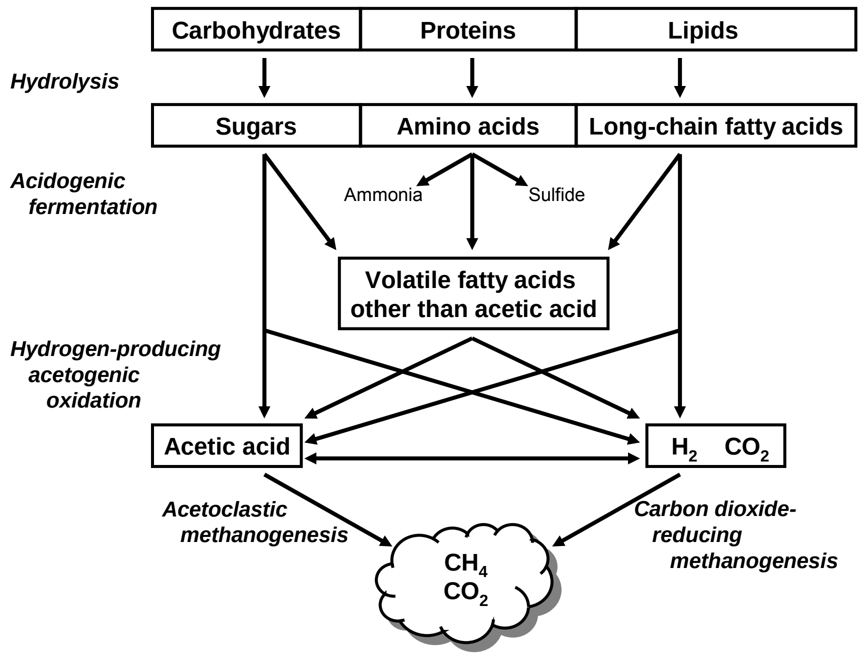


Figure 3.2. Most probable anaerobic digestion reaction scheme. Adapted from Pavlostathis & Giraldo-Gomez (1991) and Pavlostathis & Gosset (1986).

During hydrolysis, fermentative, or acid-forming, proteolytic bacteria produce extra-cellular enzymes that hydrolyze insoluble organic matter such as proteins and polypeptides into amino acids, lipids into long-chain fatty acids (LCFAs) and glycerol, and carbohydrates into soluble sugars (Koster 1989; Pavlostathis & Giraldo-Gomez 1991; Zinder 1984).

Acid-forming bacteria then convert these intermediates into acetic acid, propionic acid, butyric acid, H_2, and CO_2. Two by-products of amino acid fermentation are ammonia and sulfide, while the hydrolytic intermediates formed from LCFAs and alcohols are volatile fatty acids (VFAs), H_2, and CO_2 (Koster 1989; McInerney 1988; Zinder 1984).

Hydrogen-producing acetogenic bacteria metabolize LCFAs, VFAs with three or more carbons and neutral compounds larger than methanol, such as aldehydes, amines and phenols, to acetate, H_2, and CO_2. (Salminen & Rintala 2002). Methanogens ultimately convert acetate, H_2, and CO_2 to CH_4 and CO_2 (Vogels *et al.* 1988; Zinder 1984). *Methanosaeta* and *Methanosarcina* are considered acetoclastic methanogens because they transform acetic acid to CH_4 and CO_2. These two groups of methanogens are found in most anaerobic reactors (McCarty & Mosey 1991). Acetate appears to be the precursor for 70% (Ahring *et al.* 1995) to 75% (Mah et. al 1980) of the methane produced during anaerobic digestion. Several species (*Methanobacterium omelianski, M. formicium, Methanococcus vannilli,* and *Methanosaurcina barkerii*) are considered hydrogen-utilizing methanogens because they reduce CO_2 to CH_4. These bacteria are responsible for about 25% of the CH_4 produced in anaerobic digestion (Mah et. al 1980; Jeris & McCarty 1965).

Gujer & Zehnder (1983) describe anaerobic digestion as a similar four-step process: hydrolysis, acidogenesis, acetogenesis, and methanogenesis. In this model, acidogenesis and acetogenesis stages represent separate pathways compared to that of the decomposition steps listed above. Acidogenesis coverts monomers, such as hydrocarbons and amino acids, to VFAs and acetogenesis convert the VFAs into acetate and H_2.

Others describe the process in two phases as noted in Figure 1, but organize the phases into six independent, sequential and parallel reactions mediated by different groups of biomass under different environments (Gujer & Zehnder 1983, Mata-Alverez 1987, Noykova *et al.* 2002). These authors report that both acetogenesis and methanogenesis, as indicated by Gujer & Zehnder above, have two parallel processes. The six reactions are as follows.

1. Anaerobic hydrolysis of biopolymers (proteins, carbohydrates, lipids, cellulose, hemicellulose) into monomers (amino acids, sugars, long-chain fatty acids).
2. Acidogenic fermentation of amino acids and sugars into volatile fatty acids (VFAs).
3. Acetogenic metabolism of long-chain fatty acids to form acetate and hydrogen.
4. Anaerobic oxidation of intermediate products such as VFAs into acetate and hydrogen.
5. Aceticlastic methanogenesis where acid-utilizing methanogens convert acetate and H_2 into CH_4.
6. Hydrogenotrophic methanogenesis where hydrogen-utilizing methanogens convert CO_2 and H_2 into CH_4

Most studies and models of the anaerobic digestion of organic particles in slurry indicate that at a steady state, hydrolysis is the rate-controlling step in the overall process (Myint *et al.* 2007; Eastman & Ferguson 1981; Gossett & Belser 1982; Pavlostathis & Giraldo-Gomez 1991; Veeken *et al.* 2000; Vavilin *et al.* 2002). Hydrolysis rate is dependent on pH, temperature and concentration of VFAs (Veeken & Hamelers 1999).

Protein and carbohydrate degrading bacteria grow rapidly and these substances are quickly fermented, usually within one day (Bryant 1979). Hemicellulose is also readily degraded, while cellulose is more slowly degraded (Myint *et al.* 2007). Cellulose is the main polymer in many organic wastes and the rate of its degradation is dependent on enzymatic activity (Lee & Fan 1982). Large particles with low surface area to volume ratios hydrolyze more slowly than smaller particles (Vavilin *et al.* 1996).

Bacteria attach to substrate surfaces during hydrolysis. These hydrolytic bacteria release enzymes that produce monomers that can be used by the hydrolytic bacteria themselves or by other bacteria (Vavilin *et al.* 1996; Zavarzin 1986). Microbial daughter cells detach from the substrate particle and enter the liquid phase, eventually attaching to new particles. When particle surfaces are completely covered with microbial cells the surface will be degraded at a constant depth per unit of time (Vavilin *et al.* 1996). The rate of hydrolysis of particles as they enter the anaerobic environment is dependent on two microbial populations; the native organisms found in the manure and the anaerobic microbes found in the anaerobic environment or inoculum (Myint *et al.* 2007).

In the acid-forming stage, acidogenesis typically proceeds at a faster rate than that of hydrolysis, which means that monomers are consumed as they are produced and a change in acidogenic rate usually does not influence the rate of CH_4 production (Vavilin *et al.* 1996). However, Rozzi (1991) reports that if a substrate is easily hydrolyzed then the last step of degradation may be the limiting step. This is because methanogens grow more slowly than upstream acidogens which leads to an organic overload and buildup of metabolic intermediates such as VFA.

Inhibitors of the Biodigestion Process

Anaerobic digestion involves large numbers of diverse microorganisms, numerous serial and parallel reactions, and complex substrates. A number of factors can inhibit anaerobic digestion and thereby impact the rate of methane production. A brief description of inhibiting factors follows.

Inhibition by Long-chain Fatty Acids

The degradation of long-chain fatty acids (LCFA) may be the limiting step in anaerobic digestion for a number of reasons, including the following: (1) LCFA-consuming bacteria are slow growing (Angelidaki & Ahring 1995) which can impede the rate in which LCFA can be consumed and thereby limit conversion of lipids into methane; (2) low solution pH can inhibit anaerobic digestion because LCFA degradation requires a low H_2 partial pressure (Novak & Carlson 1970); (3) LCFAs are toxic to anaerobic microorganisms, particularly acetogens and methanogens (Angelidaki & Ahring 1992; Galbraith *et al.* 1971; Hanaki *et al.* 1981; Hwu *et al.* 1996; Koster & Cramer 1987; Rinzema *et al.* 1994; Roy *et al.* 1985); (4) LCFAs have a tendency to form floating scum (Salminen *et al.* 2001b) which can inhibit digestion by limiting bioavailability and increasing toxicity (Hobson & Wheatley 1988; Pagilla *et al.* 1997); and (5) bacterial degradation of LCFAs begins with adsorption of LCFA by the cell and this can be inhibiting depending on type of bacteria, size of LCFA, whether the LCFA is saturated or unsaturated, and concentration of LCFAs (Salminen & Rintala 2002).

Inhibition by Ammonia

Ammonia produced in the degradation of protein can inhibit anaerobic methanogens during the anaerobic digestion process (Angelidaki & Ahring 1993; DeBaere *et al.* 1984; Hansen *et al.* 1998; Hashimoto 1986; McCarty & McKinney 1961; Melbinger & Donnellon 1971; Wiegant & Zeeman 1986). Free, or unionized NH_3, is responsible for most toxic effects, although NH_4^+ is toxic at higher concentrations (McCarty & McKinney 1961; DeBaere *et al.* 1984). Methanogenic populations may adapt over time to NH_3 concentrations several times the initial threshold inhibition level (Koster & Lettinga 1988; Parkin *et al.* 1983). This is likely a reason for variation in reports of inhibiting ammonia threshold levels. Adaptation of the population results from growth of new acetate-utilizing methanogens that tolerate higher NH_3 levels rather than changes in the methanogens already present (postulated by Angelidaki & Ahring 1993).

Free ammonia concentration depends on three parameters: total ammonia concentration, temperature, and pH (Hansen *et al.* 1998).Thermophilic systems are more easily inhibited by NH_3 than mesophilic systems (Parkin & Miller 1983: Angelidaki & Ahring 1994). Higher temperatures increase NH_3 concentration in solution and the biogas process is more sensitive to NH_3 as pH values increase (Koster 1986); an increase in pH from 7 to 8 can lead to an 8 fold increase in free ammonia (Koster 1986). Angelidaki & Ahring (1993) and Angelidaki *et al.* (1993) found that the interactions between NH_3, VFAs and pH can lead to an "inhibited steady state", where biogas is steadily produced but at a low rate.

Inhibition by Sulfate

Proteins found in animal wastes contribute to manure sulfate levels. In the anaerobic digestion process sulfates are used as electron acceptors by sulfate-reducing bacteria (Hao *et al.* 1996; Petersen & Ahring 1992). Because sulfate reduction is more energetically favorable than CH_4 production, the sulfate reducing bacteria will compete with methanogens for the use of H_2 and CO_2 (Hao *et al.* 1996). Furthermore, sulfate will metabolize into sulfide, which can inhibit biogas production at concentrations of 50 mg S^{2-}/l (Karhadkar *et al.* 1987; Parkin et. al 1983), with severe inhibition observed when concentrations exceed 150 to 200 mg S^{2-}/l (Karhadkar *et al.* 1987). In a study that confirmed a combined effect of inhibition by ammonia and sulfide, Hansen *et al.* (1999) demonstrated that a concentration of 23 mg S/l inhibited CH_4 production by 40% in swine manure that contained a high NH_3 concentration.

Inhibition by Low pH

Not only can the anaerobic digestion process be inhibited by increased ammonia associated with increasing pH, but decreasing pH can also restrain biogas production. Acid-consuming bacteria are more inhibited than acid producing bacteria by decreases in pH (Anderson & Yang 1992). This can cause further acid accumulation and lead to process failure. Resistance to pH change is dependent upon the buffering capacity of the substrate (Rozzi 1991).

Inhibition from Short Retention Time

The length of time that material remains in the anaerobic system is referred to as retention time. Hydraulic retention time (HRT) refers to time that the liquid portion of the digestate material is in the digestion vessel, while solid retention time (SRT) refers to the time that the solid portion of the digestate material is in the vessel. If HRT or SRT are short the material targeted for digestion could pass through the digestion system before degradation and maximum methane production is achieved. A major challenge with digestion systems is to maintain adequate SRT while minimizing HRT (Boopathy 1998). The system must be large enough to provide sufficient solids retention time but small enough to be economically practical.

Inhibition from Antibiotic Use

A common concern with farm level anaerobic systems is that antibiotic administration to livestock may inhibit the digestion process. This occurs when antibiotics, administered to promote health and weight gain, are incompletely metabolized by the livestock and thus excreted in urine or feces. Separate studies have reported similar ranges of methane inhibition due to antibiotic use. Sanze *et al.* (1996) reported a CH_4 reducing effect from antibiotic use of 25-45%, Massé *et al.* (2000) found reduction range of 20-45% and Loftin *et al.* (2005) reported the decrease to be 25-35%.

Biogas Content and Production

The CH_4 content of biogas is dependent upon a number of factors including influent content, digestion environment and hydraulic retention time of the organic substrate being digested and generally ranges from 60 to 80% (Roos *et al.* 2004). Technologies are available to "clean" biogas to increase CH_4 concentration from the expected 60-80%. Gas yield is affected by many factors including operating temperature, retention time, loading amounts and frequency, digester design, and pretreatment of raw materials (Berglund & Börjesson 2006).

Gas production levels are commonly reported in one of three ways; volume of methane produced per volume unit of influent, volumetric methane produced per reactor volume per day, or volume of methane produced per unit weight of volatile solids (VS) added to the reactor. The theoretical maximum CH_4 production from pure fatty acid, protein, and starch substrates is 1.5, 0.9, and 0.8 L/g, respectively (Hawkes & Hawkes 1987).

Manure Changes and Digestate Qualities

Anaerobic digestion transforms the influent feedstock. The effluent is commonly called digestate. Digestate contains all non-degradable substances present in the original feedstock (Lantz *et al.* 2007). Most plant nutrients found in the feedstock will remain in the digestate (Lantz *et al.* 2007; Salminen *et al.* 2001a; Shih 1987, 1993; Sundradjat 1990; Vermeulen *et al.* 1992). Anaerobic digestion is a process that stabilizes the organic matter in a feedstock (Safley & Westerman 1994). The degradation process increases plant availability of nitrogen, which enhances fertilization efficiency of the feedstock (Lantz *et al.* 2007). Manures contain organic particles that are both soluble and particulate in nature (Massé 1995) that are fractionated through anaerobic digestion, thus a reduction in particle size occurs. Because particle sizes are reduced and carbon is removed, the anaerobic digestion reduces manure volume (Lantz *et al.* 2007; Svärd & la Cour Jansen 2003) although the amount of reduction remains largely unreported.

References

AHRING, B. K., SANDBERG, M. & ANGELIDAKI, I. (1995). Volatile fatty acids as indicators of process imbalance in anaerobic digesters. *Applied Microbiology and Biotechnology* 41, 559-565.

ANDERSON, G. K. & YANG, G. (1992). pH control in anaerobic treatment of industrial wastewaters. *Journal of Environmental Engineering* 118(4), 551-567.

ANGELIDAKI, I. & AHRING, B. K. (1992). Effect of free long-chain fatty acids on thermophilic anaerobic digestion. *Applied Microbiology and Biotechnology* 37, 808-812.

ANGELIDAKI, I. & AHRING, B. K. (1993). Thermophilic anaerobic digestion of livestock waste: effect of ammonia. *Applied Microbiology and Biotechnology* 38, 560-564.

ANGELIDAKI, I. & AHRING, B. K. (1994). Anaerobic thermophilic digestion of manure at different ammonia loads: effect of temperature. *Water Resources* 28, 727-731.

ANGELIDAKI, I. & AHRING, B. K. (1995). Establishment and characterization of an anaerobic thermophilic (55 ºC) enrichment culture degrading long-chain fatty acids. *Applied Environmental Microbiology* 61, 2442-2445.

ANGELIDAKI, I., ELLEGAARD, L. & AHRING, B. K. (1993). A mathematical model for dynamic simulation of anaerobic digestion of complex substrates: focusing on ammonia inhibition. *Biotechnology and Bioengineering* 42, 159-166.

BERGLUND, M., & BÖRJESSON, P. (2006). Assessment of energy performance in the life-cycle of biogas production. *Biomass and Bioenergy* 30, 254-266.

BOOPATHY, R. (1998). Biological treatment of swine waste using anaerobic baffled reactors. *Bioresource Technology* 64, 1-6.

BRYANT, M. P. (1979). Microbial methane production-theoretical aspects. *Journal of Animal Science* 48(1), 193-201.

DE BAERE, L. A., DEVOCHT, M., VAN ASSCHE, P. & VERSTRAETE, W. (1984). Influence of high NaCl and NH_4Cl salt levels on methanogenic associations. *Water Resources* 18, 543-548.

EASTMAN, J. A. & FERGUSON, J. G. (1981). Solubilization of particulate organic carbon during the acid phase of anaerobic digestion. *Journal of Water Pollution Control Federation* 53, 352.

GALBRAITH, H., MILLER, T. B., PATON, A. M. & THOMPSON, J. K. (1971). Antibacterial activity of long chain fatty acids and the reversal with calcium, magnesium, ergocalciferol and cholesterol. *Journal of Applied Bacteriology* 34, 808-813.

GOSSETT, J. M. & BELSER, R. L. (1982). Anaerobic digestion of waste activated sludge. *ASCE Journal of Environmental Engineering* 108, 1101.

GUJER, W. & ZEHNDER, A. J. B. (1983). Conversion process in anaerobic digestion. *Water Science and Technology* 15, 127-167.
HANAKI, K., MATSUO, T. & NAGASE, M. (1981). Mechanism of inhibition caused by long-chain fatty acids in anaerobic digestion process. *Biotechnology and Bioengineering* 23, 1591-1610.
HANSEN, K. H., ANGELIDAKI, I. & AHRING, B. K. (1998). Anaerobic digestion of swine manure inhibition by ammonia. *Water Resource* 32(1), 5-12.
HANSEN, K. H., ANGELIDAKI, I. & AHRING, B. K. (1999). Improving thermophilic anaerobic digestion of swine manure. *Water Resource* 33(8), 1805-1810.
HAO, O. J., CHEN, J. M., HUANG, L. & BURLASS, R. L. (1996). Sulfate reducing bacteria. *Environmental Science Technology* 26, 155-187.
HASHIMOTO, A. G. (1986). Ammonia inhibition of methanogenesis from cattle wastes. *Agricultural Wastes* 17, 241-261.
HAWKES, F. R. & HAWKES, D. L. (1987). Anaerobic digestion. In: *Bu'lock, J., Kristiansen, B. (Eds.), Basic Biotechnology*, Academic Press, London, pp. 337-358.
HOBSON, P. N. & WHEATLEY, A. D. (1988). *Anaerobic Digestion: Modern Theory and Practice.* Elsevier, London.
HWU, C. -S., DONLON, B. & LETTINGA, G. (1996). Comparative toxicity of long-chain fatty acid to anaerobic sludges from various origins. *Water Science and Technology* 34(5-6), 351-358.
JERIS, J. S. & MCCARTY, P. L. (1965). The biochemistry of methane fermentation using C^{14}. *Transcriptions of the Journal of the Water Pollution Control Federation* 49(10), 178-192.
KARHADKAR, P. P., AUDIC, J. -M., FAUP G. M. & KHANNA, P. (1987). Sulfide and sulfate inhibition of methanogenesis. *Water Resource* 21, 1061-1066.
KOSTER, I. W. (1986). Characteristics of the pH-influenced adaptation of methanogenic sludge to ammonium toxicity. *Journal of Chemical Technology and Biotechnology* 36, 445-455.
KOSTER, I. W. (1989). *Toxicity in anaerobic digestion with emphasis on the effect of ammonia, sulfide and long-chain fatty acids on methanogenesis.* Ph.D. Thesis, Wageningen Agricultural University.
KOSTER, I. W. & CRAMER, A. (1987). Inhibition of methanogenesis from acetate in granular sludge by long-chain fatty acids. *Applied Environmental Engineering and Microbiology* 53, 403-409.
KOSTER, I. W. & LETTINGA, G. (1988). Anaerobic digestion at extreme ammonia concentrations. *Biological Wastes* 25, 51-59.
LANTZ, M., SVENSSON, M., BJÖRNSSON, L. & BÖRJESSON, P. (2007). The prospects for an expansion of biogas systems in Sweden – incentives, barriers and potentials. *Energy Policy* 35, 1830-1843.
LEE, Y. H. & FAN, L. T. (1982). Kinetic study of enzymatic hydrolysis of insoluble cellulose: analysis of the initial rates. *Biotechnology and Bioengineering* 24, 2883-2406.
LOFTIN, K. A., HENNY, C., ADAMS, C. D., SURAMPALI, R. & MORMILE, M. R. (2005). Inhibition of microbial metabolism in anaerobic lagoons by selected sulfonamides, tetracyclines, lincomycin, and tylosin tartrate. *Environmental Technology and Chemistry* 24(4), 782-788.

MAH, R. A., SMITH, M. R., FERGUSON, T. & ZINDER, S. (1980). Methanogenesis from H_2-CO_2, methanol acetate by *Methanosarcina. In Microbial Growth on C_1 Compounds* (Ed H. Dalton), pp. 131-142. London.

MASSÉ, D. I. (1995). *Psychrophilic anaerobic digestion of swine manure slurry in intermittently fed sequencing batch reactors.* Thesis, Department of Civil Engineering, University of Ottawa, Ottawa, ON.

MASSÉ, D. I. & DROSTE, R. L. (2000). Comprehensive model of anaerobic digestion of swine manure slurry in a sequencing batch reactor. *Water Resources* 34(12), 3087-3106.

MASSÉ, D. I., LU, D., MASSE, L. & DROSTE, R. L. (2000). Effect of antibiotics on psychrophilic anaerobic digestion of swine manure slurry in sequencing batch reactors. *Bioresource Technology* 75, 205-211.

MATA-ALVAREZ, J. (1987). A dynamic simulation of a two-phase anaerobic digestion system for solid wastes. *Biotechnology and Bioengineering* 30, 844-851.

MCCARTY, P. L. & MCKINNEY, R. E. (1961). Salt toxicity in anaerobic digestion. *Journal of the Water Pollution Control Federation* 33, 399-415.

MCCARTY, P. L. & MOSEY, F. E. (1991). Modeling of an anaerobic digestion process (a discussion of concepts). *Water Science and Technology* 24(8), 17-33.

MCINERNEY, M. J. (1988). Anaerobic hydrolysis and fermentation of fats and proteins. In: *Biology of Anaerobic Microorganisms* (Ed J. B. Zender), Wiley, New York, pp. 373-415.

MELBINGER, N. R. & DONNELLON, J. (1971). Toxic effects of ammonia nitrogen in high-rate digestion. *Journal of the Water Pollution Control Federation* 43, 1658-1667.

MUNCH, V. E., KELLER, J., LANT, P. & NEWELL, R. (1999). Mathematical modeling of prefermenters-I. model development and verification. *Water Resources* 33(12), 2757-2768.

MURTO, M., BJÖRNSSON, L. & MATTIASSON, B. (2004). Impact of food industrial waste on anaerobic co-digestion of sewage sludge and pig manure. *Journal of Environmental Management* 70, 101-107.

MYINT, M., NIRMALAKHANDAN, N. & SPEECE, R. E. (2007). Anaerobic fermentation of cattle manure: Modeling of hydrolysis and acidogenesis. *Water Research* 41, 323-332.

NOVAK, J. T. & CARLSON, D. A. (1970). The kinetics of anaerobic long chain fatty acid degradation. *Journal of the Water Pollution Control Federation* 42, 1932-1943.

NOYKOVA, N., MÜLLER, T. G., GYLLENBERG, M. & TIMMER, J. (2002). Quantitative analyses of anaerobic wastewater treatment processes: identifiability and parameter estimation. *Biotechnology and Bioengineering* 78(1), 89-103.

PAGILLA, K. R., CRANY, K. C. & KIDO, W. H. (1997). Causes and effects of foaming in anaerobic sludge digesters. In: *Processing of the 8th International Conference on Anaerobic Digestion*. Scadai, vol. 2, pp. 239-246.

PARKIN, G. F. & MILLER, S. W. (1983). Response of methane fermentation to continuous addition of selected industrial toxicants. In: *Proceedings of the 37th Industrial Waste Conference*, pp. 726-743.

PARKIN, G. F., SPEECE, R. E., YANG, C. H. J. & KOCHER, W. M. (1983). Response of methane fermentation systems to industrial toxicants. *Journal of the Water Pollution Control Federation* 55, 44-53.

PAVLOSTATHIS, S. G. & GIRALDO-GOMEZ, E. (1991). Kinetics of anaerobic treatment: a critical review. Critical Review. *Environmental Control* 21(5 and 6), 411-490.

PAVLOSTATHIS, S. G. & GOSSET, J. M. (1986). A kinetics model for anaerobic digestion of biological sludge. *Biotechnology and Bioengineering* 38, 1519-1530.

PETERSEN, S. P. & AHRING, B. K. (1992). The influence of sulphate on substrate utilization in a thermophilic sewage reactor. *Applied Microbiology and Biotechnology* 36, 805-809.

RINZEMA, A., BOONE, M., VAN KNIPPENBERG, K. & LETTINGA, G. (1994). Bacterial effect of long chain fatty acids in anaerobic digestion. *Water Environment Resources* 66, 40-49.

ROOS, K. F., MARTIN, JR, J. H. & MOSER, M. A. (2004). *AgSTAR Handbook; A manual for developing biogas systems at commercial farms in the United States; Second Edition.* U.S. Environmental Protection Agency. EPA-430-B-97-015.

ROY, F., ALBAGNAC, G. & SAMAIN, E. (1985). Influence of calcium addition on growth of highly purified syntrophic cultures degrading long-chain fatty acids. *Applied Environmental Microbiology* 49, 702-705.

ROZZI, A. (1991). Alkalinity considerations with respect to anaerobic digesters. *Mededelingen Faculteit Landbouwwetenschappen Rijksuniversiteit Gent* 56(4a), 1499-1514.

SAFLEY, JR., L.M., & WESTERMAN, P. W. (1994). Low-temperature digestion of dairy and swine manure. *Bioresource Technology* 47, 165-171.

SALMINEN, E., RINTALA, J., HÄRKÖNEN, J., KUITUNEN, M., HÖGMANDER, H. & OIKARI, A. (2001a). Anaerobically digested solid poultry slaughterhouse wastes to be used as fertilizer on agricultural soil. *Bioresource Technology* 78, 81-88.

SALMINEN, E., EINOLA, J. & RINTALA, J. (2001b). Characterization and anaerobic batch degradation of materials accumulating in anaerobic digesters treating poultry slaughterhouse waste. *Environmental Technology* 22, 577-585.

SALMINEN, E. & RINTALA, J. (2002). Anaerobic digestion of organic solid poultry slaughterhouse waste – a review. *Biosource Technology* 83, 13-26.

SANZ, J. L., RODRIGUEZ, N. & AMILS, R. (1996). The action of antibiotics on the anaerobic digestion process. *Applied Microbiology and Biotechnology* 46, 587-592.

SHIH, J. C. H. (1987). Ecological benefits of anaerobic digestion. *Poultry Science* 66, 946-950.

SHIH, J. C. H. (1993).Recent development in poultry waste digestion and feather utilization – a review. *Poultry Science* 72; 1617-1620.

SUNDRADJAT, R. (1990). *Transformations and uses of digested solid organic residues.* Ph.D. Thesis, Rijksuniversiteit Gent.

SVÄRD, Å. & LA COUR JANSEN, J. (2003). *Svenska biogasanläggningar – erfarenhetssammanställning och rapporteringssystem (Biogas plants in Sweden – compilation of experiences and reporting system).* Report 2003:14. Swedish Water Association (in Swedish with English summary).

VAVILIN, V. A., RYTOV, S. V. & LOKSHINA, L. YA. (1996). A description of hydrolysis kinetics in anaerobic degradation of particulate organic matter. *Bioresource Technology* 56, 229-237.

VAVILIN, V. A., RYTOV, S. V., LOKSHINA, L. YA., PAVLOSTATHIS, S.G. & BARLAZ, M.A. (2002). Distribution model of solid waste digestion-effects of leachate recirculation and pH adjustment. *Biotechnology and Bioengineering* 81, 66-73

VEEKEN, A. & HAMELERS, B. (1999). Effect of temperature on hydrolysis rate of selected biowaste components. *Bioresource Technology* 69, 249-254.

VEEKEN, A., KALYUZHNYI, S., SCHARFF, H. & HAMELERS, B. (2000). Effects of pH and VFA on hydrolysis and organic waste. *ASCE Journal of Environmental Engineering* 126, 1076-1081.

VERMEULEN, J., HUYSMANS, A., CRESPO, M., VAN LIERDE, A., DE RYCKE, A. & VERSTRAETE, A. (1992). Processing of biowaste by anaerobic composting to plant growth substrates. In: *Proceedings of the International Symposium on Anaerobic Digestion of Solid Waste, Venice (14–17 April, 1992)*, pp. 147–157.

VOGELS, G. D., KELTJENS, J. T. & VAN DER DRIFT, C. (1988). Biochemistry of methane production. In: *Biology of Anaerobic Microorganisms.* Wiley (Ed J. B. Zender), New York, pp. 707-770.

WIEGANT, W. M. & ZEEMAN, G. (1986). The mechanism of ammonia inhibition in thermophilic digestion of livestock wastes. *Agricultural Wastes* 16, 243-253.

ZAVARZIN, G. A. (1986). Trophic relationships in methanogenic consortium. *Izvestia AN SSSR, Ser. Biology No.3*, 341-360.

ZINDER (1984). S. H. Zinder, Microbiology of anaerobic conversion of organic wastes to methane: recent developments. *ASM News* 50 **(**1984), pp. 294–298.

Chapter 4. Application and Benefits of Anaerobic Digestion of Manure

Digester Designs

The following is a brief description of the four basic farm level anaerobic digestion vessel designs, as described by Roos *et al.* (2004).

1. *Covered lagoon digesters.* Covered lagoon digestion systems are ambient (psychrophilic) temperature systems that require manure with solid content of 3% or less. The covered manure storage vessel maintains an anaerobic environment. These systems are typically the largest type of digester with the longest hydraulic retention time (HRT).

2. *Complete mix digesters.* These digesters consist of an engineered digestion vessel designed to handle manure slurries with a solid content from 3-10%. A mixing system enhances bacterial contact with organic matter. Often, supplemental heat is added for operation in the mesophilic temperature range, which promotes bacterial growth and shorter HRT.

3. *Plug flow digesters.* Plug flow systems use a tank or vessel that receives manure on one end and discharges from the opposite end with no mixing or agitation. These systems are typically heated to a mesophilic temperature and require slurry with solid content of 11-13%.

   One system variation, called modified three-stage methane fermentation, uses influent management timing to control HRT in a manner that allows three separate vessels to each contain specialized microbial populations and operate in series. The first vessel conducts semi-anaerobic hydrolysis, the second vessel, anaerobic acidogenesis and the final vessel strictly performs anaerobic methanogenesis. These systems decrease necessary HRT by increasing the rates of hydrolysis, acidogenesis, and methanogenesis, without affecting pH. A high CH_4 yield has been observed from this treatment (Kim *et al.* 2006; Kim *et al.* 2000).

4. *Fixed film digesters.* The term fixed film refers to a medium placed in the vessel on which bacteria can grow. Dilute manures with solid content of 3% or less are passed across (or through) the medium in these systems. Some examples of media used in fixed film digestion are rope, plastic mesh, stones and plastic beads. While other systems rely solely on suspended microbial growth, these systems also feature attached microbial growth.

All but the plug flow design can also operate with all manure being removed, followed by subsequent refilling of the vessel, termed batch flow. All of these system types can receive doses of manure influent at regular intervals, termed continuous flow.

Environmental Benefits of Anaerobic Digestion

Biodegradation of organic matter leads to a significant improvement in resource recovery and reduction of environmental impacts compared to traditional agriculture practices and current manure handling systems (Lantz *et al.* 2007; Sundberg *et al.* 1997). Because anaerobic digestion has low sludge production and energy requirements it is widely used to remove organic matter from high strength industrial and municipal wastewaters (Bernet *et al.* 2000). The process offers many environmental benefits including renewable energy (DeBaere 2000), and possible nutrient recycling and reduction of waste volumes (Murto *et al.* 2004; Ghosh *et al.* 1975; Hawkes & Hawkes 1987; van Lier *et al.* 2001). A discussion of some of the environmental benefits of anaerobic digestion follows.

Odor Reduction

Manure is a complex mixture of undigested dietary residues, endogenous secretions, and bacterial cells; these organic compounds include volatile fatty acids, alcohols, aromatic compounds, amides (including ammonia), and sulfides produced by the animal during digestion that, along with compounds formed from microbial activity during manure storage may produce odorous compounds (Mackie *et al.* 1998). Odors arise primarily from anaerobic degradation of manure and are divided into four principle classes of odor compounds; branched- and straight-chain VFA, ammonia and volatile amines, indoles and phenols, and volatile sulfur-containing compounds (Mackie *et al.* 1998). O'Neill & Phillips (1992) report that swine manure odors contain over 160 chemical compounds. When manure surfaces are exposed to the atmosphere volatile products and intermediates are emitted into the environment (Mackie, *et al.* 1998).

The anaerobic digestion process reduces the odorous potential of manure by metabolizing volatile organic compounds. When biogas is collected from an anaerobic system the gas can be combusted or treated for removal of odors. Because the anaerobic digestion process can take a long time, longer hydraulic retention times can lead to more complete digestion and deodorization of manure. Fischer *et al.* (1984) found that the odor from swine manure was significantly reduced at 20 days HRT while manure treated with a 10 day HRT was not. Figure 4.1 schematically demonstrates how anaerobic digestion helps to deodorize manure.

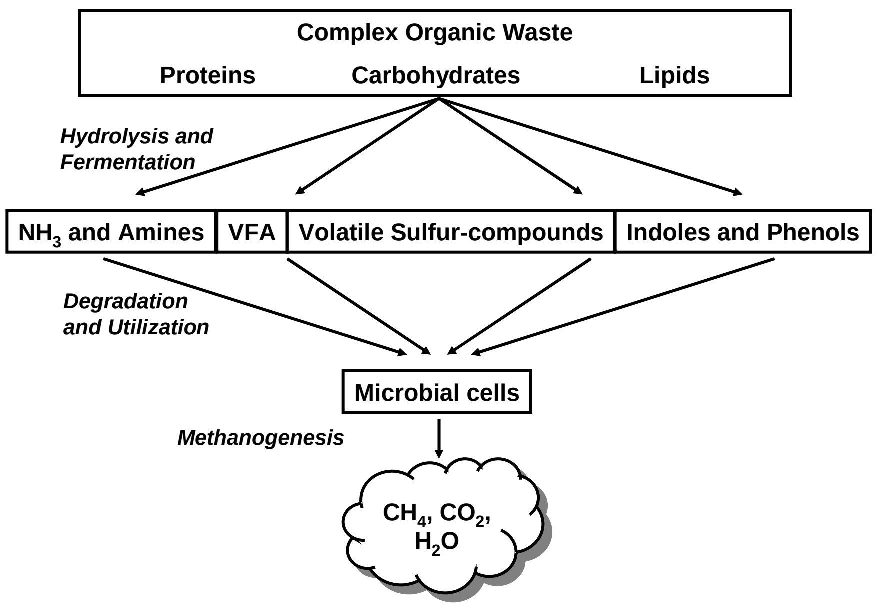


Figure 4.1. Anaerobic degradation of odorous manure compounds.
Adapted from Mackie *et al.* (1998).

Agricultural policies that favor the reduction of odors may provide incentives to livestock producers to implement anaerobic manure treatment systems. In Pennsylvania, regulations are under development that will require new or expanding livestock facilities, which meet certain animal density definitions when populated, to have a state approved Odor Management Plan in place prior to stocking the facility. One of the proposed odor management tools that producers can adopt to minimize odor emissions is anaerobic digestion (PA SCC 2007).

Manure Nutrient Benefits

Loria & Sawyer (2005) conducted field application tests of raw and digested manure, concluding that the anaerobic digestion process does not substantially affect manure nutrient supply and that digested manure can provide similar plant available nitrogen (N) and phosphorus (P) as expected from the raw manure. Others report that anaerobic digestion increases the plant availability of nitrogen and enhances fertilization efficiency of treated feedstock (Lantz *et al.* 2007; Börjesson & Berglund 2003, 2007; Nielsen *et al.* 2002). Stabilization of organic matter during the digestion process is shown to reduce N leaching, while improving plant use of N (Lantz *et al.* 2007; Börjesson & Berglund 2003, 2007; Svenson *et al.* 2005). Organic-bound N is converted into ammonium (NH_4^+) available to plants (Börjesson & Berglund 2007), with the amount of NH_4^+ representing 70 and 85% of total nitrogen content in digested and undigested manure, respectively (Sommer *et al.* 2001). Due to a positive charge NH_4^+ is more likely to be held in soil by negatively charged soil particles than other forms of nitrogen, such as nitrate (NO_3^-) that may leach. Berglund & Börjesson (2006) report that because of this change in nitrogen distribution, applying digested swine manure in place of undigested swine manure reduces nitrogen leakage to the environment, and Blomqvist (1993) reported a leaching reduction of about 20%. Because the potential of nitrogen loss is lower, one of the benefits of anaerobic digestion is the possibility to replace and reduce use of mineral-based fertilizers (Lantz *et al.* 2007; Galloway 1998; Galloway et al 2004; Isermann &

Isermann 1998). However, in soil, NH_4^+ can oxidize to NO_3^- through the action of nitrifying bacteria, which can lead to a rapid accumulation of NO_3^- in the soil following land application (Whalen & DeBerardinis 2007). NO_3^- has a greater potential for leaching than NH_4^+.

Reduction of Chemical Oxygen Demand

Chemical Oxygen Demand (COD) is a measure of the oxygen required to oxidize all organic compounds found in a sample of water or wastewater. The COD value assigned to wastewater or manure is an indicator of the pollution potential of the waste should it enter a waterway; lower COD implies a lower pollution impact potential. Anaerobic digestion reduces COD of manure. At the mesophilic temperature of 35 ºC, Boopathy (1998) found that digestion of swine manure produced a COD reduction of 70-78%, while Andara & Esteban (1999) reported a COD reduction of 61-65% in swine manure.

Manure contains both particulate and soluble organic compounds (Massé & Droste 2000). The COD of the soluble organic fraction of manure is considered the Soluble Chemical Oxygen Demand (SCOD). Based on composition reports, the SCOD of swine manure has been found to be composed mainly of carbohydrates (Massé 1995). In a psychrophilic digestion experiment Massé *et al.* (2003) found that reductions in SCOD and COD of swine manure were 84-96% and 41-83%, respectively. The authors noted that less SCOD reduction occurred at lower temperatures within the psychrophilic range.

COD reduction is dependent upon the completeness of the anaerobic digestion process and can be impacted by HRT. When HRT was decreased in two anaerobic swine manure treatment systems the COD removal efficiencies of the systems decreased from above 95% to 57 and 61% (Lo *et al.* 1994).

Pathogen Reduction

Anaerobic digestion can help reduce manure pathogens. Swine manure contains viral, bacterial and protozoan pathogens, which can cause human and livestock disease (Black *et al.* 1982; Lund & Niessen 1983; Marti *et al.* 1983). Studies are limited in pathogen reduction efficiencies of digestion systems (Côté *et al.* 2006). Thermophilic (50 ºC) anaerobic digestion systems effectively inactivate enteric pathogens (Hashimoto 1983) and may destroy all viruses with appropriate HRT (Salminen & Rintala 2002). Destruction of manure pathogens is more effective for thermophilic than mesophilic systems, with complete eradication of fecal coliforms and salmonella observed at thermophilic (50 ºC) digestion, while those pathogens were only partially destroyed under mesophilic (35 ºC) digestion (Shih 1987; Bendixen 1994). *E.coli* and *Salmonella* can survive digestion temperatures of 20 and 35 ºC (Kumar *et al.* 1999), but are successfully removed at temperatures of 37 and 54.9 ºC (Duarte *et al.* 1992). However, even psychrophilic systems (20 ºC for 20 days) can significantly reduce total coliforms (97.94-100%), *E.coli* (99.67-100%) and indigenous strains of *Salmonella*, *Cryptosporidium* and *Giardia* (Côté *et al.* 2006). Besides temperature, the destruction of pathogens in anaerobic treatment systems is dependent upon HRT, with longer retention time yielding greater bacterial and viral destruction (Kun *et al.* 1989).

Greenhouse Gas Reduction and Fossil Fuel Replacement

Manure is commonly stored in open liquid slurry systems until a time when the manure can be land-applied as fertilizer for crop production. Increasing use of liquid systems can be attributed to an industry shift toward farms with larger animal numbers and facilities, as well as changing manure application regulations that limit frequent manure application and cause a need for longer storage periods. Liquid systems contribute to CH_4 emissions, while manure stored in solid form produces little CH_4 (U.S. Environmental Protection Agency 2006a).

Liquid manure systems may be aerobic or anaerobic in nature. Some systems will contain both aerobic and anaerobic zones. The microbial populations in aerobic systems produce significant amounts of CO_2 at the slurry-air interface (Møller *et al.* 2004), while anaerobic systems release both CH_4 and CO_2.

Estimated annual amounts of CH_4 emitted from all agricultural manure in the United States are 1966 Gg CH_4 [gigagrams (10^9) methane], or 41.3 Tg CO_2 equivalent [teragrams (10^{12}) carbon dioxide]. These values represent 25.6% of all agricultural CH_4 emissions and 1.9% of total CH_4 emissions from the United States. Of these emissions most originate from dairy (851 Gg CH_4 and 17.9 Tg CO_2 Eq.) and swine (852 Gg CH_4 and 17.9 Tg CO_2 Eq.) manures with each of these agricultural sectors contributing nearly the same amount of emissions, approximately 43.3% of the US manure CH_4 emissions each (U.S. Environmental Protection Agency 2006a).

Methane is a greenhouse gas that is 23 times more potent than carbon dioxide on a 100 year timeline (IPCC 2001). By collecting the methane emitted from manure storage and oxidizing the CH_4 molecule through combustion to form CO_2 and H_2O the overall impact on greenhouse gases is reduced. Methane emissions that would have come from manure storages are avoided. Under Danish conditions, CH_4 emissions have been reduced, on average, by 1.6 kg CH_4/ton (reduced from 3.1 to 1.5 kg CH_4/ton) of swine slurry when digested (Sommer *et al.* 2001). Little CH_4 emissions occur during and after land application of manure because the handling of manure in this manner is an aerobic process (U.S. Environmental Protection Agency 2006b).

When collected CH_4 is combusted and used for heat or energy purposes, a replacement of fossil fuel use occurs, thereby reducing carbon dioxide emissions from the fossil fuels (Salminen & Rintala 2002). Replacing fossil fuels with biogas reduces the emissions of greenhouse gases, nitrogen oxides, hydrocarbons, and particulates (Börjesson & Berglund 2006). Since the carbon in the CH_4 has come from organic sources the combusted carbon is part of the short-term (biological) carbon cycle, whereas carbon from fossil fuel is released from the long-term (geological) carbon cycle.

Another greenhouse gas emitted from manure is nitrous oxide (N_2O). The Global Warming Potential (GWP) of N_2O is 296 for a 100-year time horizon (IPCC 2001), meaning that a molecule of gaseous N_2O causes 296 times the impact of a molecule of CO_2 on global warming. Agricultural contributions to N_2O largely come from soil disturbances associated with crop management and manure application. Estimated annual amounts of N_2O emitted from agricultural manure storage in the United States are 31 Gg N_2O, or 9.5 Tg CO_2 equivalent. These values represent roughly 2.5% of all U.S. agricultural N_2O emissions from all sources. Agricultural soil management accounts for 1178 Gg N_2O, or 365.1 Tg CO_2 Eq. which is over 97.3% of all U.S. agricultural emissions. It is important to note that the soil management category includes N_2O emissions from land-applied manure (U.S. Environmental Protection Agency 2006a). Anaerobic digestion decreases the emission of N_2O from land-applied manure compared to non-digested manure (Petersen 1999) because the organic matter remaining in digested manure is less likely to undergo microbial decomposition than that found in untreated manure. Furthermore the smaller organic molecules found in the decomposed organic matter provide less energy to support the growth of nitrous oxide forming microorganisms (Sommer *et al.* 2000).

Using Biogas as an Alternative Fuel to Produce Heat and Power

Biogas can be used to produce heat and energy. At the farm level biogas can be used as a fuel to generate electricity through an electric generation system such as an engine that operates an electric generator. Often the heat produced in the engine is captured and used to maintain mesophilic or thermophilic temperatures in the anaerobic digestion vessel. Such systems are termed Combined Heat and Power (CHP) systems. In this manner anaerobic digestion is used to form an alternative energy source that can move animal confinement facilities towards energy self sufficiency. CHP systems provide higher profitability than stand-alone power production (Lantz 2004). Additionally, biogas may be upgraded to replace fossil fuels as vehicle fuel or direct injection into natural gas grid systems (Lantz *et al.* 2007).

In 1997, the Kyoto Protocol, ratified by 169 countries, called for a 5% decrease in greenhouse gases and encouraged the adoption of alternative fuels. The European Commission published a white paper (EC 1997) stating that European Countries should produce 12% of their energy from alternative sources by 2010.

In 2004, Pennsylvania enacted an Alternative Energy Portfolio Standard (Pa Act 213, November 2004). The standard calls for 18% of the retail electricity in the state to be provided by alternative energy sources by 31 May 2021. Net metering regulations in Pennsylvania determine how electric utilities compensate customers who generate their own electric with alternative sources. These new policies lessen previous disincentives for small businesses and contain several provisions that favor farm level electric production.

The U.S. Environmental Protection Agency (2006b, 2006c) reports that the potential annual biogas recovery in the U.S. from dairy and swine combined is approximately 96 000 000 000 cu. ft. which could provide an electricity output of 6 332 000 MWh (Table 4.1).

Table 4.1. Potential annual biogas production and energy output from U.S. Swine and Dairy farms. (U.S. Environmental Protection Agency 2006b, 2006c).

	Number of Candidate Farms	Methane Emissions Reduction (1000 Tons)	Carbon Dioxide Emission Reduction Equivalent (1000 Tons)	Methane Production Potential (billion cu.ft./year)	Electric Generation Potential (1000 MWh/year)
Pigs	4281	773	17 779	48	3184
Dairy	2623	573	13 179	48	3148
Total	6904	1346	30 958	96	6332

Environmental Credits from Anaerobic Digestion

Another economic benefit for farms may be the sale of environmental credits or "green" certificates. Environmental credits can be accumulated for measured pollution reduction. The credits can then be purchased by a polluter in lieu of implementation of costly pollution prevention measures. The overall impact on the environment is improved. Market forces affecting credit prices will encourage some polluters to implement new pollution prevention technologies when credit prices are high. A number of credit programs are in place on state, regional, national, and global scales.

At this time sale of carbon credits offer the largest potential environmental profit center for bio-methane producers. The premise of carbon credits is based on the destruction of methane through combustion, either for heat, electricity production, or flaring. Methane that would otherwise enter the atmosphere as an emission is captured from sources such as manure or landfills. Combustion of CH_4 to form CO_2 and water reduces anthropogenic greenhouse gases.

Different Substrates

Another benefit of anaerobic digestion is versatility. Nearly all organic wastes can be anaerobically degraded. Animal manures can be combined with other organic wastes to stabilize the wastes and increase the methane production potential of the manure. Combining materials from different sources is known as co-digestion. Wiese & Haeck (2006) list the following input sources that can be used in manure-based biogas systems:

Farm products: cattle and swine liquid manure, wheat, poultry excrements, dung, straw, rye, barley, oat, maize, rape, sunflowers, peas, beans, lupins, clover, grass, alfalfa, cabbage, potatoes, sugar beets, etc.

Organic waste (primarily from food industries): canteen kitchens, food markets, viniculture, brewery, distillery, gelatin production, jam production, glycerin production, fuller's earth, slaughter houses, etc.

Sewage sludge: from waste treatment plants.

The use of additional biomass in combination with manure must be considered with caution as changes in effluent could lead to environmental concerns. For instance, the use of slaughterhouse waste can lead to manure accumulation of metals, drugs, or other chemicals that had previously accrued in the animals (Salminen & Rintala 2002).

Additional Benefits of Anaerobic Digestion

Biodegradation of organic matter by anaerobic digestion leads to a significant improvement in resource efficiency and environmental impacts compared to those of traditional agriculture practices and current manure handling systems (Lantz *et al.* 2007;

Börjesson & Berglund 2003; Sundberg *et al.* 1997) as well as other benefits. The range of benefits include heat, light, electricity production, transformation of organic wastes into high-quality fertilizer, improvement of hygienic conditions through reduction of pathogens, reduction of work and deforestation from the collection of firewood, and environmental protection of soil, water, air and woody vegetation (Tafdrup 1995; National Academy of Sciences 2001)

Possibilities exist to co-digest different substrates from different geographic areas at single locations; such facilities are known as centralized plants and are common in Europe. Excess manure from farms can be redistributed to others, who can manage the manure treatment. Time and expenses associated with transportation of the manure can be conducted by the management of the centralized plant, affording the farmer more time to attend to farm production. Centralized plants also eliminate the need for the farm to invest in digestion plant capital (Raven & Gregersen 2007). Additionally, centralized plants allow for management of co-digestion where organic wastes from other industries can be mixed with manure at the plant, while creating a renewable energy source and recycling of organic wastes. Centralized plants are typically operated in thermophilic ranges which improve pathogen control and efficiency.

Barriers of Anaerobic Digestion

Widespread adoption of the technology has not occurred because of minimal economic return (Hill *et al.* 1985; Safley and Westerman 1994; Braber 1995). Low economic feasibility can be contributed to many factors. Local circumstances impacting digester economics include: capital costs, construction costs, labor costs, treatment capacity, energy recovery costs, energy prices, fossil fuel prices, energy taxes or tariffs, land price, markets, and availability, quality and costs of digested material (Mata-Alvarez *et al.* 2000).

Traditionally there is little incentive for livestock producers to seek alternative energy sources due to historically affordable fossil fuels and electricity. The competitiveness of biogas with other fuel used for heat or combined heat and power (CHP) is limited (Lantz *et al.* 2007). Many believe that anaerobic digestion will become more affordable as advances in technology, lower capital investment requirements, and rising costs of non-renewable fuels will make biogas systems more economically reasonable (Wiese & Haeck 2006).

Among the farming community, system variation between digesters and limited knowledge among farmers hinders digester adoption. Furthermore, markets for the use of biogas and alternatively produced electricity remain relatively undeveloped. For instance, vehicles that can utilize biogas are more costly than similar vehicles that operate on non-renewable fuels, and competing fuels (such as ethanol) are more readily and economically available (Lantz *et al.* 2007).

Investment for specialized digestion equipment can be costly. Maximum biogas production requires long retention times, and therefore large, expensive digestion vessels, in order to attain complete digestion and maximum biogas output (Fischer 1984).

Digestion vessels can become smaller per volume unit of manure when higher thermophilic temperatures are utilized but increased investment for heating and monitoring equipment is needed for these high temperature systems. At a farm-scale one third of the energy produced by the biogas is needed to provide heat to the digestion vessel (Berglund & Börjesson 2003).

While anaerobic manure treatment offers the benefits of organic matter stabilization and increases the plant availability of the nitrogen fraction of the manure, it does not reduce the nitrogen and phosphorus found in the feedstock (Loria & Sawyer 2005). Farms that produce more manure nutrients than can be utilized by crops grown on the farm will need to export manure from the farm or further treat the manure to reduce nutrient content.

During the anaerobic digestion process nitrous compounds found in substrates such as proteins, amino acids and urea are reduced to ammonia. No further degradation of these compounds occurs under anaerobic conditions (Bernet *et al.* 2000). Ammonia remains in the aqueous solution of the digestate (Bernet *et al.* 2000; Hafner 2006). Aqueous NH_3 can be problematic if it enters waterways, causing excessive oxygen demand in the water and enhancing eutrophication (Bernet *et al.* 2000). Ammonia can also volatilize from the manure into the atmosphere. While ammonia is not a greenhouse gas, it does contribute to particulate pollution, acting as a precursor to fine particulate formation and thereby can lead to negative health impacts. For this reason the United States Environmental Protection Agency regulates ammonia emissions, and is expected to include agriculture soon in its enforcement. Final regulations are not yet available as the impacts of ammonia emissions from agriculture are currently being investigated (U.S. Environmental Protection Agency 2007). The amount of NH_3 that enters the atmosphere from manure is dependent upon two thermodynamic equilibria: ammonia gas/liquid equilibrium and ammonia dissociation equilibrium in the liquid. These characteristics are dependent on pH and temperature, and NH_3 losses increase with increases in either or both of these conditions (Bonmatí & Flotats 2003). In cases where manure is exposed to air, moderate pH and ambient temperatures are usually present, which does not allow complete NH_3 emission from the manure solution.

Biological nitrification-denitrification treatment systems can remove NH_3 from digestate and are widely used in public wastewater treatment systems (Odegaard 1988). Because ammonia is formed from both aerobic and anaerobic manure systems it is difficult to compare possible emissions between digestion systems to that of untreated manure. Nonetheless ammonia emissions may be considered a barrier to digester implementation, the degree of concern will become defined by future developments in science and policy.

When biogas is combusted air quality benefits are realized, however some biogas constituents may increase atmospheric pollution. Of particular concern are sulfur dioxide (SO_2) and nitrogen oxides (NO_X), which are contributors to air pollution and acid rain. Biogas commonly contains the end product of sulfate and other sulfur containing compounds, such as hydrogen sulfide (H_2S), which can be converted to SO_2. Concentrations of biogas H_2S reported in literature are as high as 5.7% (57 000 ppm) (Braun 1982). For swine waste, concentrations as high as 609 ppm have been reported

(Pagilla *et al.* 2000). Increased emissions of NO_X as a result of biogas production are less likely as NOx is a product of aerobic processes. Currently in the United States emissions from farm level biogas combustion remain largely unregulated and emission compliance for such systems is not currently a disincentive to construction.

Current Applications of Anaerobic Digestion

Biogas production is increasing worldwide. European Union countries have signed a directive agreement to decrease landfill deposits of organic matter, including household food waste, by 65% by 2016 (Murto *et al.* 2004). Sweden has a stricter goal, imposing a 25 Euro/Ton tax on landfill organic wastes after 2005 (Murto *et al.* 2004). A common alternative to landfilling in these countries is to transport the organic matter to a large-scale anaerobic digester.

Recently the U.S. Environmental Protection Agency (2006c) has reported the farm demand for anaerobic digestion technologies has increased, with the number of operating digesters more than doubling since 2003. The report reveals the total number of operating farm-scale digesters in the U.S. as 104, with an additional 55 digesters planned.

Raven & Gregersen (2007) summarized several policy changes and market forces that have influenced digester adoption in Denmark. The energy crisis of 1973 spurred interest in anaerobic digestion in Denmark to utilize the technology to process both manure and various organic wastes. Focus on digestion plants in the 1970's was toward farm level construction. However a 1981 survey pointed out that many of the farm-level systems had failed and that gas yields were below predicted values. Developmental focus then shifted towards centralization. In 2002, 20 centralized plants were operating and more than 35 farm-scale digestion plants remained in place. Three factors contributed to digester development in Denmark: (1) in the 1970's the Danish government applied a bottom-up strategy that stimulated interaction and learning between various social groups; (2) a dedicated social network and long-term stimulation enabled continuous development of biogas plants into the late 1990's; (3) Danish government programs have included policies that encourage digester adaptation, taxes on energy, and the preference of Danish farmers to cooperate in small communities where centralized plants can operate. However, no new centralized plants have been constructed in Denmark since 1998. The current setback in digestion development is caused by a shift in energy and environmental policies, liberation of the energy sector, and limited availability of organic waste. Raven & Gergersen concluded that alignment between technical, economic, regulatory and social influences are needed to provide momentum to the anaerobic digestion industry until the technology can survive on its own.

A broad environmental systems analysis of biogas production has been lacking in literature. Recently, Berglund & Börjesson from Lund University, Lund, Sweden have published a series of three papers concerning energy and environmental performances of anaerobic digestion systems that utilize CHP under Swedish conditions. The first paper entitled "Assessment of energy performance in the life-cycle of biogas production" (Berglund & Börjesson 2006) analyzes biogas production potentials in Sweden. The authors considered eight materials for digestion feedstock: cow manure, swine manure, grease separator sludge, ley crops (such as leguminous plants and grasses), municipal organic waste, slaughterhouse waste, tops and leaves of sugar beet, and straw. Both centralized and farm-scale biogas plants were studied. The following observations are reported:

1. Energy balances of anaerobic digestion plants may act as either incentive or barrier depending on many factors. For the energy balance of digestion to be positive the system must use biogas as an end product to create heat or electricity. Energy balances were determined through the input/output ratio of energy calculated as the sum of primary energy input into a biogas system divided by the energy content in the biogas produced. Higher ratios indicate less energy efficient systems and ratios that exceed 100% indicate negative energy.
2. High net energy is not always the main objective as other benefits may be the catalyst for creation of new biogas plants. Some raw materials may not be suitable for digestion if the material contains substrates unsuitable for land application.
3. Operation of the biogas plant is generally the most energy consuming process in the biogas system, corresponding to 50-80% of energy input.
4. Distance is very important in determining economic and energy efficiency of transportation of feedstock to the digestion plant. Longer transportation distances and material with less digestible organic matter decrease energy efficiency. Raw materials can be transported for 200 km (manure) and 700 km (slaughterhouse waste) before energy balances turn negative. Large variations in handling and pre-treatment requirements among feedstocks were noted. Input energy needed to

manage materials that require extensive handling offsets much of the energy produced from biogas where the materials are used.

5. Introducing digestate to agricultural lands leads to energy savings achieved from reduced need of chemical fertilizer production and use.
6. In most cases energy input required to run the system was substantially lower than energy output; typically 20-40%. However, the authors cautioned that it is difficult to reach conclusions on average energy performance since biogas production and system performance are significantly affected by system design and raw materials digested.

The second paper entitled "Environmental systems analysis of biogas systems—Part I: Fuel-cycle emissions" (Börjesson & Berglund 2006) explores the impact of digestion systems on the overall fuel-cycle emissions of carbon dioxide (CO_2), carbon monoxide (CO), nitrogen oxides (NO_X), sulfur dioxide (SO_2), hydrocarbons (HC), methane (CH_4), and particulates. A comparison is drawn between the emissions from the life-cycle of raw materials used for anaerobic digestion and the emission from systems that the anaerobic digestion process replaces. Six feedstocks are considered; ley crops, straw, tops and leaves of sugar beet, liquid swine manure, food industry waste, and municipal organic waste. Both centralized and farm-scale mesophilic digesters are considered. The following observations are made in the paper.

1. Heat demands for farm-scale plants is assumed to be higher than those of large-scale plants because poorer insulation and limited efficiencies of heat exchangers.
2. Energy inputs for biogas plants vary depending on raw material pretreatment requirements. Electricity inputs needed to operate biogas plants contribute to

overall emissions from the plants. These electric inputs, and related emissions, can be offset when electricity is created and used to operate the plant.

3. Ley cropping operations generate the most emissions per unit of energy produced, especially regarding emissions of NO_X, SO_2, HC, and particles. This is largely explained by high diesel consumption in ley cropping operations, corresponding to 13% of energy content of the biogas energy produced, as well as the emissions that stem from the production and application of fertilizers used in ley cropping.
4. One of the largest concerns found in this report involves the uncontrolled losses of methane from biogas plants from system leakages either at the digestion vessel or between the vessel and the biogas utilization point. Limited information exists for levels of actual emissions. Typical losses are reported to range from 5-20% of total biogas produced (Bjurling & Svärd 1998; Sommer *et al.* 2001). Additionally, losses reported at upgrading facilities range from 0.2 to 13% (Persson 2003). Losses of end-use CH_4 typically exceed those from the biogas production chain. Methane losses affect environmental impact of biogas systems in two ways; (1) greenhouse gas emissions increase significantly because CH_4 is a more potent greenhouse gas than CO_2, and (2) all fuel-cycle emissions increase in proportion to CH_4 loss when fuel-cycle emissions are expressed per energy unit of usable biogas. Even small losses of CH_4 are significant. Any CH_4 loss during biogas production corresponded to fuel-cycle increases that almost totally dominated fuel-cycle emissions.
5. Factors that affect fuel-cycle emissions include; (1) biogas yield, (2) energy efficiency in the biogas production chain, (3) transportation distances, (4) method

used for allocation of energy used in the operation of the biogas plant and from spreading the digestates, (5) alternatives to the electricity production, (6) conversion efficiency in the final use of the biogas, and (7) emission data for the end-use technologies and vehicles.

6. Factors that affect the environmental impact of biogas systems include; (1) the raw material digested, (2) the energy efficiency in the biogas production chain, (3) uncontrolled loss of methane, and (4) the status of the end-use technology used. Börjesson & Berglund state that because of these factors it is not possible to specify average fuel-cycle emissions for biogas systems with reasonable reliability. For example, between two biogas systems fuel-cycle emissions may differ by a factor of 3-4 for CO_2, CO, NO_X, HC, and particles, and as high as a factor of 11 for SO_2. Fuel-cycle emissions are normally significantly higher for systems that require extensive handling of feedstock material. The differences are smaller for biogas systems based on agriculture waste products, such as manures and crop residues, as well as municipal organic wastes; where fuel-cycle emissions typical differ by ±10-40% for both large-scale and farm-scale biogas production.

The third paper entitled "Environmental systems analysis of biogas systems—Part II: The environmental impact of replacing various reference systems" (Börjesson & Berglund 2007), explores the overall environmental impact when biogas systems replace various energy production reference systems. The investigation is based on Swedish conditions using a life-cycle perspective, while considering both direct and indirect effects between different biogas and reference systems. The end use technologies considered were large-and farm-scale boilers for heat production, large- and farm-scale gas turbines for CHP, and heavy- and light-duty vehicles. Farm-scale biogas systems are based on liquid swine manure, including land application of manure, and compared to fossil fuel-based reference systems. As in the previous paper six feedstocks are considered (ley crops, straw, tops and leaves of sugar beet, liquid swine manure, food industry waste, and municipal organic waste). Some of the important items found in this report include the following.

1. Fuel-cycle emissions are defined as emissions from production and final use of energy carriers. Indirect environmental effects are defined to be caused by emissions that are not directly related to energy production. Indirect effects are divided into two categories; (1) changed emissions from handling and storage of raw materials and digestate, and (2) changed nutrient leaching due to changed cropping practices.
2. Greenhouse emissions per unit of heat were calculated to be approximately 75-90% when biogas-based heat replaces fossil fuel based heat. Biogas systems emissions contributed 60-75% and 25-40% of the life-cycle emissions for CO_2 and CH_4, respectively. Replacing fossil fuel based heat with biogas-based heat typically increased photochemical ozone creative potential (POCP) by 20-70%. POCP is measure of precursors of tropospheric ozone. Some principle precursors to this ground level ozone formation include NO_X, VOCs (including CH_4), and CO.
3. The consideration of biogas as vehicle fuel yielded varying results depending on which biogas system was used in comparison with fossil fuel systems. For

example when biogas replaced petrol and diesel GWP of emissions was reduced, between 50-80%, except when ley crop-based biogas replaced methanol, in which case GHG emissions increased by 30-50%. GHG emissions are always lower for biogas systems based on biomass in comparison to those based on manure.

4. This third paper deduces that biogas systems offer effective strategies capable of combating several serious environmental problems, including climate change, eutrophication, acidification and air pollution. Indirect benefits such as reduced nitrogen leaching and ammonia emissions may be the most important benefits. The authors conclude that to maximize the potential benefits and minimize potential negative impacts it is crucial that biogas systems be designed and located wisely.

References

ANDARA, A. R. & ESTEBAN, J. M. L. (1999). Kinetic study of the anaerobic digestion of the solid fraction of piggery slurries. *Biomass and Bioenergy* 17, 435-443.

BENDIXEN, H.J. (1994). Safeguards against pathogens in Danish biogas plants. *Water Science and Technology* 30(12), 171-180.

BERGLUND, M. & BÖRJESSON, P. (2003). *Energy systems analysis of biogas systems. Report no. 44*. Lund: Department of Technology and Society, Lund University (in Swedish).

BERGLUND, M. & BÖRJESSON, P. (2006). Assessment of energy performance in the life-cycle of biogas production. *Biomass and Bioenergy* 30, 254-266.

BERNET, N., DELGENES, N., AKUNNA, J. C., DELGENES, J. P. & MOLLETTA, R. (2000). Combined anaerobic-aerobic SBR for the treatment of piggery wastewater. *Water Resources* 34(2), 611-619.

BJURLING, K., SVÄRD, Å. (1998). *Samrötning av organiskt avfall: en studie av svenska biogasanläggningar [Co-digestion of organic waste: a study of Swedish biogas plants]*. Master's Thesis, Lund, Sweden: Department of Water and Environmental Engineering, Lund University (in Swedish).

BLACK, M. I., SCARPINO, P. V., O'DONNELL, C. J., MEYER, K. B., JONES, J. V. & KANESHIRO, E. S. (1982). Survival rates of parasitic eggs in sludge during aerobic and anaerobic digestion. *Applied Environmental Microbiology* 44, 1138-1143.

BLOMQVIST, J. (1993). *Flytgödselns kväveeffekt och kväveulakning i kombination med och utan fånggröda [The effects on nitrogen supply and leakage from liquid manure*

fertilization in combination with and without catch crops]. Borgeby, Sweden: Hushållningsssällskapet i Malmöhus län.
BONMATÍ, A. & FLOTATS, X. (2003). Air stripping of ammonia from pig slurry: characterization and feasibility as a pre- or post-treatment to mesophilic anaerobic digestion. *Waste Management* 23, 261-272.
BOOPATHY, R. (1998). Biological treatment of swine waste using anaerobic baffled reactors. *Bioresource Technology* 64, 1-6.
BÖRJESSON, P. & BERGLUND, M. (2003). *Environmental analysis of biogas systems. Report nr. 45.* Department of Environmental and Energy Systems Studies, Lund University, Lund Sweden, (Report in Swedish with English Summary).
BÖRJESSON, P. & BERGLUND, M. (2006). Environmental systems analysis of biogas systems--Part I: Fuel-cycle emissions. *Biomass and Bioenergy* 30, 496-485.
BÖRJESSON, P. & BERGLUND, M. (2007). Environmental systems analysis of biogas systems--Part II: The environmental impact of replacing various reference systems. *Biomass and Bioenergy* 31(5), 326-344.
BRABER, K. (1995). Anaerobic digestion of municipal solid waste: a modern waste disposal option on the verge of breakthrough. *Biomass Bioenergy* 9, 365-376.
BRAUN, R. (1982). *Biogas mehangärung organischer abfallstoffe*. Springer-Verlan Wien, New York.
CÔTÉ, C., MASSÉ, D. I. & QUESSY. S. (2006). Reduction of indicator and pathogenic microorganisms by psychrophilic anaerobic digestion in swine slurries. *Bioresource Technology* 97, 686-691.
DEBAERE, L. (2000). Anaerobic digestion of solid waste: state-of the art. *Water Science and Technology* 41(3), 282-290.
DUARTE, E. A., MENDES, B. & OLIVEIRA, J. S. (1992). Removal of *Salmonella, Streptococci* and coliforms in pig breeding effluent by anaerobic mesophilic digestion. *Water Science and Technology* 26(9-11), 2169-2172.
EC (THE EUROPEAN COMMISSION) (1997). *White paper – energy for the future: renewable sources of energy, COM/1996/0576 final*.
FISCHER, J. R., IANOTTI, E. L. & PORTER, J. H. (1984). Anaerobic digestion of swine manure at various influent solids concentrations. *Agricultural Wastes* 11, 157-166.
GALLOWAY, J. N. (1998). The global nitrogen cycle: changes and consequences. *Environmental Pollution* 102, 15-24.
GALLOWAY, J. N., DENTENER, F. J., CAPONE, D. G., BOYER, E. W., HOWARTH, R. W., SEITZINGER, S. P., ASNER, G. P., CLEVELAND, C. C., GREEN, P. A., HOLLAND, E. A., EARL, D. M., MICHAELS, A. F., PORTER, J. H., TOWNSEND, A. R. & VOROSMARTY, C. J. (2004). Nitrogen cycles: past, present and future. *Biochemistry* 70, 153-226.
GHOSH, S., CONRAD, J. R. & KLASS, D. L. (1975). Anaerobic acidogenesis of wastewater sludge. J*ournal of Water Pollution Control Federation* 47(1), 30-45.
HAFNER, S. D., BISOGNI, JR, J. J. & JEWELL, W. J. (2006). Measurement of Un-ionized ammonia in complex mixtures. *Environmental Science and Technology* 40, 1597-1602.

HASHIMOTO, A. G. (1983). Conversion of straw-manure mixtures to methane at mesophilic and thermophilic temperatures. *Biotechnology and Bioengineering* 25, 185-200.
HAWKES, F. R. & HAWKES, D. L. (1987). Anaerobic digestion. In: *Bu'lock, J.* (Ed B. Kristiansen), Basic Biotechnology, Academic Press, London, pp. 337-358.
HILL, D. T., HOLMBERG, R. D. & BOLTE, J. P. (1985). Operating and performance characteristics of scraped swine manure as a thermophilic anaerobic digestion substrate. *Agricultural Wastes* 14, 37-49.
IPCC (2001). *Climate change 2001: the scientific basis. Contribution of Working Group I to the Third Assessment Report on the Intergovernmental Panel on Climate Change.* Cambridge University Press, Cambridge, UK.
ISERMANN, K. & ISERMANN, R. (1998). Food production and consumption in Germany: N flows and emissions. *Nutrient Cycling in Agroecosystems* 52, 289-301.
KIM, J. K, OH, B. R., CHUN, Y. N. & KIM, S. W. (2006). Effects of temperature and hydraulic retention time on anaerobic digestion of food waste. *Journal of Bioscience and Bioengineering* 102(4), 328-332.
KIM, S. W., PARK, J. Y., KIM, J. K., CHO, J. H., CHUN, Y. N., LEE, I. H., LEE, J. S., PARK, J. S. & PARK, D. H. (2000). Development of a modified three-stage methane production process using food wastes. *Applied Biochemistry and Biotechnology* 84-86, 731-741.
KUMAR, R., GUPTA, M. K. & KANWAR, S. S. (1999). Fate of bacterial pathogens in cattle dung slurry subjected to anaerobic digestion. *World Journal of Microbiology and Biotechnology* 15, 335-338.
KUN, M. L., BRUNNER, J. B. F. & ATAL, E. E. (1989). Destruction of enteric bacteria and viruses during two-phase digestion. *Journal of the Water Pollution Control Federation* 61, 1421-1429.
LANTZ, M. (2004). *Gårdsbaserad produktion av biogas för kraftvärme—ekonomi och teknik [Farm based biogas production for combined heat and power production-economy and technology].* Master's thesis, Environmental and Energy Systems Studies. Lund, Sweden: Lund University (in Swedish).
LANTZ, M., SVENSSON, M., BJÖRNSSON, L. & BÖRJESSON, P. (2007). *The prospects for an expansion of biogas systems in Sweden – incentives, barriers and potentials*. Energy Policy 35, 1830-1843.
LO, K. V., LIAO, P. H. & GAO, Y. C. (1994). Anaerobic treatment of swine wastewater using hybrid UASB reactors. *Bioresource Technology* 47, 153-157.
LORIA, E. R. & SAWYER, J. E. (2005). Extractable soil phosphorus and inorganic nitrogen following application of raw and anaerobically digested swine manure. *Agronomy Journal* 97, 879-885.
LUND, E. & NIESSEN, B. (1983). The survival of entero-viruses in aerated and unaerated cattle and pig slurry. *Agriculture Wastes* 7, 221-233.
MACKIE, R. I., STROOT, P. G. & VAREL, V. H. (1998). Biochemical identification and biological origin of key odor components in livestock waste. *Journal of Animal Science* 76, 1331-1342.
MARTI, O. G., BOORAM, C. V. & HALE, O. M. (1983). Survival of eggs and larvae of swine nematode parasites in aerobic and anaerobic waste treatment systems. *Journal of Environmental Quality* 9, 401-405.

MASSÉ, D. I. (1995). *Psychrophilic anaerobic digestion of swine manure slurry in intermittently fed sequencing batch reactors.* Thesis, Department of Civil Engineering, University of Ottawa, Ottawa, ON.

MASSÉ, D. I. & DROSTE, R. L. (2000). Comprehensive model of anaerobic digestion of swine manure slurry in a sequencing batch reactor. *Water Resources* 34(12), 3087-3106.

MASSÉ, D. I., MASSE, L. & CROTEAU, F. (2003). The effect of temperature fluctuations on psychrophilic anaerobic batch reactors treating swine manure. *Bioresource Technology* 89, 57-62.

MATA-ALVAREZ, J., MACÉ, S. & LLABRÉS, P. (2000). Anaerobic digestion of organic solid wastes. An overview of research achievements and perspectives. *Bioresource Technology* 74, 3-16.

MØLLER, H. B., SOMMER, S. G. & AHRING, B. K. (2004). Biological degradation and greenhouse gas emissions during pre-storage of liquid animal manure. *Journal of Environmental Quality* 33, 27-36.

MURTO, M., BJÖRNSSON, L. & MATTIASSON, B. (2004). Impact of food industrial waste on anaerobic co-digestion of sewage sludge and pig manure. *Journal of Environmental Management* 70, 101-107.

NATIONAL ACADEMY OF SCIENCES (2001). *Methane generation from human, animal and agricultural wastes.* New York, Hong Kong: Books for Business; 2001.

NIELSEN, L. H., HJORT-GREGERSEN, K., THYGESEN, P. & CHRISTENSEN, J. (2002). *Socio-economic analysis of centralized Biogas Plants: with technical and corporate economic analysis.report nr. 136.* The Danish Institute of Food Economics (in Danish, with extensive English summary).

ODEGAARD, H. (1988). Treatment of anaerobically pretreated effluents. In: *5th International Symposium on Anaerobic Digestion* (Eds E. R. Hall & P. N. Hobson), pp. 225-238. Pergamon Press, Bologna, Italy.

O'NEILL, D. H. & PHILLIPS, V. R. (1992). A review of the control of odour nuisance from livestock buildings: Part 3. Properties of the odorous substances which have been identified in livestock wastes or in the air around them. *Journal of Agricultural Engineering Research* 53, 23-50.

PA ACT 213 (2004). *Alternative Energy Portfolio Standards Act [PA Senate Bill 1030; Session of 2004]*. Pennsylvania Act No. 213, November 30, 2004.

PAGILLA, K. R., KIM, H. & CHEUNBARN, T. (2000). Aerobic thermophilic and anaerobic mesophilic treatment of swine waste. *Water Resource* 34(10), 2747-2753.

PA SCC (Pennsylvania State Conservation Commission) (2007). *Proposed Rulemaking; State Conservation Commission [25 PA. CODE CH. 83], Facility Odor Management [37 Pa.B. 4780].* The Pennsylvania Bulletin, September 1, 2007.

PERSSON, M. (2003). *Utvärdering av uppgraderingstekniker för biogas [Evaluation of biogas upgrading technologies]*. Rapport SGC 142, Malmö, Sweden: Swedish Gas Centre (in Swedish).

PETERSEN, S. (1999). Nitrous oxide emissions from manure and inorganic fertilsers applied to spring barley. *Journal of Environmental Quality* 28, 744-751.

RAVEN, R. P. J. M. & GREGERSEN, K. H. (2007). Biogas plants in Denmark: success and setbacks. *Renewable and Sustainable Energy Reviews* 11, 116-132.

ROOS, K. F., MARTIN, JR, J.H. & MOSER, M. A., 2004. *AgSTAR Handbook; A manual for developing biogas systems at commercial farms in the United States; Second Edition.* U.S. Environmental Protection Agency. EPA-430-B-97-015.

SAFLEY, JR, L. M. & WESTERMAN, P. W. (1994). Low-temperature digestion of dairy and swine manure. *Bioresource Technology* 47, 165-171.

SALMINEN, E. & RINTALA, J. (2002). Anaerobic digestion of organic solid poultry slaughterhouse waste – a review. *Biosource Technology* 83, 13-26.

SHIH, J. C. H. (1987). Ecological benefits of anaerobic digestion. *Poultry Science* 66, 946-950.

SOMMER, S. G., MØLLER, H. B. & PETERSEN, S. O. (2001). *Reduktion af drivhusgasemission fra gylle og organisk affald ved biogasbehandling [The reduction of greenhouse gases from manure and organic waste using digestion and biogas production]. DJF-rapport nr 31*, Tjele, Denmark: Husdyrbrug, Danish Institute of Agricultural Sciences (in Danish).

SOMMER, S., PETERSEN, S. & SÖRGARD, H. (2000). Greenhouse gas emission from stored livestock slurry. *Journal of Environmental Quality* 28, 1610-1618.

SUNDBERG, M., JOHANSSON, W. & HJORSTBERG, H. (1997). *Biogas in future agriculture and sustainable societies: Effects on soil, environment and economy. Report nr. 12 (Recycle and Waste).* Swedish Institute of Agricultural Engineering (in Swedish).

SVENSSON, M., CHRISTENSSON, K. & BJÖRNSSON, L. (2005). Biogas production from crop residues on a farm-scale level: is it economically feasible under conditions in Sweden? *Bioprocess and Biosystems Engineering* 28, 139-148.

TAFDRUP S. (1995). Viable energy production and waste recycling from anaerobic digestion of manure and other biomass materials. *Biomass and Bioenergy* 9(1-5):303-314.

U.S. ENVIRONMENTAL PROTECTION AGENCY (2006a). *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990 – 2005*. April 15, 2007. U.S. Environmental Protection Agency, Washington, DC.

U.S. ENVIRONMENTAL PROTECTION AGENCY (2006b). *Market Opportunities for Biogas Recovery Systems: A Guide to Identifying Candidates for On-Farm and Centralized Systems.* EPA-430-8-06-004, June 15, 2006. U.S. Environmental Protection Agency, Washington DC.

U.S. ENVIRONMENTAL PROTECTION AGENCY (2006c). AgSTAR Digesters Continue Accelerating in the U.S. Livestock Market. *AgSTAR Digest, Winter 2006*.

U.S. ENVIRONMENTAL PROTECTION AGENCY (2007). *Environmental Protection Agency, 40 CFR Part 51 [EPA-HQ-OAR-2003-0062; FRL-], RIN 2060-AK74. Clean Air Fine Particle Implementation Rule, March 29, 2007.* U.S. Environmental Protection Agency, Washington, DC.

VAN LIER, J. B., TILCHE, E., AHRING, B. K., MACARIE, H., MOLETTA, R. & DOHANYOS, M. (2001). New perspectives in anaerobic digestion. *Water Science and Technology* 43(1), 1-18.

WHALEN, S.C. & DEBERARDINIS, J.T. (2007). Nitrogen mass balance in fields irrigated with liquid swine waste. *Nutrient Cyclin in Agroecosystems* 78, 37-50.

WIESE, J. & HAECK, M. (2006). Instrumentation, control and automation for full-scale manure-based biogas systems. *Water Science and Technology* 54(9), 1-8.

Chapter 5. Overview of Swine Lying and Dunging Behavior

Pigs in swine housing commonly excrete and rest in different areas within the floor space of their living space. Many factors appear to contribute to this behavior including pig comfort, crowding, temperature, ventilation factors, floor type, pen shape, pig body weight and lighting.

Behavioral Aspects

By nature pigs are clean animals preferring to urinate (Whatson 1978; Baxter 1984; Stolba & Wood-Gush 1989, Huynh *et al.* 2005) and dung (Stolba & Wood-Gush 1989) away from resting areas. It has been suggested that pigs like to dung in open areas (Fritschen 1975) and in areas of the pen that are not busy with pig activity (Baxter 1982), which is a reason that pigs dung away from food (Aarnink *et al.* 1997). The posture of a pig during excretion has been labeled as 'unstable' and increases the vulnerability of the pig to attack causing pigs to seek an isolated place for dunging (Randall *et al.* 1983).

During a study of both rearing and fattening pigs in pens with combined solid and slatted flooring pigs were found to urinate an average of 7.1 times per day (s.d 0.9) and defecate 6.1 times per day (s.d. 1.6) (Aarnink *et al.* 1996).

Temperature Affects on Behavior

To adapt to ambient temperature fluctuations pigs will change behavior, which is known as social thermoregulation (Boon 1981; Sællvik & Walberg 1984). Temperature influences the location and posture of lying pigs (Close 1981; Hacker *et al.* 1994;

Blackshaw & Blackshaw 1994; Beattie *et al.* 1996; Aarnink *et al.* 2001, 2006; Peishi & Toshio 2001). Temperature has a larger impact on lying and excreting behavior than relative humidity (Huynh *et al.* 2005). Because pigs are unable to sweat they are considered more sensitive to hot than cold conditions (Ingram 1965; Andersen *et al.* 1998). Heavier pigs generally have thicker fat layers and will lose less heat per unit area of body surface than smaller pigs (Brown-Brandl *et al.* 2004). Tolerance of heat and pig body weight are inversely related (Botermans & Andersson 1995; Hillmann *et al.* 2004). Exposure to both extreme heat and cold can cause stress in pigs as indicated by increases in cortisol concentrations (Bate & Hacker 1985; Becker *et al.* 1997).

Wallowing, defined as rolling or rubbing in feces and/or urine, is a thermoregulatory behavior pigs use to cool (Huynh *et al.* 2005). Pigs in confinement finishing pens will lie in dung areas to enhance heat loss when temperatures rise above 21 °C (Hillmann *et al.* 2004). Shi *et al.* (2006) modified pen floor temperatures, finding that greater than 85% of pigs rested in a lying area when temperature was less than 26 °C, 10-20% lie in the area when the temperature was approximately 30 °C, and hardly any pigs were found in the lying area when floor temperatures were above 33 °C.

Huddling is a behavior pigs use to stay warm (Riskowski *et al.* 1990; Geers *et al.* 1986 1987). Huddling is widely considered a behavior reaction of mammals (Blumberg *et al.* 1992; Jones *et al.* 1999). Huddling is defined as pigs lying in a pile or with over 50% of their lying side in contact with another pig, huddling will increase when temperatures decrease (Hillmann *et al.* 2004; Huynh *et al.* 2005). For pigs, huddling likely reflects discomfort due to exposure to cold air (Boon 1981; Riskowski *et al.* 1990). Pigs less than 50 kg will lie on top of one another (Boon 1981). For medium and heavy pigs huddling can result in a state of discomfort and lead to increases in standing because pigs are uncomfortable (Riskowski *et al.* 1990; Hillman *et al.* 2004). Smaller pigs are less affected by this.

Lying without body contact is one way for pigs to increase body heat loss (Baxter 1984). Maximizing distance from other pigs allows for more radiant heat loss, while minimizing the radiant heat gain from other pigs (Huynh *et al.* 2005). As temperatures increase in a confinement situation pigs increase lying without body contact, while decreasing sternal lying postures (lying on belly) and increasing lateral postures (lying on side) (Olsen *et al.* 2001; Riskowski et. al 1990; Geers *et al.* 1986; Huynh *et al.* 2005). This increases heat loss due to greater body surface contact with flooring (Mount 1979; Close 1981; Huynh *et al.* 2005). With each 1 ºC temperature increase, the number of pigs lying laterally increases by 1.8%, while physical contact between pigs decreases 3.7% (Huynh *et al.* 2005). Pig skin temperature increases 0.25 ºC for every 1 ºC increase in ambient temperature (Huynh *et al.* 2004). Baxter (1984) found that increasing ventilation rates and air velocity, increased body heat loss, leading to an increase in body contact of recumbent pigs.

Relaxation is important for health and growth of pigs. In thermonuetral conditions of 16-18 ºC (defined by Petherick 1983) fattening pigs spend 88% (Huynh *et al.* 2005) to 90% (Ekkel *et al.* 2003) of the 24 hour day lying, with lateral lying position accounting for

60% of observed lying posture at these times (Ekkel *et al.* 2003). Others have reported lying time as 78% (Taylor *et al.* 2006) and 80% (Haugse *et al.* 1965). Pigs demonstrate circadian patterns and are almost totally inactive for long periods at night (Aarnink *et al.* 1996). Pigs consume greater floor space when they are lying verses standing; therefore, more floor space is needed at night (McGlone & Newby 1994). Pigs are more likely to eat during daylight hours. As pigs grow they spend less time eating (Hyun *et al.* 1997).

Temperature also influences dunging behavior. The number of excretions decreases with increases in temperature, possibly due to increased respiration rate used to aid thermoregulation during higher temperatures (Huynh *et al.* 2005; Aarnink *et al.* 2006). Huynh *et al.* (2004) reported that urination frequency decreases with rising temperatures due to water loss through increased respiration rate. However, Aarnink *et al.* (2006) report that no change in urination frequency was found with increasing temperatures.

Space Needs and Pen Shape

The space available for movement is shared among pigs in grouped pens. The number of animals in a given pen as well as the manner in which pigs utilize the pen determine the amount of free space in the pen. If most animals are resting in a preferred lying area, active animals have use of the remainder of the pen. This shared space leads to much larger free space than if animals are housed individually (Baxter 1992). Space needs per pig decrease slightly as group size increases. For this reason, it has been reported that commercial operations may decrease the space allotment per pig as group size increases and still allow sufficient lying area and free space (McGlone & Newby 1994; Wolter *et* *al.* 2000). However, a recent factorial study failed to detect a performance difference response to crowding in large and small pens (Street & Gonyou 2005).

Pigs are reported to space themselves near pen perimeters (Grandin 1980) as are cattle (Stricklin *et al.* 1979). Corners can act as a hiding area, which decreases the aggression in newly formed groups (McGlone & Curtis 1985). Weigand *et al.* (1994) conducted a study of groups of 15 pigs in pens of various sizes and shapes found that pigs defecated in corners, except for circular pens. In this study, 32 and 27% of the pen floor area was used as a dunging area for small and large pens, respectively. Pen size had no impact on inter-animal space in this study. Animal perception of pen space depends on the amount of pen wall available to the animals. The authors report three ways to improve space quality; (1) increasing the perimeter of pens by maximizing the ratio of pen perimeter to pen area, which decreases both competition for lying space and aggression, thus promoting positive social contact and well-being, (2) increase maximum distance that two conflicting animals can separate within a pen, and (3) provide corners where semi-isolation allows for avoidance of aggression, formation of small social groups, and distinct dunging patterns. The shape of a pen affects all of these qualities.

Floor space is important for economic and welfare reasons. As crowding increases, individual pig production decreases (Gonyou & Stricklin 1998). Crowding reduces feed intake and average daily gain (Gehlbach *et al.* 1966) and can lead to aggressive behavior (Randolph, *et al.* 1981; Kornegay 1986). Elevation of plasma glucocorticoid levels, an indication of stress, can be found in swine when space allowance is decreased (Meunier-

Salaun *et al.* 1987). Pigs with less space spend more time standing and making vocal protests (Weigand *et al.* 1994).

Greater pig production will occur as space per animal increases up to a critical value, at which time a plateau in production will occur (Robbins 1986). Investment costs to construct housing with more space per animal can lead to an inverse relationship between economics and welfare. A stocking point beyond which productivity diminishes suggests that physiological, health and welfare issues may exist. Welfare standards provide pig space allocations. Based on space per animal, standards for finishing pigs are 0.74 m^2 per pig in the US (NPB 2003), 1.00 m^2 per pig in the EU (European Community 2001), and 0.7 m^2 per pig in Canada (Canada Plan Service 1986). This area accounts for both static and free space within the pen. McGlone & Newby (1994) report that the minimum static space needed for resting grow-finish pigs is 0.54 m^2. The authors further report that if this space is not provided beyond this amount performance will suffer. Components of quantity (amount of space provided) and quality (features of space which facilitate or restrict animal usefulness) should be considered during pen design (Weigand *et al.* 1994). Space quality can be manipulated by changing the perimeter to area ratio.

In a study of allometric relationships within piggeries Gonyou *et al.* (2006) report three common means to express space allowance for pigs. The most common expression is space per animal (e.g. m^2/pig), while another is weight density (e.g. kg pig/m^2). The third is an allometric approach that converts body weight (BW) into a two-dimensional concept to calculating floor space allowances (A), yielding the expression A = *k* *

$BW^{0.667}$, where *k* represents the space allowance coefficient (proposed by Petherick 1983; Baxter 1984; supported by Hurnik & Lewis 1991; applied by Edwards *et al.* 1988; Gonyou & Stricklin 1998). Petherick (1983) found that this approach applies, and allows comparison, over a wide range of weights as long as the allometric relations of shape and density remain constant, and although the shape of pigs may vary somewhat with different sizes and differ with genotypes (McGlone *et al.* 2004) it is likely to have a much smaller effect on area measures than does BW. Gonyou *et al.* (2006) found a minimum *k* value of 0.034 for fully slatted floor housing, reporting that space allowanced below this level resulted in a linear depression of growth and feed intake. Edwards *et al.* (1988) report a *k* value of 0.030 for grow-finish hogs on fully slatted floors. Adopted recommendations for the space allowance coefficient, *k*, include 0.034 at 68 kg in the US (NPB 2003), 0.028 for grower-finisher pigs in the EU (European Community 2001), and 0.035 for pigs on fully slatted floors in Canada (AAFC 1993).

Turner & Edwards (2004) report interesting findings concerning the behavior of swine in large verses small pens. Stocking density seems to be more important than pen size. Pigs in larger groups do not exhibit increased aggression when compared to pigs in smaller pens with similar stocking density. Aggression toward novel individuals is diminished in larger groups, which may be explained by more effective avoidance (pigs can get further away from aggressors). Pigs in groups of all sizes exhibit fighting behavior when first introduced to a pen. However, pigs in large groups do not fight more than those in small groups, indicating that pigs may fight after mixing until a ceiling, or fatigue, or injury is reached, or that they become more selective over which group members they chose to fight.

Flooring Considerations

The type of flooring found within pig housing may influence lying and dunging preferences. Solid floors have been found to be warmer than slatted floors by 3-5 °C (Randall *et al.* 1983) and 3.6 °C (Huynh *et al.* 2005). In swine houses where pens contain both solid and slatted floors the solid portion of the floor has been found to be warmer, and in general pigs prefer to dung on the cooler slatted floors (Steiger *et al.* 1979; Randall *et al.* 1983). As pigs grow in these pens, free space decreases, which causes more dunging on the solid floor and more lying on slats (Hacker *et al.* 1994; Aarnink *et al.* 1996; Huynh *et al.* 2005; Aarnink *et al.* 2006). Similarly in summer, when warmer ambient temperatures cause pigs in these pens to utilize more floor space when lying with minimum body contact and lateral posture, dunging on solid floors increases and lying in slatted areas increases, as compared to winter (Aarnink *et al.* 1996). Aarnink *et al.* (1996) found no difference in excretory and lying behavior of fattening pigs in a comparison of pens with 25 and 50% slatted floor areas.

For pigs, little is known about inflection temperature (IT), the exact temperature above which pigs alter lying and excretion behavior. Aarnink *et al.* (2006) considered IT for fattening pigs in pens with combined solid and slatted flooring reporting that IT decreases with increased body weight (BW) as larger pigs can become heat-stressed at lower temperatures, supporting previous reports from Nienaber *et al.* (1999). Rises in IT also increase lateral lying, while sternal lying and body contact while lying decrease. Excretion on solid flooring remained constant as temperatures were raised until IT was reached

Lighting

A preference of pigs to dung in well-illuminated areas has been reported (Randall *et al.* 1983). Taylor *et al.* (2006) found a predilection of pigs to rest in dimmer areas and dung in brighter illuminance, noting that the experiment did not determine whether this dunging pattern could emerge due to pig preference to defecate away from resting and sleeping areas (Olsen *et al.* 2001), or due to the preference of pigs to defecate in bright area and rest away from the dunging area.

References

AAFC (1993). *Recommended Code of Practice for the Care and Handling of Farm Animals: Pigs.* Agriculture and Agri-Food Canada 1898/E, Ottawa, Canada, page 55.

AARNINK, A. J. A., SCHRAMA, M. J. W., HEETKAMP, M. J. W., STEFANOWSKA, J. & HUYNH, T. T. T. (2006). Temperature and body weight affect fouling of pig pens. *Journal of Animal Science* 84, 2224-2231.

AARNINK, A. J. A., SCHRAMA, M. J. W., VERHEIJEN, R. J. E. & STEFANOWSKA, J. (2001). *Pen fouling in pig houses affected by temperature. InL R.R., S., R., B., R.W., B. (Eds.), Livestock Environment VI.* The Society for Engineering in Agricultural, Food, and Biological Systems, Gault House Hotel Louisville, Kentucky, pp. 180-186.

AARNINK, A. J. A., SWIERSTRA, D., VAN DEN BERG, A. J. & SPELLMAN, L. (1997). Effect of type of slatted floor on ammonia emission rates from fattening piggeries. *Journal of Agricultural Engineering Research* 66, 93-102.

AARNINK, A. J. A., VAN DER BERG, A. J., KEEN, A., HOCKSMA,P. & VERSTEGEN, M. W. A. (1996). Effect of slatted floor area on ammonia emission and on the excretoryand lying behaviour of growing pigs. *Journal of Agricultural Engineering Research* 64, 299-310.

ANDERSEN, I. L., BOE, K. E. & HOVE, K. (1998). Cold housing of pregnant sows. In: *Proceedings of the International Conference on Agricultural Engineering, Oslo*, pp. 1-8.

BATE, L. A. & HACKER, R. R. (1985). Effect of cannulation and environmental temperature on the concentration of serum cortisol in pregnant sows. *Canadian Journal of Animal Science* 65, 399-404.
BAXTER, M. R. (1982). Environmental determination of excretory and lying areas in domestic pigs. *Applied Animal Ethology* 9, 195.
BAXTER, M. R. (1992). The space requirements of housed livestock. In: *Farm Animals and the Environment* (Eds C. Phillips & D. Piggins), C.A.B. International, London, pp. 67-79.
BAXTER, S. (1984). *Intensive Pig Production: Environmental Management and Design.* Granada Publishing Ltd., London, UK.
BEATTIE, V. E., WALKER, N. & SNEDDON, I. A. (1996). An investigation of the effect of environmental enrichment and space allowance on the behaviour and production of growing pigs. *Applied Animal Behaviour Science* 48, 151-158.
BECKER, B. A., KLIR, J. J., MATTERI, R. L., SPIERS, D. E., ELLERSIEK,M. & MISFELDT, M. L. (1997). Endocrine and thermoregulatory response to acute thermal exposures in 6-month-old pigs reared in different neonatal environments. *Journal of Thermal Biology* 22(2), 87-93.
BLACKSHAW, J. K. & BLACKSHAW, A. W. (1994). Shade-seeking and lying behaviour in pigs of mixed sex and age, with access to outside pens. *Applied Animal Behaviour Science* 39, 249-257.
BLUMBERG, M. S., EFIMOVA, I. V. & ALBERTS, J. R. (1992). Ultrasonic vocalizations by rat pups: the primary importance of ambient temperatures and the thermal significance of contact comfort. *Developmental Physiobiology* 25(4), 229-250.
BOON, C. R. (1981). The effect of departures from lower critical temperatures on the group postural behavior of pigs. *Animal Production* 33, 71-79.
BOTERMANS, J. & ANDERSSON, M. (1995). Growing-finishing pigs in an uninsulated house. *Swedish Journal of Agricultural Research* 25, 83-92.
BROWN-BRANDL, T. M., NIENABER, J. A., XIN, H. & GATES, R. S. (2004). A literature review of swine heat production. *Transactions of the ASAE* 47, 259-270.
CANADA PLAN SERVICE (1986). *Plan M-3000: Swine Housing and Equipment*.
CLOSE, W. H. (1981). The climatic requirements of the pig. In: *Environmental Aspects of Housing for Animal Production* (Ed J. A. Clark), Pages 149-167, Butterworths, London, UK.
EDWARDS, S. A., ARMSBY, A. W. & SPECHTER, H. H. (1988). Effects of floor area allowance on performance of growing pigs kept on fully slatted floors. *Animal Production* 46, 453-459.
EKKEL, E. D., SPOOLDER, H. A. M., HULSEGGE, I. & HOPSTER, H. (2003). Lying characteristics as determinants for space requirements in pigs. *Applied Animal Behaviour Science* 80, 19-30.
EUROPEAN COMMUNITY (2001). European Council Directive 2001/88/EC of 23 October 2001 amending Directive 91/630/EEC laying down minimum standards for protection of pigs. *Official Journal of the European Communities* L316 44, 1-4.
FRITSCHEN, R. D. (1975). *Toilet Training of Pigs On Partly Slatted Floors*. University of Nebraska, Lincoln, NebGuide G 74-40.

GEERS, R., GOEDSEELS, V., PARDUYNS, G. & VERCRUYSSE, G. (1986). The group postural behavior of growing pigs in relation to air velocity, air and floor temperature. *Applied Animal Behaviour Science* 16, 353-362.
GEERS, R., VAN DER HEL, W., VERHAGEN, J., VERSTEGEN, M., GOEDSELS, V., BRANDSMA, H., HENSKEN, A., SCHÖLLER, J. & BERCKMANS, D. (1987). Surface temperatures of growing pigs in relation to the duration of acclimation to air temperature or draught. *Journal of Thermal Biology* 12(4), 249-255.
GELBACH, G. D., BECKER, D. E., COX, J. L., HARMON, B. G. & JENSEN, A. H. (1966). Effects of floor space allowance and number per group on performance of growing-finishing swine. *Journal of Animal Science* 25, 386-391.
GONYOU, H. W., BRUMM, M. C., BUSH, E., DEEN, J., EDWARDS, S. A., FANGMAN, T., MCGLONE, J. J., MEUNIER-SALAUN, M., MORRISON, R. B., SPOOLDER, H., SUNDBERG, P. L. & JOHNSON, A. K. (2006). Application of broken-line analysis to assess floor space requirements of nursery and grower-finisher pigs expressed on an allometric basis. *Journal of Animal Science* 84, 229-235.
GONYOU, H. W. & STRICKLIN, W. R. (1998). Effects of floor area allowance and group size on the productivity of growing/finishing pigs. *Journal of Animal Science* 76, 1326-1330.
GRANDIN, T. (1980). Livestock behavior as related to handling facilities design. *International Journal for the Study of Animal Problems* 1, 33-52.
HACKER, R. R., OGILVIE, J. R., MORRISON, W. D. & KAINS, F. (1994). Factors affecting excretory behavior of pigs. *Journal of Animal Science* 72, 1455-1460.
HAUGSE, C. N., DINUSSON, W. E., ERICKSON, D. O., JOHNSON, J. N. & BUCHANAN, M. L. (1965). A day in the life of a pig. *Feedstuffs* 37, pp. 18-23.
HILLMANN, E., MAYER, C., SCHÖN, P-C., PUPPE, B. & SCHRADER, L. (2004). Vocalisation of domestic pigs (*Sus scrofa domestica*) as an indicator for their adaptation towards ambient temperatures. *Applied Animal Behaviour Science* 89, 195-206.
HURNIK, J. F. & LEWIS, N. J. (1991). Use of body surface area to set minimum space allowance and virginiamycin supplementation on feedlot performance of swine. *Canadian Journal of Animal Science* 71, 577-580.
HUYNH, T. T. T., AARNINK, A. J. A., VERSTEGEN, M. W. A., GERRITS, W. J. J. & HEETKAMP, M. J. H. (2004). Pigs' physiological responses at different relative humidities and increasing temperatures. In: *2004 ASAE/CSAE Annual International Meeting, Fairmont Chateau Laurier, The Westin, Government Centre – Ottawa, Ontario, Canada*. p. 16, paper number 044033.
HUYNH, T. T. T., AARNINK, A. J. A., GERRITS, W. J. J., HEETKAMP, M. J. H., CANH, T. T., SPOOLDER, H. A. M., VERSTEGEN, M. W. A. & KEMP, B. (2005). Thermal behavior adaptation of growing pigs as affected by temperature and humidity. *Applied Animal Behaviour Science* 91, 1-16.
HYUN, Y., ELLIS, M., MCKEITH, F. K. & WILSON, E. R. (1997). Feed intake pattern of group-housed growing-finishing pigs monitored using a computerized feed intake recording system. *Journal of Animal Science* 75; 1443-1451.
INGRAM, D. L. (1965). Evaporative cooling in the pig. *Nature* 207; 415-416.

JONES, J. B., WEBSTER, A. J. F. & MATHES, C. M. (1999). Trade-off between ammonia exposure and thermal comfort in pigs and the influence of social contact. *Animal Science* 68, 387-398.
KORNEGAY, E. T. (1986). Stocking density for growing-finishing swine. *Proceedings of the Annual Meeting of the American Association of Swine Practitioners, 16-18 March 1986, Minneapolis, MN.* American Association of Swine Practitioners, Des Moines, IA.
MCGLONE, J. J. & CURTIS, S. E. (1985). Behavior and performance of weanling pigs in pens equipped with hide areas. *Journal of Animal Science* 60, 20-24.
MCGLONE, J. J. & NEWBY, B. E. (1994). Space requirements for finishing pigs in confinement: behavior and performance while group size and space vary. *Applied Animal Behaviour Science* 39, 331-338.
MCGLONE, J. J., VINES, B., RUDINE, A. C. & DUBOIS, P. (2004). The physical size of gestating sows. *Journal of Animal Science* 82, 2421-2427.
MEUNIER-SALAUN, M. C., VANTRIMPONTE, M. N., RAAB, A. & DANTZER, R. (1987). Effects of floor area restriction upon performance, behavior and physiology of growing-finishing pigs. *Journal of Animal Science* 64, 1371-1377.
MOUNT, L. E. (1979). *Adaptation to thermal environment: Man and his productive animals*. Edward Arnold Limited, Thomson Litho Ltd., East Kilbride, UK.
NIENABER, J. A., HAHN, G. L. & EIGENBERG, R. A. (1999). Quantifying livestock responses for heat stress management: A review. *International Journal of Biometeorology* 42, 183-188.
NPB (2003). *Swine Care Handbook*. National Pork Board, Des Moines, IA, page 34.
OLSEN, A. W., DYBKJÆR, L. & SIMONSEN, H. B. (2001). Behaviour of growing pigs kept in pens with outdoor runs II. Temperature regulatory behaviour, comfort behaviour and dunging preferences. *Livestock Production Science* 69, 265-278.
PEISHI, Y. & TOSHIO, I. (2001). Variation of resting position of miniature pigs by environmental temperature and food intake. *Journal of Animal Science* 72, J62-J68.
PETHERICK, J. C. (1983). A biological basis for the design of space in livestock housing. In: *Farm Animal Housing and Welfare* (Eds S. H. Baxter, M. R. Baxter & J. A. S. C. MacCormack), pp 103-120, Martinus Nijoff Publisher, Boston, MA.
RANDALL, J. M., ARMSBY, A. W. & SHARP, J. R. (1983). Cooling gradients across pens in a finishing piggery. *Journal of Agricultural Engineering Research* 28, 247-257.
RANDOLPH, J. H., CROMWELL, G. L., STAHLY, T. S. & KRATZER, D. D. (1981). Effects of group size and space allowance on performance and behavior of swine. *Journal of Animal Science* 53, 922-927.
RISKOWSKI, G. L., BUNDY, D. S. & MATTHEWS, J. A. (1990). Huddling behaviour and hematology of weanling pigs as affected by air velocity and temperature. *Transactions of the ASAE* 33(5), 1677-1685.
ROBBINS, K. R. (1986). A method, SAS program, and example for fitting the broken-line to growth data. *University of Tennessee Agricultural Experiment Station Research Report* 86-09, 8 pp.
SÆLLVIK, K. & WALBERG, K. (1984). The effects of air velocity and temperarure on the behaviour and growth of pigs. *Journal of Agricultural Engineering Research* 30, 305-312.

SHI, Z., LI, B., ZHANG, X., WANG, C., ZHOU, D. & ZHANG, G. (2006). Using floor cooling as an approach to improve the thermal environment in the sleeping area in an open pig house. *Biosystems Engineering* 93(3), 359-364.

STEIGER, A., TSCHANZ, B., JAKOB, P. & SCHOLL, E. (1979). Verhaltungsuntersuchunger bei Mastschweinen auf verchiedenen Bodenbelagen und bei verchiedenen Besatzdichte [Behavior of fattening pigs on different floor types and at different densities]. *Scweizer-Archiv für Tierheikunde und Tierzucht* 121, 109-126.

STOLBA, A. & WOOD-GUSH, D. G. M. (1989). The behavior of pigs in a seminatural environment. *Animal Production* 48, 419-425.

STREET, B. R. & GONYOU, H. W. (2005). The effects of housing grow-finish pigs at two different group sizes and space allocations. *Journal of Animal Science* 83(Suppl. 2):38. (Abstr.)

STRICKLIN, W. R., GRAVES, H. B. & WILSON, L. L. (1979). Some theoretical and observed relationships of fixed and portable spacing behavior in animals. *Applied Animal Ethology* 5, 201-214.

TAYLOR, N., PRESCOTT, N., PERRY, G., POTTER, M., LE SUEUR, C. & WATHES, C. (2006). Preference of growing pigs for illuminance. *Applied Animal Behaviour Science* 96, 19-31.

TURNER, S. P. & EDWARDS, S. A. (2004). Housing immature domestic pigs in large social groups: implications for social organization in a hierarchical society. *Applied Animal Behaviour Science* 87, 239-253.

WHATSON, T. S. (1978). The development of dunging preference in piglets. *Applied Animal Ethology* 4(3), 293.

WEIGAND, R. M., GONYOU, H. W. & CURTIS, S. E. (1994). Pen shape and size: effects on pig behavior and performance. *Applied Animal Behaviour Science* 39, 49-61.

WOLTER, B. F., ELLIS, M., CURTIS, S. E., PARR, E. N. & WEBEL, D. M. (2000). Group size and floor-space allowance can affect weanling-pig performance. *Journal of Animal Science* 78, 2026-2067.

Chapter 6. Airborne Pollutants in Swine Housing Systems

The cleanliness of swine houses can impact performance. Lee *et al.* (2005) compared swine performance indicators between pigs living in 'clean' verses 'dirty' environments. Prior to stocking the clean facility received washing and disinfection of the living area and, once stocked, was cleaned daily with a hose, had effluent flushed daily from manure channels, and was fogged twice daily with a virucidal agent. The dirty area received no cleaning and effluent was continuously cycled through the manure channels in an effort to maintain increased noxious gas levels. Ammonia concentration in the air of the dirty environment was higher than that of the clean environment (13.0 vs. 6.0 ppm, $P < 0.001$). Pigs in the dirty environment ate less and grew more slowly than that of pigs in the clean environment. Animals in the dirty environment exhibited increases in cortisol and β-endorphin concentrations and a decrease in insulin like growth factor I (IGF-I). Others report 10-25% decreases in growth rate of pigs in commercial verses highly sanitized facilities (Coffey & Cromwell 1994; Ekkel *et al.* 1995).

Storage of liquid manure in under-floor pits prior to land application is common practice for intensive swine production units. A disadvantage is that the manure is stored inside, under the swine housing area, where air is subject to contamination by gases and odor released from manure decomposition. Manure pits must be well managed to decrease the threat to indoor air quality and the environment. Some of the gases of concern are ammonia (NH_3), hydrogen sulfide (H_2S), carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4). Ammonia and hydrogen sulfide are known to couple with dust in etiology of disease for both humans and pigs (Zhang *et al.* 1994). Carbon dioxide, hydrogen sulfide, and ammonia have been shown to have negative physiological effects on livestock, such as respiratory stress (Anderson *et al.* 1987). Methane seems to have no negative health impact unless the concentration is so high that it displaces oxygen but it can be explosive and is a potent green house gas (Muehling 1969; Lioa 1996). Other gases such as amides and sulfur compounds are found in confinement atmospheres (Miner & Hazen 1969; Kreis 1978, Hammond & Smith 1981), and most of these compounds can contribute to odor but have no known physiological effects (Anderson *et al.* 1987). Gas concentrations vary along the length of a building (Robertson & Galbraith 1971) as well as within the height the building (Skarp 1975).

Ventilation of swine barns is important to remove pollutants, provide fresh air and regulate temperature (Wathes *et al.* 1983). The susceptibility of the pig to respiratory disease as a result of prolonged exposure to irritating airborne pollutants has long been suspected, as noted in 1962 by Switzer (as cited by Muehling 1969). Others have suggested a need for identification of contaminants and data collection on aerial pollutant concentrations (Day *et al.* 1965; Roller 1965). Dust and gases identified from confinement buildings were soon identified as irritants and organic agents were identified as factors that could etiologically influence respiratory disease (Kovács *et al.* 1967; Jericho 1968). Inhalation of irritants can increase mucus secretion, adrenaline secretion, engorgement of nasal epithelium, reflexive interruption of inhalation, and sneezing (Tucker 1963a, b, 1971; Cain 1988; Doty 1995).

Ammonia and Pigs

Ammonia is a colorless, highly irritating alkaline gas that forms naturally from the decomposition of organic material such as manure (Anderson *et al.* 1964). Ammonia has long been implemented as a major atmospheric pollutant in animal houses (e.g. Curtis 1972; Donham *et al.* 1977; Drummond *et al.* 1980), and as an irritant of the respiratory tract. Ammonia is soluble, can be adsorbed onto dust particles, and may cause intracellular damage when absorbed by mucus membranes (Visek 1968; Oyetunde *et al.* 1978; Al-mashhadani & Beck 1983). A link exists between NH_3 exposure and the incidence and severity of common respiratory diseases which are endemic to confined swine, such as pneumonia and rhinitis (Robertson *et al.* 1990). Chronic exposure of pigs to NH_3 concentrations of 10-100 ppm often results in irritation and histological damage to respiratory tract, altered physiology and immunology, enhanced pathogenesis of respiratory diseases, decreased feed intake, decrease growth rates, and possibly a sense of malaise (e.g. Stombaugh *et al.* 1969; Drummond *et al.* 1980; Robertson *et al.* 1990; Urbain *et al.* 1994; Gustin *et al.* 1994; Hamilton *et al.* 1996; Jones, *et al.* 1996). Ammonia concentrations of 50 ppm can inhibit pig growth; it can lead to mild respiratory disorders and is a gastrointestinal irritant (Drummond *et al.* 1980). Acute exposures, defined as single, non-repetitive exposures of not more than 8 hours, to the highest NH_3 levels recorded in commercial piggeries (100-200 ppm) may result in irritation and secretion from ocular, conjunctival, nasal, and pharyngeal membranes (e.g. Stombaugh *et al.* 1969; Curtis *et al.* 1975; Donham *et al.* 1989). Ammonia that accumulates in the air space of under-floor manure pits of pig living areas can delay the onset of puberty in gilts

(Malayer *et al.* 1987) possibly because olfactory stimulatory influence from boars is hampered by NH_3 concentrations (Malayer *et al.* 1988).

The recommended maximum level of NH_3 in piggeries is 7 ppm (Cargill & Skirrow 1997). At NH_3 levels of 10 ppm growth and feed intake depression can occur in growing pigs (Jones *et al.* 1996). Ammonia concentrations in swine houses of 1-30 ppm are common, with higher concentrations being reported (Do Boer & Morrison 1988).

Jones *et al.* (1998) studied the behavioral response of pigs to introductions of NH_3 concentrations of 40 and 100 ppm to their living areas. Novel introduction of 40 ppm NH_3 concentrations did not significantly influence pig behavior. An initial avoidance of the 100 ppm concentration was observed attributed to irritation rather than novel effects, however NH_3 at this concentration was not totally avoided and acclimatization was described as rapid. The concentrations used in this study were consistent with moderate (40 ppm), and peak (100 ppm) concentrations recorded in commercial piggeries (e.g. Donham *et al.* 1977; Groot Koerkamp *et al.* 1996; von Hoy *et al.* 1994). After initial acclimatization, exposures to even higher concentrations may be necessary for pigs to become aversive again, as shown in humans to other irritants (Katz & Talbert 1930). Previous work reported swine avoidance to chronic exposure to 40 ppm concentrations after 35-40 minutes of exposure (Jones *et al.* 1996) and found that pigs would overcome initial preferences in order to avoid NH_3 (Smith *et al.* 1996).

Ammonia and chickens

Ammonia is the most plentiful aerial pollutant from poultry facilities (Wathes *et al.* 1983). The recommended exposure limit of NH_3 for poultry is 25 ppm (MAFF 1987). Of the total nitrogen consumed by commercial poultry, atmospheric losses can reach 18.4, 31.5, and 40.0% for commercial broilers (Patterson *et al.* 1998), pullets (Patterson & Lorenz 1997), and laying hens (Patterson & Lorenz 1996), respectively.

Ammonia and Humans

The literature on ammonia toxicity in humans largely consists of case reports. In a 1996 literature review, de la Hoz *et al.* found only 94 previously reported cases; of these cases, 20 resulted in fatality and only 35 required clinical follow-up of one year or more. Despite lack of data, most literature is consistent regarding clinical presentation and treatment of ammonia toxicity.

Because of high water solubility ammonia has a tendency to absorb into water-rich mucosa. Unlike most highly water-soluble irritants that tend to affect the upper respiratory tract exclusively, ammonia can damage proximally and distally. In animal confinement buildings, ammonia is adsorbed onto dust particles that transport it more directly to small airways. Because of this synergistic effect, symptoms (as described below) may be reported within minutes of entering confinement buildings. Over 50% of swine building operators have reported respiratory stress (Donham 1982; Donham *et al.* 1977).

In humans acute exposure to NH_3 can be uncomfortable and painful as the exposure not only stimulates the olfactory nerve but also sensory endings in the trigeminal, vagus and glossopharyngeal nerves in the nose, mouth and pharynx, which is collectively known as the 'common chemical sense' (e.g. Allen 1937; Tucker 1971; Cain 1974, 1988). The common chemical sense is stimulated in proportion to ammonia concentration (Cain 1976). Higher NH_3 concentrations are likely to be both novel and irritating, while lower NH_3 concentrations are more likely to be just novel. Ammonia is associated with nuisance odors (Liu *et al.* 1993).

Humans are known to acclimatize to ammonia (Farbman 1992; Ferguson *et al.* 1977; Stombaugh *et al.* 1969), meaning that ammonia causes olfactory fatigue or adaptation making its presence difficult to detect when exposure is prolonged. Negative responses and health impacts are well accepted and standards have been adopted to protect workers from ammonia exposure. The United States Department of Health and Human Services' Agency for Toxic Substances & Disease Registry (ATSDR 2007) provides the following information concerning ammonia and its impact on human health.

- Inhalation of ammonia may cause burns to nasalpharyngeal and pharyngeal areas, edema to bronchiolar and alveolar tissues, and destruction of the airway resulting in respiratory distress or failure. Irritation of eyes and respiratory tract can be followed by swelling and narrowing of the throat and bronchi, coughing, and accumulation of fluid in the lungs can occur. Upper airway swelling and pulmonary edema may lead to airway obstruction. This may cause low blood oxygen and an altered mental status. Ammonia has a greater tendency to penetrate and damage the eyes more than any other alkali. Even low concentrations of ammonia vapor (100 ppm) can produce rapid onset of eye irritation, swelling, and sloughing of surface cells of the eye, which may result in temporary or permanent blindness. Survivors of severe injury due to acute exposure to airborne NH_3 may suffer chronic effects to the eyes or respiratory tract. No data exists to evaluate the reproductive and developmental effects of ammonia in humans, but decreased egg production and conception rates have been observed in animals, and NH_3 has been shown to cross the ovine placental barrier.

- Exposure to NH_3 can be acute or chronic. The odor threshold for ammonia is low and it is detected as a pungent odor at concentrations as low as 5 ppm for most individuals, which provides adequate warning of its presence. Eye irritation can develop at 20 ppm. The U.S. Department of Labor, Occupational Safety and Health Administration's (OSHA) Permissible Exposure Limit (PEL) for gaseous ammonia is 50 ppm. PEL is a Time Weighted Average (TWA) limit meaning that this level should not be exceeded as an average concentration over an 8 hour period. The National Institute for Occupational Safety and Health (NIOSH) lists the concentration of NH_3 that is Immediately Dangerous to Life or Health (IDLH) as 300 ppm (NIOSH 2005). The maximum airborne concentration below which it is believed that nearly all individuals could be exposed for up to 1 hour without experiencing or developing irreversible or other serious health effects or symptoms which could impair an individual's ability to take protective action is 200 ppm for NH_3.

- Irritation and burning sensations of eyes and mucous membranes can be attributed to the formation of ammonium hydroxide when NH_3 reacts with moisture in the membranes. This alkaline solution can be corrosive to mucous membranes of the eyes, lungs and gastrointestinal tract.
- Ammonia gas is about 40% lighter than air and tends to rise; however vapors from liquefied NH_3 gas are initially heavier than air and may not rise, spreading along the ground. Asphyxiation may occur in poorly ventilated or enclosed spaces.
- Flammable concentrations of ammonia in air range from 16-25%, however the ignition temperature of 650 °C makes the gas difficult to burn.
- The boiling point of ammonia is -33.4 ºC, which means that it will be in a gaseous form in most agricultural settings. Ammonia has a water solubility of 33.1% at 20 ºC and can be found in aqueous solutions at ambient temperatures
- Gaseous ammonia effects for humans at various concentrations are as follows:
    - 25-50 ppm - Detectable odor; unlikely to experience adverse effects
    - 50-100 ppm - Mild eye, nose, and throat irritation; may develop tolerance in 1-2 weeks with no adverse effects thereafter
    - 140 ppm - Moderate eye irritation; no long-term sequelae in exposures of less than 2 hours
    - 400 ppm - Moderate throat irritation
    - 500 ppm – Immediately Dangerous to Life and Health (IDLH)
    - 700 ppm - Immediate eye injury
    - 1000 ppm - Directly caustic to airway

- 1700 ppm - Laryngospasm
- 2500 ppm - Fatality (after half-hour exposure)
- 2500-6500 ppm - Sloughing and necrosis of airway mucosa, chest pain, pulmonary edema, and bronchospasm
- 5000 ppm - Rapidly fatal exposure

In the atmosphere ammonia reacts with gaseous nitric acid (HNO_3) and sulfate (SO_4^-) to form ammonium nitrate (NH_4NO_3) and ammonium sulfate ($(NH_4)_2SO_4$), respectively. These two molecules are prevalent forms of particulate matter, which can contribute to a variety of adverse health effects including premature mortality, chronic bronchitis, hospital admissions, and asthma attacks (McCubbin *et al.* 2002). Reducing NH_3 emissions may not initially decrease the atmospheric concentrations of ammonium sulfate and ammonium nitrate because the limiting concentration in air may be the levels of HNO_3 and SO_4^-. Ammonia may need to be reduced to a level where it becomes the limiting reactant. The concentration ratios between NH_3 and HNO_3, and NH_3 and SO_4^- may vary according to local ammonia emissions.

Ammonia Emissions from Manure

Ammonia release is a major pollution problem, playing an important role in atmospheric chemistry and acid deposition (Kamin *et al.* 1979) and can affect ecosystems at relatively low concentrations (Genfa *et al.* 1998). Environmental consequences of ammonia include aerosol formation in the atmosphere, and deposition impacts of eutrophication and soil acidification from ammonia and molecules formed from its presence in air (Roeloffs & Houdijk 1991; Aneja *et al.* 2001). Once released, removal of atmospheric NH_3 may occur through wet or dry deposition. With an atmospheric lifetime expectancy of less than 1-5 days (Warneck 1988) NH_3 is likely to be deposited on the Earth's surface near its source. Some ammonia is converted to aerosol ammonium, which has a lifetime of 1-15 days and is more likely to travel further distances prior to deposition (Aneja *et al.* 1998). The rate of conversion from NH_3 to NH_4^+ is largely unknown, but the reaction rate depends largely on acid concentration, humidity, and temperature of the air (Seinfeld & Pandis 1998). Atmospheric NH_3 can also react with acidic species in air such as sulfuric, nitric, or hydrochloric acids (H_2SO_4, HNO_3, or HCl) (Aneja *et al.* 2001). Approximately 10% of airborne NH_3 is oxidized by hydroxyl radicals (OH) to form amide radicals (NH_2) (Finlayson-Pitts & Pitts 1996). Removal of atmospheric NH_3 may also occur through wet or dry deposition.

Globally, domestic livestock contribute the largest emission of atmospheric NH_3 with an estimated annual emission ranging from 20-35 Tg N/yr (Bouwman *et al.* 1997; Warneck 1998). Agriculture accounts for 80-90% of total global NH_3 emissions (Pain *et al.* 1998). Other global sources include soils, biomass burning, and vehicle emissions. In Europe, 90% or more of anthropogenic NH_3 emissions result from agriculture, specifically livestock and fertilizer (Buijsman *et al.* 1987; Krapfenbauer & Wriessnig 1995), contributing to acidification and forest die-back in Western Europe (van Breeman *et al.* 1982; Fangmeier *et al.* 1994, Slanina 1994). In most Asian countries livestock and fertilizer account for 77% of anthropogenic NH_3 emissions, with livestock alone accounting for about 30% of total emission (Zhao & Wang 1994). In the US, the state of North Carolina where costal waters are impaired, 35-60% of the nitrogen load to the damaged waters is associated with atmospheric deposition of NH_3. About 47% of North Carolina's total NH_3 emissions originate from the state's swine manures (Aneja *et al.* 2001). The total annual ammonia emissions in the United States originating from animal husbandry operations in 2002 were estimated to be over 2.4 million tons with approximately 558 000, 657 000, 664 000, and 429 000 tons emitted from dairy, beef, poultry, and swine operations, respectively (EPA 2004). Ammonia loss from manure decreases its fertilization value (Pain, *et al.* 1989), with an estimated loss of 2-10% of total nitrogen in slurry and 17-23% for farmyard waste (Svensson 1991) attributed to ammonia volatilization.

Ammonia exists in liquid manure in the form of ammonium (NH_4^+), and as free ammonia (NH_3). The sum of NH_4^+ and NH_3 expressed as a mass is termed Total Ammonia (TA) and includes the mass of hydrogen nuclei. The interchangeable terms Total Ammoniacal Nitrogen (TAN) and Ammoniacal Nitrogen (AN) refer to only the mass of the N nuclei found in the solution (Ni 1999). Because ammonium and water are polar molecules, NH_4^+ is not likely to volatilize from solution.

Most ammonia emissions originate from the hydrolysis of urea ($(NH_2)_2CO$) excreted from cattle and hogs (Pfeiffer & Henkel 1991; Näsi 1993; Aarnink *et al.* 1996), and uric acid ($C_5H_4N_4O_3$) from poultry (McCubbin *et al.* 2002). Catalyzed by the enzyme urease, conversion of urea forms ammonia and carbon dioxide in accordance to the equation $(NH_2)_2CO + H_20 \rightarrow CO_2 + 2\ NH_3$. When urease activity is high the limiting factor of NH_3 emission is the urea supply (Aarnink & Elzing 1998; Muck & Steenhuis 1981). The conversion of uric acid is catalyzed by uricase enzyme and involves several chemical steps. These avenues for NH_3 formation can occur in both aerobic and anaerobic conditions (Zhang *et al.* 1994). The principles described herein apply to manure slurry, manure on floors, and urine.

Urea concentrations in slurry and ammonia emission are positively related (as calculated by Aarnink & Elzing 1998). Research with dairy manure has affirmed this relationship (Elzing & Kroodsma 1993). The conversion of urea to NH_3 can occur with little lag time once urine is deposited on a floor or into a manure storage with high urease activity (Elzing & Swierstra 1993; Aarnink *et al.* 1996). Generally, the amount of urine added to manure slurry is small in comparison to the overall volume of the existing slurry. This means that small amounts of urease activity within the slurry are sufficient to quickly convert urea to ammonia (Aarnink & Elzing 1998). However, the addition of urine to manure slurry can increase TAN on the manure surface, which can lead to increased NH_3 emission rate (Aarnink *et al.* 1996).

A close relationship exists between urea concentration and total ammoniacal nitrogen concentration in slurry. A 42% decrease of urea concentration in cattle manure yielded a 39% decrease of NH_3 emissions (Smits *et al.* 1995). Diet modifications may decrease NH_3 emissions. In research that used a fixed water to feed ratio to decrease nitrogen excretion from urine by 14.7%, ammonia emissions were reduced by an average of 10.7%, although the researchers reported that an interaction between feeding and housing treatments seemed to exist (Van der Peet-Schwering *et al.* 1996). The frequency of urination is positively related to NH_3 emissions (Aarnink *et al.* 1996). This could lead to diurnal patterns in NH_3 emissions that mimic patterns of urination frequency of pigs within a housing unit (Aarnink *et al.* 1996).

Other factors that influence ammonia emissions are pH (Stevens *et al.* 1989), ammoniacal nitrogen content (Elzing & Kroodsma 1993), emitting surface area (Hartung & Büscher 1995; Aarnink *et al.* 1996) air temperature (Muck & Richards 1983), and air speed (Olesen & Sommer 1993; Zhang *et al.* 1994).

Ammonia is very soluble in water therefore its release from manure slurry solution is a slow process (Srinath & Loehr 1974; Muck & Steenhuis 1981; Freney *et al.* 1983). The rate and amount of ammonia emission from manure depends on the rate of NH_3 formation, the transport of NH_3 to the manure surface, and factors that influence volatilization. Models are commonly used to predict amounts and rates of NH_3 release, originating from manure. Ni (1999) reviewed thirty models that predict NH_3 release. A common trait of all models was a physical insight based on; (1) enzymatic and microbial generation of NH_3, (2) diffusion mass transfer of NH_3 in manure, (3) chemistry of NH_3 in aqueous solution, and (4) convection mass transfer of NH_3 gas from manure surface to free air stream. Some important factors of NH_3 release represented in most models include the free ammonia concentration in the manure, pH of the manure, temperature of manure, temperature of air above the manure, air velocity over manure surface, and gaseous NH_3 concentration at the manure surface. The rate of release of NH_3 from manure is a function of release surface area, convection mass transfer coefficient, and

NH_3 concentration difference between the liquid and air stream (Ni 1999). The basis of models is represented by Ni in an illustration of the mechanism of ammonia release from manure. Figure 6.1 is representative of the diagram developed by Ni.

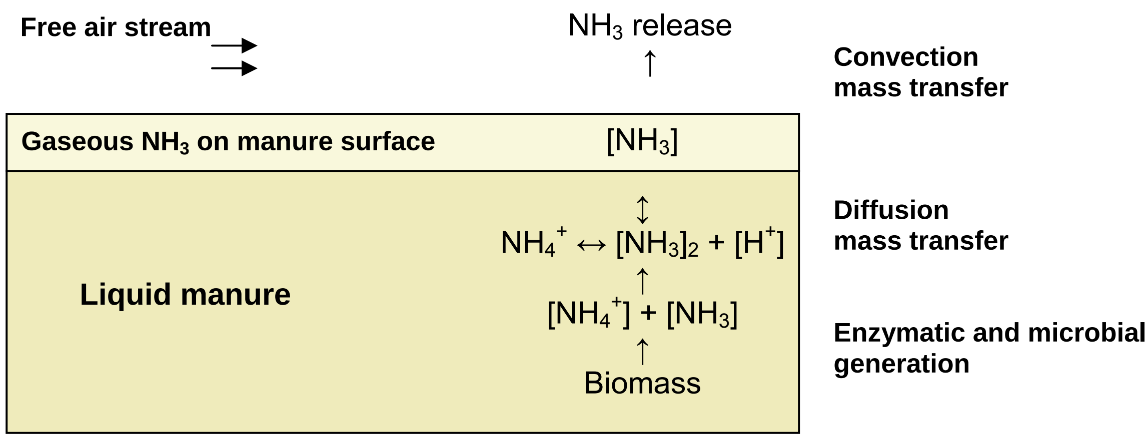


Figure 6.1. Illustration representing the mechanism related to ammonia release from manure. (Adapted from Ni 1999).

A factor which is positively related to the release of ammonia to the atmosphere is temperature (Aneja *et al.* 2000; Chauhan 1999; Sommer 1997; Dewes 1996; Aarnink *et al.* 1995; Sommer, *et al.* 1991; Muck & Steenhuis 1982). Higher temperatures facilitate the production of NH_3 by enhancing the growth of microbes that decompose urine and feces, while at the same time increasing volatilization (Aneja *et al.* 2001).

The release of gas from the liquid follows Henry's Law, which states that the NH_3 concentrations in both the liquid portion of manure and the free air stream above the manure will move towards a liquid-gas phase equilibrium. Henry's Law does not apply to TAN, but rather only to free NH_3. At a given temperature the NH_3 concentration in manure is related to equilibrium partial pressure in air for NH_3 (Zhang *et al.* 1994). Increases in temperature of the manure will increase NH_3 release rate because the higher temperatures increase the generation rate of NH_3 by microbes in the manure, which increases the free ammonia concentration in the manure, thus increasing the potential for release. Higher temperatures also enhance the diffusion ability of NH_3 in manure. This increases the rate at which NH_3 can diffuse to the manure surface (Zhang *et al.* 1994).

In accordance with Fick's Law the mass diffusion of ammonia from the manure aqueous solution to the free air stream above the manure forms a concentration gradient which moves toward equilibrium. This convective mass transfer from higher concentrations found in the manure to lower concentrations found in air creates a flux of NH_3. Flux in this situation refers to the transfer of ammonia to the air per unit area of manure surface over time (e.g. kg m^{-2} s^{-1}). Convective mass transfer across the surface of manure is the most important aspect of NH_3 release models. A concentration gradient exists in the surface region, which is incorporated into most models as a two-film layer or a boundary layer concept. Two-layer theory was developed in 1923 by Whitman (as reported by Welty *et al.* 1984), and has been used in models of ammonia release from swine manure (e.g. Anderson *et al.* 1987; Zhang 1992; Cumby *et al.* 1995). The two-layer theory separates the boundary between the manure surface and air stream into liquid and gas film layers. Diffusion is predicted in (1) the manure liquid, (2) between the manure liquid and the liquid film layer, (3) between the liquid and gas film layers, and (4) between the gas film and the free air stream. The latter diffusion can be referred to as a mass convection. This theory for NH_3 release from manure is represented in Figure 6.2 (adapted from Ni 1999).

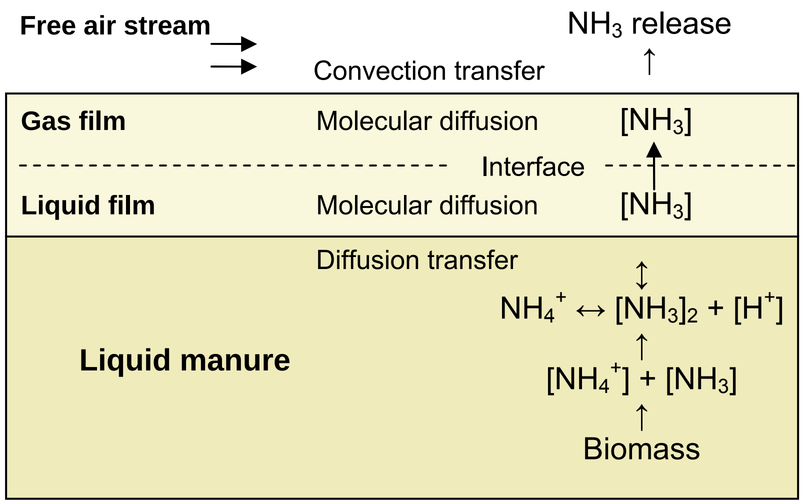


Figure 6.2. Illustration representing the mechanism of ammonia release using the two-layer theory as a model. (Adapted from Ni 1999).

Boundary layer models are simpler in structure than two-layer models. The boundary layer is the region where NH_3 concentrations differ between the free air stream and the manure surface; a concentration gradient exists in this region. Conditions within the boundary layer determine convective mass transfer. These models consider only the convective transfer of NH_3 from the manure surface to the air stream and the diffusion transfer inside the bulk manure. Boundary layer models can be used to predict NH_3 release from swine manure (e.g. Olesen & Sommer 1993).

Ammonia release flux rates from swine slurry have been investigated and reported by several authors. In 1995, Andersson obtained a release flux rate of 103 mg/m^2 h at a temperature of 16.2 ºC. Aarnink *et al.* (1996) reported NH_3 emissions from a swine house in the Netherlands, and Ni *et al.* (2000a) calculated the flux from the study to be equivalent to 350 mg/m^2 h. The release flux from a 135 day test at a commercial pig house in Belgium yielded an average flux rate of 449 mg/m^2 h (Ni 1998). Ni *et al.* (2000a) reported the flux rates from overnight tests of two stocked swine finishing manure pits as 233 and 319 mg/m^2 h. Total mass of emissions is reported as 0.85 g/pig/day for nursery pigs and 6.10 g/pig/day for finishing pigs (Aarnink *et al.* 1996). Finally, in a study of six outdoor anaerobic lagoons, ammonia release flux rates were found to positively relate to both manure surface temperature and aqueous ammonia concentrations by Aneja *et al.* (2001) in a study that found a range of flux rates of 40.7-120.3 μg N/m^2 min (2.4-7.2 mg/m^2 h).

In solution, ammonia and ammonium follow the equilibrium equation $NH_3(aq) + H_2O(aq) \leftrightarrow NH_4^+(aq) + OH^-(aq)$ (Warneck 1988). The equilibrium direction is controlled by pH. Increasing pH implies an increase in the hydroxyl ion (OH^-), which will react with ammonium to shift the equilibrium to the left. This increases NH_3 concentration, thereby increasing NH_3 liberation from solution. The addition of water will have the opposite effect, shifting the equilibrium equation towards NH_4^+ and lowering ammonia release. As ammonia is released from solution, hydrogen ions (H^+) are left behind, which can result in lower pH. The addition of fresh manure will cause shifts in slurry pH. When manure is added to the system the hydrogen ions commonly react with bicarbonate ions (HCO_3^-), a product of hydrolysis of urea and microbial degradation of organic matter, following the equation $CO_2(aq) + H_2O(aq) \leftrightarrow HCO_3^-(aq) + H^+(aq)$ (Sommer *et al.* 1991). For this reason the pH of manure slurry is expected to rise as organic matter is added and bicarbonate reacts with the hydrogen ion. Because CO_2 is much less soluble than ammonia it will quickly release from the solution. After the organic material is decomposed and CO_2 and NH_3 escape the pH will decrease (Aneja *et al.* 2001). A calculated NH_3 emission increase of 9.0% was predicted for every 0.1 unit pH increase in manure (Aarnink & Elzing 1998). Lowering of the pH of the top layer of swine manure in a scale model pig house by 0.2 units decreased NH_3 emissions 7-18% (Elzing & Aarnink 1996). In a real swine facility a decrease of one unit of pH was shown to equate to decrease in NH_3 emissions of about 10% (Aarnink *et al.* 1996). The pH of the top layer of mixed swine slurry was clearly higher than the rest of the slurry. Three days after mixing, the top layer had a pH reading 0.84 units higher than the initial pH of the mixed slurry (Olesen & Sommer 1993). Higher concentrations of CO_2 and VFAs generally lower pH, and pH increases with higher levels of NH_3. Compared to NH_3, CO_2 volatilizes 5.5 times faster (Husted *et al.* 1991), is less soluble, and transports to the surface area more slowly (Olesen & Sommer 1993). For these reasons the CO_2 to NH_3 ratio is usually lower on slurry surface and seems to be the main reason for higher pH on the surface.

Zhang *et al.* (1994) describes the release response to ventilation rate increases, stating that as the movement of free air across the manure surface increases so does the release rate of ammonia. However, as the ventilation rate operates at the higher velocity the release rate will eventually decrease due to faster decreases of NH_3 at the manure surface. In this situation the diffusion of NH_3 towards the manure surface is slower than the release rate. Once the initial high flux is removed into the air stream the release rate of NH_3 may actually be lower at the high air velocity than if the ventilation had remained at a lower velocity. This may not impact the total amount of NH_3 released because the total amount of NH_3 production remains unchanged.

In commercial swine houses there is always some amount of fouling and dust on floor space. Urine puddles tend to be deeper in dirty areas, which can increase NH_3 emissions due to the larger volume of urine that would occupy the same area of a clean floor. Volume of urine is more important to total emissions than is depth or size of puddle (Aarnink & Elzing 1998).

Ammonia and Manure Pit Storage

The amount of manure stored in an under-floor storage has no significant impact on ammonia emissions. It is surface area of manure, not volume, which is proportional to NH_3 release (Ni *et al.* 1999). The continuous feeding of manure pits in commercial piggeries affects the air volume in the pit above the manure. This change may influence airflow patterns inside the pits.

The concentration of NH_3 will change within the depth of a manure store. Zhang *et al.* (1994) reported that NH_3 concentration variation was small in different depths of a manure pit when pit depth was low because NH_3 could easily diffuse to the surface and release. As manure depth increased, stratified layers within the manure pit depth increased in thickness, slowing diffusion to the surface. Meanwhile microbial activity in the manure continues to produce NH_3. These factors cause the concentration of NH_3 to become stratified and increase as depth increases. With increases of NH_3 concentrations in lower pit depths, the concentration of NH_3 near the surface did not increase significantly.

Ammonia and Temperature and Ventilation Rate

In a study of swine manure gas release Ni *et al.* (2000a) measured free air stream concentrations of NH_3, H_2S, and CO_2 above under-floor manure pits in a swine finishing building and found nearly immediate increases in all three gas concentrations when propane heaters in the building turned on. The response occurred in less than 10 minutes. The accelerated release flux most likely occurred due to increased manure surface temperature through radiant heat. The study compared various ventilation schemes in the buildings. Ventilation removes air that contains pollutants that have emitted from manure and also introduces fresh air to the animal living space that dilutes and exhausts pollutant laden air found in the building. Ni *et al.* (2000a) found that pit ventilation fans at a high airflow rate (21 000 m^3/h) sufficiently ventilated ammonia from the building, but when the pit fans operated at a lower airflow rate (9100 m^3/h) a zone of elevated NH_3 could be found in the center of the room. The authors concluded that tunnel ventilation (22 000 m^3/h) appeared more effective for diluting all three pollutant gases. Authors have observed only small effects on NH_3 emissions when air was exhausted through the ceiling verses underneath or just above slatted floors (Jungbluth & Büscher 1996; Aarnink & Wagemans 1997). When sufficient space is provided between the slatted floor and the manure surface little air movement occurs above the slurry surface (Jungbluth & Büscher 1996). Ventilation systems can cause variations in air flow patterns and velocities near slurry surfaces (Randall *et al.* 1983). Diurnal ventilation variation may lead to daily patterns in NH_3 emissions.

Ammonia and Floor Type

The type of floor in swine housing has some bearing on the NH_3 concentration in the air of the pig house. In swine houses with both solid (38%) and slatted floors (62%) above manure storage pits, approximately 1/3 of NH_3 emissions originate on the slatted floor and 2/3 from under-floor pits, calculated as 60-70% from pits (Hoeksma *et al.* 1992). Another study reported that in pens with 25% and 50% slatted floors, 40% and 23% of NH_3 emissions originated from the slats, respectively (Aarnink *et al.* 1996). Surface area of dung in these buildings affects NH_3 emissions (Hesse 1994; Aarnink *et al.* 1996; Jungbluth & Büscher 1996). Because floor contamination increases the surface area of manure; a positive linear relationship between steady state NH_3 emissions and area of floor contamination exists (Hesse 1994, Ni *et al.* 1999). Higher floor contamination is related to weight of pigs and inside temperature (Hoeksma *et al.* 1992; Ni *et al.* 1999). Greater areas are befouled in summer than winter (Voermans & Hendriks 1995). The influence of ventilation rate and inside air temperature on NH_3 release is stronger when floors have higher contamination rates (Ni *et al.* 1999). In most agricultural settings manure surface area is influenced by animal activity, with a renewal of the surface layer of the manure on the floor or in the pit with excretion, urination, or other physical disturbance. These disruptions of surfaces where NH_3 is released make floors and pits continuously fed systems (Ni *et al.* 1999).

When walls, penning, or pig bodies are partially covered with manure the surface area of manure is increased and ammonia can release from this manure. However, the quantity of NH_3 emitted from these sources is insignificant when compared to floor and pit emissions (Ni *et al.* 1999).

Temperature, ammonia concentration, and pH are significant predictors of lagoon ammonia flux. Seventy-five percent of the variation in daily average NH_3 flux from lagoons can be explained by ammonia concentration and manure surface temperature (Aneja *et al.* 2001). Other factors that affect ammonia flux in a manure system include animal numbers, animal weight, animal feeding patterns, variations in the amount of manure added, level of storage, and the addition or evaporation of water (Chauhan 1999).

Ammonia Gas Concentrations in Pig Houses

There are several strategies that can be employed to reduce ammonia emissions. The emission of ammonia could be cut in half by decreasing the surface area of a manure storage (Muck *et al.* 1984). Manure storage covers offer the greatest potential reduction of NH_3 loss, reducing NH_3 emissions by 50% (Nicholson *et al.* 2002). Covers were found to reduce NH_3 emissions from swine manure by up to 93% in a laboratory and 68% in the field (Williams & Nigro 1997). Crusts on manure storages can decrease NH_3 emissions. Slurries that contain bedding material that can assist crust formation provide a simple way of reduction. A crust can reduce NH_3 emissions to 20% of that released from a stirred storage (Sommer *et al.* 1993). Swine manure typically contains no bedding material, however a floating layer made of straw or oil can reduce NH_3 emissions by 90%

(Hörnig *et al.* 1999) though the stability of floating layers may disintegrate during filling, mixing, or manure removal.

Treatment of manure can help reduce ammonia emissions by converting ammonium in the manure into nitrite and nitrate through nitrification. Cheng *et al.* (2004) found success in this conversion using a trickling nitrification biofilter. The work treated swine manure from an anaerobic digester in North Carolina and converted almost 90% of the ammonium to nitrate over time and during warm weather the conversion was almost complete (100%), but lowered when weather cooled the ambient system. The effluent from the biofilter contained nitrogen that was more readily available to plants when used as fertilizer.

Swine diets can be formulated to minimize nitrogen excretion and thereby limit the amount of ammonia emissions from manure. Adapting diets of growing or breeding pigs to actual physiological demand can help keep emissions low (Gruber & Steinwidder 1996). Two-phase feeding of sows in gestation and lactation periods can reduce NH_3 emissions by 12% (Kirchgessner *et al.* 1993) and further reductions are found with four-phase feeding (Andree & Heege 1997; cited by Döhler *et al.* 1999). Nitrogen excretions will be lower when feed formulation optimizes amino acid utilization (Gruber & Steinwidder 1996).

Reports of ammonia concentrations in swine housing facilities are varied (Table 6.1). This can like be attributed to variations in the factors influencing concentration such as those discussed herein.

Table 6.1. Ammonia concentrations in swine housing facilities from various research data.

Source	NH_3 concentration (ppm)
Lebeda *et al.* 1964	8.1
McAllister & McQuitty 1965	4-14 000
Robertson & Galbraith 1971	8.5-17.5
Grub *et al.* 1974	16-54
Skarp 1975	20-75
Ni *et al.* 2000a	15.2 ± 0.6
Ni *et al.* 2000a	17.2 ± 1.7

Hydrogen Sulfide in Swine Housing

Hydrogen sulfide is a noxious gas emitted from fermentation of manure and considered the most dangerous gas at concentrations > 1400 ppb and has been responsible for many human and animal deaths (Donham *et al.* 1982). Intoxication for humans is reported as chronic, subacute, and acute at concentrations of 70-140 ppb 140-1400 ppb, and >1400, respectively (Smith *et al.* 1979). Hydrogen sulfide is reportedly dangerous to humans and animals at concentrations ≥280 ppb (Ni *et al.* 2002). The US Occupational Safety and Health Administration publishes guidelines for time-weighted average (8 h) exposure of 14 ppb and a short-term exposure limit of 21 ppb (OSHA 1999).

Information on H_2S release and its dynamics from swine manure is lacking in literature. Avery *et al.* (1975) reported that concentrations of H_2S in the air of a swine facility are correlated with air temperature, pit-to-room volume, and air-retention time within the building. The authors observed H_2S concentrations over a wide range, from 120-1274 ppb. Other reports of swine housing air concentrations for H_2S include 0.127 ppb in a typically ventilated swine building and 0.396 ppb after ventilation shut off for 6 hours (converted by Ni *et al.* 2002 assuming 20 °C and 1.013 X 10^5 Pa atmos. pressure; Muehling 1969). In two naturally ventilated pig houses concentrations were 0.235 ppb over 63 days (Heber *et al.* 1997). In a mechanically ventilated building when deep-pit slurry was agitated, H_2S concentration was 141 ppb (Patni & Clarke 1991).

Ni *et al.* (2000a) reported a novel phenomenon that the authors termed 'H_2S burst', which occurred in both occupied and unoccupied buildings and lasted from 1 to 5 hours. The peculiar behavior of H_2S was characterized by burst releases and could not be explained by any known factors in the tests. The burst is described as a sudden increase of the release of H_2S of more than two times the previously recorded level. In a subsequent study Ni *et al.* (2002) published a release flux rate for H_2S from deep pit finisher swine manure as 0.74 g/d/m^2. Some other published release rates include 0.12, 0.39, and 2.29 g/d/m^2 from three deep-pit finishers in Minnesota (Jacobsen *et al.* 1999); 0.59 g/d/m^2 average over 30 days from five deep-pit naturally ventilated swine finishers (Bicudo *et al.* 2000); and 0.44 and 0.70 g/d/m^2 as measured from mechanically and naturally ventilated buildings, respectively (Zhu *et al.* 2000).

The amount of H_2S emitted from manure is typically much lower than other noxious gases such as NH_3 and CO_2 (Ni *et al.* 2002). Unlike, NH_3, which shows diurnal emission variation related to pig urination patterns (Aarnink *et al.* 1993), ventilation rate, and temperature, diurnal variation for H_2S release appears to be more complex with variations occurring when animal activity, ventilation, and temperatures are relatively stable, for example, at night and early morning (Ni *et al.* 2000 b,c). Ni *et al.* (2002) report that H_2S release is directly and positively related to temperature and ventilation rate. This is attributed to general positive effect of higher temperatures to enhance physical, chemical, and biological processes that contribute to H_2S generation and release, as well as significant influence on convective mass transfer of pollutants from liquid manure to the free air stream when ventilation rates increase. For these reason H_2S emissions can be expected to be greater during summer months. Similar to findings with NH_3, pig size and depth of manure in under-floor storage pits did not appear to have impact on H_2S release in this study.

Carbon Dioxide in Swine Housing

Carbon dioxide production from swine manure can be a concern for health of swine and human workers. Respiration also contributes to CO_2 concentrations. Health concerns such as sneezing, cough, and pneumonia are greater for animals in circumstances where CO_2 concentrations are 2000-9000 ppm compared to that of situations where the levels are 1000-3000 ppm (Busse 1993). The suggested maximum CO_2 exposure concentration for animals is 3000 ppm and for workers is 5000 ppm (CIGR 1992). Carbon dioxide is produced microbially during aerobic and anaerobic degradation of manure. Carbon dioxide is also produced during the combustion of fossil fuels and its atmospheric concentration has increased by 26% since pre-industrial times (Houghton *et al.* 1990).

Nitrous Oxide in Swine Housing

Nitrous Oxide (N_2O) is a greenhouse gas produced from manure under aerobic conditions with a Global Warming Potential (GWP) of 296 for a 100-year time horizon (IPCC 2001). In the stratosphere N_2O emissions deplete ozone when it is converted to nitric oxide (NO) (Olivier *et al.* 1998). The atmospheric concentration of N_2O has increased by 7% since pre-industrial times (Houghton *et al.* 1990). Emission of N_2O from animal digestive systems is not known and is likely negligible (Kroeze 1998). N_2O is not directly formed from compounds present in feed or manure; rather it is a result of the conversion of urea or uric acid into ammonium, which is consumed by nitrifying bacteria in the presence of oxygen. Nitrous oxide is an intermediate reaction product in the denitrification process (Monteny *et al.* 2001).

Because manure slurries are frequently stored with largely anaerobic conditions N_2O is formed only in the manure surface/free air boundary area or in manure crust (Wulf *et al.* 2006). Globally, livestock manure contributes to 7% of N_2O emissions (Khalil & Rasmussen 1992; Mosier & Kroeze 1998). N_2O emissions from manure are positively influenced by temperature (Sommer *et al.* 2004). Microbial activity in aerobic manure can produce heat, which will enhance further microbial growth and production of N_2O (Clemens & Ahlgrimm 2001). Anaerobic digestion decreased the emissions of N_2O from manure by more than 50% when the manure was field applied, because the volatile solids in the manure are reduced (Sommer *et al.* 2004). No N_2O emissions are formed during anaerobic digestion of manure (Clemens & Ahlgrimm 2001).

Methane in Swine Housing

Methane (CH_4), produced anaerobically, is released from manure through a bubbling process known as ebullition (Wulf *et al.* 2006). The concentrations of CH_4 rise very quickly in the airspace above manure. Martinez *et al.* (2003) found that the CH_4 concentration in fresh air, with a background CH_4 concentration of 10-20 ppm (0.001-0.002%), introduced above raw manure had increases to concentrations of 100-1000 ppm (0.01-0.1%) after only a few minutes, 1-5% after several hours, and frequently reached 20-30% after 2-3 days. The study confirmed rapid generation and release of CH_4 from raw pig slurry, varying from a few g [CH_4 C]/m^3 d to up to nearly 100g [CH_4 C]/m^3 d.

Aeration can decrease CH_4 emissions from manure by 70-100% because most carbonaceous emissions occur in the form of CO_2 from aerobic degradation. This aerobic decomposition can lead to increased N_2O emissions (Burton *et al.* 1993; Béline *et al.* 1999). Steed & Hashimoto (1994) described the mass balance of carbon under anaerobic conditions as $C_{initial} = C_{remaining} + C_{CO2} + C_{CH4}$; where $C_{initial}$ is the initial amount of carbon in manure, $C_{remaining}$ is the carbon remaining at the end of the storage period, and C_{CO2} and C_{CH4} are the carbon from CO_2 and CH_4 that evolve from the manure during the storage period. Using this approach Martinez *et al.* (2003) showed that 93.8% (s.d.=9%) of the carbon mass balance could be accounted for during a 50 day storage period.

Methane production in animal agriculture is an endogenous process with cattle producing the largest amount of methane in agriculture (IPCC 2001). Enteric ruminant emission of CH_4 is about 10% of digestible feed intake (Corré & Oenema 1998) or 5.5% of gross energy intake (Pelchen *et al.* 1998). Methane is also produced by pigs and other monogastric animals during the digestive process by anaerobic bacteria fermentation in the hindgut (Kirchgessner *et al.* 1993, 1994). From the pig diet, methane originates from less than 1% of the digestible feed intake (Corré & Oenema 1998), and around 0.6% of gross energy intake (Crutzen *et al.* 1986). Diets fed to pigs with highly bacterially fermentable substances, such as cellulose and hemicellulose, can lead to greater CH_4 production during digestion in pigs than other diets. Christensen & Thorbek (1987) measured CH_4 emissions from the breath and flatus of pigs, finding emissions ranged from about 1 liter (0.67 g; specific mass of CH_4 is 0.67 kg/m^3 at 20ºC) per day from 20-25 kg pigs to 12 liters per day from 120 kg pigs. Methane emissions from swine account for only about 1% of those of dairy cattle so strategies for reduction of CH_4 emission for swine may not be necessary (Clemens & Ahlgrimm 2001). No efficient or economic means exist to remove CH_4 from exhausted air of animal housing facilities (Hahne *et al.* 1999). Strategies to reduce CH_4 emissions should focus on using emissions as fuel or to prevent its creation through control of critical processes and influential factors of CH_4 formation (Monteny *et al.* 2001).

A summary of annual global emissions of CH_4 and NO_2 are presented in Table 6.2. Based on these reports the proportions of total emissions originating from livestock production are 19% of total methane emissions and 35% of total nitrous oxide emissions.

Table 6.2. Annual global emissions of methane and nitrous oxide in Tg (1 Tg = 10^{12} g). The emissions labeled Livestock production are a portion of anthropogenic emissions.

Annual emissions (Tg)	CH_4	N_2O
Natural	160 (Houghton et al 1996)	9.6 (Kroeze et al 1999)
Anthropogenic	375 (Houghton et al 1996)	8.0 (Kroeze et al 1999)
Global Total	535 (Houghton et al 1996)	17.7 (Kroeze et al 1999)
Livestock production*	103 (Subak et al 1993)	6.2 (Kroeze et al 1999)

*Livestock is portion of anthropogenic sources.

Clemens & Ahlgrimm (2001) conclude that the most efficient manure management methods to minimize greenhouse gas emissions from collection and storage is to remove the manure quickly from housing areas and into an anaerobic biogas plant. Digested material should then be stored in a closed system. It is then suggested that if the final storage area could be cooled, further benefits would be realized.

Ammonia, hydrogen sulfide and carbon dioxide are the main gases of health concern related to manure in pig houses (Muehling 1969; Heber *et al.* 1997). Methane, nitrous oxide and carbon dioxide are gases associated with global warming that originate from manure. Aerobic processes can produce CO_2, NH_3, and N_2O, while anaerobic degradation can produce CO_2, NH_3, and CH_4.

Resources

AARNINK, A. J. A. & ELZING, A. (1998). Dynamic model for ammonia volatilization in housing with partially slatted floors, for fattening pigs. *Livestock Production Science* 53, 153-169.

AARNINK, A. J. A., KEEN, A., METZ, J. H. M., SPEELMAN, L. & VERSTEGAN, M. W. A. (1995). Ammonia emission patterns during the growing periods of hogs housed on partially slatted floors. *Journal of Agricultural Engineering Research* 62, 105-116.

AARNINK, A. J. A., KOETSIER, A. C. & VAN DEN BERG, A. J. (1993). Dunging and lying behavior of fattening pigs in relation to pen design and ammonia emission. In: *Fourth International Symposium of Livestock Environment* (Eds E. Collins & C. Boon), pp. 1176-1183.St. Joseph, MI: ASAE.

AARNINK, A. J. A., VAN DEN BERG, A. J., KEEN, A., HOCKSMA, P., & VERSTEGEN, M. W. A. (1996). Effect of slatted floor area on ammonia emission and on the excretory and lying behaviour of growing pigs. *Journal of Agricultural Engineering Research* 64, 299-310.

AARNINK, A. J. A. & WAGEMANS, M. J. M. (1997). Ammonia volatilization and dust concentration as affected by ventilation systems in houses for fattening pigs. *Transactions of the ASAE* 40(4), 1161-1170.

ALLEN, W. F. (1937). Olfactory and trigeminal conditioned reflexes in dogs. *American Journal of Physiology* 118, 532-540.

AL-MASHHADANI, E. H. & BECK, M. M. (1983). An SEM study of pulmonary ultrastructure in chickens subjected to various levels of atmospheric ammonia. *Poultry Science* 1715-1716.

ANDERSON, D. P., BEARD, G. W. & HANSON, R. P. (1964). The adverse effects of ammonia on chickens including resistance to infection with Newcastle disease virus. *Avian Diseases* 8, 369-379.

ANDERSON, G. A., SMITH, R. J., BUNDY, D. S. & HAMMOND, E. G. (1987). Model to predict gaseous contaminants in swine confinement buildings. *Journal of Agricultural Engineering Research* 37, 235-253.

ANDERSSON, M. (1995). *Cooling manure in manure culverts. A method of reducing ammonia emissions in pig building*. Institutionen för Jordbrukets Biosystem och Teknologi, Sveriges Lantbruksuniversitet, Lund, Sweden, 40 pp.

ANEJA, V. P., CHAUHAN, J. P. & WALKER, J. T. (2000). Characterization of atmospheric ammonia emissions from swine waste storage and treatment lagoons. *Journal of Geophysical Research – Atmospheres* 105, 11535-11545.

ANEJA, V. P., BUNTON, B., WALKER, J. T. & MALIK, B. P. (2001). Measurement and analysis of atmospheric ammonia emissions from anaerobic lagoons. *Atmospheric Environment* 35, 1949-1958.

ANEJA, V. P., MURRAY, G. & SOUTHERLAND, J. (1998). Atmospheric nitrogen compounds: emissions, transport, transformation, deposition, and assessment. *Environmental Manager*, April, 22-25.

ATSDR (2007). *Ammonia (NH_3) CAS 7664-41-7; UN 2672 (between 12% and 44% solution), UN 2073 (>44% solution), UN 1005 (anhydrous gas or >50% solution).* United States Department of Health and Human Services, Agency for Toxic Substances and Disease Registry.

AVERY, G. L., MERVA, G. E. & GERRISH, J. B. (1975). Hydrogen sulphide production in swine confinement units. *Transactions of the ASAE* 18, 149-151.

BÉLINE, F., MARTINEZ, J., CHADWICK, D., GUIZIOU, F. & COSTE, C. M. (1999). Factors affecting nitrogen transformations and related nitrous oxide emissions from aerobically treated piggery slurry. *Journal of Agricultural Engineering Research* 73, 235-243.

BICUDO, J. R., TENGMAN, C. L., JACOBSEN, L. D. & SULLIVAN, J. E. (2000). Odor, hydrogen sulfide and ammonia emissions from swine farms in Minnesota. In: *Conference Proceedings Odors and VOC Emissions 2000, Session 8*. Cincinnati, OH: Water Environment Federation.

BOUWMAN, A. F., LEE, D. S., ASMAN, W. A. H., DENTENER, F. J., VAN DER HOEK, K. W. & OLIVIER, J. G. J. (1997). A global high-resolution emission inventory for ammonia. *Global Biogeochemical Cycles* 11(4), 561-587.

BUIJSMAN, E., MAAS, H. M. & ASMAN, W. H. (1987). Anthropogenic NH_3 emissions in Europe. *Atmospheric Environment* 21(5), 1009-1022.

BURTON, C. H., SNEATH, R. W. & FARRENT, J. W. (1993). Emissions of nitrogen oxide gases during aerobic treatment of animal slurries. *Bioresource Technology* 45, 233-235.

BUSSE, F-W. (1993). Comparison measurements of the house climate in swine stables with and without repiratory diseases or cannibalism. In: *Livestock Environment* (Eds E. Collins & C. Boon), pp. 904-908. Fourth International Symposium, University or Warwick, Coventry, England, ASAE, St. Joseph, MI.

CAIN, W. S. (1974). Contribution of the trigeminal nerve to perceived odour magnitude. *Annals of the New York Academy of Science* 237, 28-34.

CAIN, W. S. (1976). Olfaction and the common chemical sense: some psychophysical contrasts. *Sensory Processes* 1, 57-67.

CAIN, W. S. (1988). Olfaction. In: *Stevens' Handbook of Experimental Psychology, I: Perception and Motivation*, Wiley, New York, pp. 409-459.

CARGILL, C. & SKIRROW, S. C. (1997). Air quality in pig housing facilities. *Pig Production Processes.* Post Graduate Foundation Veterinary Science. The University of Sydney, pp. 85-103.

CHENG, J., SHEARIN, T. E., PEET, M. M. & WILLITS, D. H. (2004). Utilization of treated swine wastewater for greenhouse tomato production. *Water Science and Technology* 50, 77-82.

CIGR (1992). *Climatisation of Animal Houses. (2nd Edn.).* Commission Internationale du Génie Rurale, Faculty of Agricultural Sciences, State University of Gent, Gent, Belgium.

CHAUHAN, J. P. (1999). *Characteristion of ammonia emissions from swine waste storage and treatment lagoons*. M.S. Thesis, North Carolina State University, Raleigh, NC. 27695-8208, p. 36.

CHRISTENSEN, K. & THORBEK, G. (1987). Methane excretion in growing pig. *British Journal of Nutrition* 57, 233-235.

CLEMENS, J. & AHLGRIMM, H. -J. (2001). Greenhouse gases from animal husbandry: mitigation options. *Nutrient Cycling in Agroecosystems* 60, 287-300.

COFFEY, R. D. & CROMWELL, G. L. (1994). Spray dried porcine plasma in diets for early weaned pigs housed either in an experimental or conventional nurser4y setting. *Journal of Animal Science* 72, 69.

CORRÉ, W. & OENEMA, O. (1998). Methane from animals and animal waste. In *Proceedings of the Workshop on Biogenic Emissions of Greenhouse Gases caused by Arable and Animal Agriculture – Measurement Technology and Emission Factors,* (Eds A. Freibauer & M. Kaltschmitt) pp. 86-93. University of Stuttgart, Stuttgart.

CRUTZEN, P., ASELMANN, I. & SEILER, W. (1986). *Methane production by domestic animals, wild ruminants, other herbivorous fauna, and humans*. Tellus 38B; 271-284.

CUMBY, T. R., MOSES, B. S. & NIGRO, E. (1995). Gases from livestock slurries: emission kinetics. In: *Proceedings of the Seventh International Symposium on*

Agricultural and Food Processing Wastes (ISAFPW95) (Ed C. C. Ross), Hyatt Regency Chicago, Chicago, Ills, ASAE, 230-240.

CURTIS, S. E. (1972). Air environment and animal performance. *Journal of Animal Science* 36, 628.

CURTIS, S. E., ANDERSON, C. R., SIMON, J., JENSEN, A. H., DAY, D. L. & KELLY, K. W. (1975). Effects of aerial ammonia, hydrogen sulfide and swine-house dust on rate of gain and respiratory tract structure in swine. *Journal of Animal Science* 41, 735-739.

DAY, D. L., HANSEN, E. L. & ANDERSON, S. (1965). Gases and odors in confinement swine buildings. *Transactions of the American Society of Engineering*, 8: 118-121.

DE LA HOZ, R. E., SCHLUETER D. P. & ROM W. N. (1996). Chronic lung disease secondary to ammonia inhalation injury: a report on three cases. *American Journal of Industrial Medicine* 29(2):209-14.

DEWES, T. (1996). Effect of pH, temperature, amount of litter and storage density on ammonia emissions from stable manure. *Journal of Agricultural Science* 127, 501-509.

DO BOER, S. & MORRISON, W.D. (1988). *The effects of the quality of the environment in livestock buildings on the productivity of swine and safety of humans. A literature review*. Department of Animal and Poultry Science, University of Guelph, Guelph, Ontario, Canada.

DÖHLER, H., SCHIEBL, K., SCHWAB, M. & KUHN, E. (1999). *Umweltverträgliche Gülleaufbereitung und –verwertung, Kuratorium für Technik und Bauwesen in der Landwirtschaft e.V. (KTBL),* Darmstadt, Arbeitspapier 272.

DONHAM, K. (1982). *Respiratory disease hazards of swine confinement workers. Termination Report for National Institute of Occupational Safety and Health Grant*, National Institute of Health, 1 Rol OH 00825.

DONHAM, K., HAGLIND, P., PETERSON, Y., RYLANDER, R. & BELIN, L. (1989). Environmental and health studies of farm workers in Swedish confinement buildings. *Journal of Industrial Medicine* 46, 31-37.

DONHAM, K. J., KNAPP, L. W., MONSON, R. & GUSTAFSON, K. (1982). Acute toxic exposure to gases from liquid manure. *Journal of Occupational Medicine* 24(2), 142-145.

DONHAM, K. J., RUBINO, M., THEDELL, T. D. & KAMMERMEYER, J. (1977). Potential health hazards to agricultural workers in swine confinement buildings. *Journal of Occupational Medicine* 19, 383-387.

DOTY, R. L. (1995). Intranasal trigeminal chemoreception. In: *Handbook of Olfaction and Gustation* (Ed R. L. Doty), Marcel Dekker, New York, pp. 821-833.

DRUMMOND, J. G., CURTIS, S. E., SIMON, J. & NORTON, H. W. (1980). Effects of aerial ammonia on growth and health of young pigs. *Journal of Animal Science* 50(6), 1085-1091.

EKKEL, E. D., VAN DOOM, C. E., HESSING, M. J. & TIELEN, M. J. (1995). The specific-stress-free housing system has positive effects on productivity, health, and welfare of pigs. *Journal of Animal Science* 73, 1544-1551.

ELZING, A. & AARNINK, A.J.A. (1996). Invloed van zuutegraad van varkensurine op de ammoniakemissie. *Report 96-02, IMAG-DLO Wageningen*, pp. 27.

ELZING, A. & KROODSMA, W. (1993). Relative tussen ammoniakemissie en stikstofconcentratie in urine van melkvee. *Report 93-3, IMAG-DLO Wegeningen*, pp. 27.

ELZING, A. & SWIERSTRA, D. (1993). Ammonia emission measurements in a model system of a pig house. In: *Proceedings Congress on Nitrogen Flow in Pig Production and Environmental Consequences* (Eds M. W. A. Verstegen, L. A. den Hartog, G. J. M. van Kampen & J. H. M. Metz), Wageningen: Pudoc, 280-285.

EPA (2004). *National emission inventory – ammonia emissions from animal husbandry.* US Environmental Protection Agency, Washington DC, January 30, 2004.

FANGMEIER, A., HADWIGER-FANGMEIER, A., VAN DER EERDEN, L. & JÄGER, H. -J. (1994). Effects of atmospheric ammonia on vegetation – a review. *Environmental Pollution* 86, 43-82.

FARBMAN, A. I. (1992). *Cell Biology of Olfaction*. Cambridge Univ. Press, Cambridge, p. 71.

FERGUSON, W. S., KOCH, W. C., WEBSTER, L. B. & GOULD, J. R. (1977). Human physiological response and adaptation to ammonia. *Journal of Occupational Medicine* 19, 319-326.

FINLAYSON-PITTS, B. J. & PITTS, J. N. (1986). *Atmospheric Chemistry*. Wiley-Interscience, New York, p. 1698.

FRENEY, J. R., SIMPSON, J. R. & DENMEAD, O. T. (1983). Volatilization of ammonia. *Developments in Plant and Soil Sciences* 9, 1-32.

GENFA, Z., UEHARA, T., DASGUPTA, P. K., CLARKE, A. D. & WINIWARTER, W. (1998). Measurement of diffusive flux of ammonia from water. *Analytical Chemistry* 70(17), 3656-3666.

GROOT KOERKAMP, P. W. G., METZ, J. H. M., UENK, G. H., PHILLIPS, V. R., HOLDEN, M. R., SNEATH, R. W., SHORT, J. L., WHITE, R. P., HARTUNG, J., SEEDORF, J., SCHROEDER, M., LINKERT, K. H., TAKAI, H., PEDERSEN, S., JOHNSON, J. O. & WATHES, C. M. (1996). Concentration and emissions of ammonia in livestock buildings in Northern Europe. *Journal of Agricultural Engineering Research* 70, 79-95.

GRUB, W., FOERSTER, E. P. & TRIBBLE, L. F. (1974). Swine building air contaminant control with pit ventilation. American Society of Agricultural Engineers, St. Joseph, Michigan, *Mimeographed paper ASAE* 74-4532.

GRUBER, L. & STEINWIDDER, A. (1996). Einfluβ der Fütterung auf die Sticstoff- und Phophorausscheidung landwirtschaftlicher Nutziere, Modellkalkuklation auf Basis einer Übersicht. *Die Bodenkultur* 47: 255-277.

GUSTIN, P., URBAIN, B., PROUVOST, J. F. & ANSAY, M. (1994). Effects of atmospheric ammonia on pulmonary hemodynamics and vascular-permeability in pigs – interaction with endotoxins. *Toxicology and Applied Pharmacology* 125(4), 17-26.

HAHNE, J., HESSE, D. & VORLOP, K. D. (1999). Spurengasemissionen aus der schweinehaltung. *Landtechnik* 54, 180-181.

HAMILTON, T. D. C., ROE, J. M. & WEBSTER, A. J. F. (1996). Synergistic role of gaseous ammonia in etiology of *Pasteurella multocida*-induced atrophic rhinitis in swine. *Journal of Clinical Microbiology* 34, 2185-2190.

HAMMOND, E. G. & SMITH, R. J. (1981). Survey of some molecularly dispersed odorous constituents in swinehouse air. *Iowa State Journal of Research* 55(4), 393-399.

HARTUNG, E. & BÜSCHER, W. (1995). Ammoniakfreisetzung. Einfluβfaktoren auf den Stoffübergang aus Flüssigmist. *Landtechnik* 50(3), 164-165.

HEBER, A. J., DUGGIRALA, R. K., NI, J. Q., SPENCE, M. L., HAYMORE, B. L., ADAMCHUC, V. I., BUNDY, D. S., SUTTON, A. L., KELLY, D. T. & KEENER, K. M. (1997). Manure treatment to reduce gas emissions from large swine houses. In: International *Symposium on Ammonia and Odour Control from Animal Production Facilities* (Eds J. A. M. Voermans & G. J. Monteny), Vol. II, pp. 449-458. Rosmalen, The Netherlands: NVTL.

HESSE, D. (1994). Comparison of different old and new fattening pig husbandrys with focus on environment and animal welfare. In: *Proceedings of the XII World Congress on Agricultural Engineering, Milano, Aug. 29-Sept 1*, pp. 559-566.

HOEKSMA, P., VERDOES, N., OOSTHOEK, J. & VOERMANS, J. A. M. (1992). Reduction of ammonia volatilization from pig houses using aerated slurry as recirculation liquid. *Livestock Production Science* 31, 121-132.

HÖRNIG, G., TÜRK, M. & WANKU, U. (1999). Slurry covers to reduce ammonia emission and odour nuisance. *Journal of Agricultural Engineering Research* 73, 151-157.

HOUGHTON, J. T., JENKINS, G. J. & EPHRAUMS, J. J., (ED.) (1990). IPCC-Intergovernmental Panel on Climate Change, 1990: *Climate Change: The IPCC Scientific Assessment*. Cambridge University Press, Cambridge, UK.

HOUGHTON, J. T., MEIRA FILHO, L. G., CALLANDER, B. A., HARRIS, N., KATTENBERG, A. & MASKELL, K. (1996). Intergovernmental Panel on Climate Change 1995. *The Science of Climate Change*. Cambridge University Press, Cambridge, UK.

HUSTED, S., JENSEN, L. S. & JØRGENSEN, S. S. (1991). Reducing ammonia loss from cattle slurry by the use of acidifying additives: the role of the buffer system. *Journal of the Science of Food and Agriculture* 57, 335-349.

IPCC (2001). *Climate change 2001: the scientific basis. Contribution of Working Group I to the Third Assessment Report on the Intergovernmental Panel on Climate Change.* Cambridge University Press, Cambridge, UK.

JACOBSEN, L. D., MARCINEK-PASZEK, D., NICOLAI, R. E., SCHMIDT, D. R., HETCHLER, B. & ZHU, T. (1999). Odor and gas emissions from animal manure storage units and buildings. In: *ASAE/CSAE-SCGR Annual International Meeting, Paper No. 994004*, 11 pp. St. Joseph, MI: ASAE.

JERICHO, K.W.F. (1968). Pathogenesis of pneumonia in pigs. *Veterinarian Record* 82: 507-517.

JONES, J. B., BURGESS, L. R., WEBSTER, A. J. F. & WATHES, C. M. (1996). Behavioural responses of pigs to atmospheric ammonia in a chronic choice test. *Animal Science* 63, 437-445.

JONES, J. B., WATHES, C. M. & WEBSTER, A. J. F. (1998). Operant responses of pigs to atmospheric ammonia. *Applied Animal Behaviour Science* 58, 35-47.

JUNGBLUTH, T. & BÜSCHER, W. (1996). Reduction of ammonia emissions from piggeries. In: *1996 ASAE Annual International Meeting, Pheonix, Arizona, 14-18 July. Paper No. 964091. ASAE*, 2950 Niles Rd., St. Joseph, MI 49085-9659 USA, pp. 1-16.

KAMIN, H., BARBER, J. C., BROWN, S. I., DELWICHE, C. C., GROSJEAN, D., HALES, J. M., KNAPP, J. L. W., LEMON, E. R., MARTENS, C. S., NIDEN, A. H., WILSON, R. P. & FRAZIER, J. A. (1979). *Ammonia*. University Park Press, Baltimore.

KATZ, S. H. & TALBERT, E. J. (1930). Intensities of odors and irritating effects of warning agents for inflammable and poisonous gases. *US Dept. Commerce, Bureau of Mines. Technical Paper 480*, Washington, DC.

KHALIL, M. A. K. & RASMUSSEN, R. A. (1992). The global sources of nitrous oxide. *Journal of Geophysic Research* 97, 14,651-14,660.

KIRCHGESSNER, M., ROTH, F. X. & WINDISCH, W. (1993). Verminderung der Stickstoff- und Methanausscheidung von Schwein und Rind durch die Fütterung. *Tierernährung* 21: 89-120.

KIRCHGESSNER, M., WINDISCH, W. & ROTH, F. X. (1994). The efficiency of nitrogen conversion in animal nutrition. *Nova Acta Leopoldina* NF 70, 393-412.

KOVÁCS, F., NAGY, A. & SALLAI, J. (1967). Effect of environmental factors on the health and productivity of pigs. II. Dust, micro-organisms and chemical pollution of the air in piggeries. Magy Allatorv. Lap., 22: 496 (Abstract No. 4371 In: *Veterinarian Bulletin*, 1968. 38: 727.)

KRAPFENBAUER, A. & WRIESSNIG, K. (1995). Anthropogenic environmental-pollution – the share of agriculture. *Bodenkultur* 46, 269-283.

KREIS, D. (1978). Control of animal production odors: the state of the art. *U.S. Environmental Protection Agency, Environmental Protection Technology Series EPA-600*, 2-78-083.

KROEZE, C. (1998). N_2O from animal waste. In: *Proceedings of the Workshop on Biogenic Emissions of Greenhouse Gases caused by Arable and Animal Agriculture – Measurement Technology and Emission Factors* (Eds A. Freibauer & M. Kaltschmitt), pp. 119-128. University of Stuttgart, Stuttgart.

KROEZE, C., MOSIER, A. & BOUWMAN, L. (1999). Closing the global N_2O budget: a retrospective analysis 1500-1994. *Global Biochemical Cycles* 13(1), 1-8.

LEBEDA, D. L., DAY, D. L. & HAYAKAWA, T. (1964). Air pollutants in swine buildings with fluid waste handling. American Society of Agricultural Engineers, Saint Joseph, Michigan, *Mimeographed paper ASAE* 64-940.

LEE, C., GILES, L. R., BRYDEN, W. L., DOWNING J. L., OWENS, P. C., KIRBY, A. C. & WYNN, P. C. (2005). Performnance and endocrine responses of group housed weaner pigs exposed to the air quality of a commercial environment. *Livestock Production Science* 93, 255-262.

LIAO, C. -M. (1996). Model to predict human and pig exposure to gaseous contaminants emitted from stored manure. *Applied Mathematical Modelling* 20, 731-740.

LIU, Q., BUNDY, D. S. & HOFF, S. F. (1993). Utilizing ammonia concentrations as an odour threshold indicator for swine facilities. In: *Livestock Environment* (Eds E. Collins & C. Boon), Fourth International Symposium. Coventry, University of Warwick, England, ASAE, St. Joseph, MI.

MAFF (1987). *Codes of Recommendations for the Welfare of Livestock: Domestic Fowls*. Ministry of Agriculture, Fisheries and Food, MAFF Publications, London.
MALAYER, J. R., BRANDT, K. E., GREEN, M. L., KELLY, D. T., SUTTON, A. L. & DIEKMAN, M. A. (1988). Influence of manure gases on the onset of puberty of replacement gilts. *Animal Production* 46, 277-282.
MALAYER, J. R., KELLY, D. T., DIEKMAN, M. A., BRANDT, K. E., SUTTON, A. L., LONG, G. G. & JONES, D. D. (1987). Influence of manure gases on puberty in gilts. *Journal of Animal Science* 64, 1476-1483.
MARTINEZ, J., GUIZIOU, F., PEU, P. & GUEUTIER, V. (2003). Influence of treatment techniques for pig slurry on methane emissions during subsequent storage. *Biosystems Engineering* 85, 347-354.
MCALLISTER, J. S. V. & MCQUITTY, J. B. (1965). Release of gas from slurry.*Records of Agricultural Research, Northern Ireland* 14(2), 73-78.
MCCUBBIN, D. R., APELBERG, B. J., ROE, S. & DIVITA, JR., F. (2002). Livestock ammonia management and particulate-related health benefits. *Environmental Science and Technology* 36(6), 1,141-1,146.
MINER, J. R. & HAZEN, T. E. (1969). Ammonia and amines: components of swine building odor. *Transactions of the ASAE* 12, 772-774.
MONTENY, G. J., GROENESTEIN, C. M. & HILHORST, M. A. (2001). Interactions and coupling between emissions of methane and nitrous oxide from animal husbandry. *Nutrient Cycling in Agroecosystems* 60, 123-132.
MOSIER, A. & KROEZE, C. (1998). A new approach to estimate emissions of nitrous oxide from agriculture and its implications to the global N_2O budget. *IGBP Global Change Newsletter. No. 34*, June, pp. 8-13.
MUCK, R. E., GUEST, R. W. & RICHARDS, B. K. (1984). Effects of manure storage design on nitrogen conservation. *Agricultural Engineering Research* 66, 239-250.
MUCK, R. E. & RICHARDS, B. K. (1983). Losses of manorial nitrogen in free-stall barns. *Agricultural Wastes* 7, 65-79.
MUCK, R. E. & STEENHUIS, T. S. (1981). Nitrogen losses in free stall dairy barns. In: *Proceedings 4th International Symposium on Livestock Waste: A Renewable Resource. ASAE, St. Joseph, MI*, pp. 406-409.
MUCK, R. E. & STEENHUIS, T. S. (1982). Nitrogen losses from manure storage. *Agricultural Wastes* 4, 41-54.
MUEHLING, A. J. (1969). Gases and odors from stored swine waste. *Journal of Animal Science* 30, 526-531.
NÄSI, M. (1993). Nitrogen utilization responses of growing pigs to supplemental lysine in barley—barley protein diet. In: *Proceedings of Congress on Nitrogen Flow in Pig Production and Environmental Consequences. Pudoc Wageningen* (Eds M. W. A. Verstegen, L. A. den Hartog, G. J. M. van Kempen & J. H. M. Metz, J. H. M.), pp. 224-229.
NI, J. Q. (1998). *Emission of carbon dioxide and ammonia from mechanically ventilated pig house.* PhD Thesis. Catholic University of Leuven, Belgium, 227 pp.
NI, J. (1999). Mechanistic models of ammonia release from liquid manure: a review. *Journal of Agricultural Engineering Research* 72, 1-17.

NI, J. Q., HEBER, A. J., DIEHL, C. A. & LIM, T. T. (2000a). Ammonia, hydrogen sulphide and carbon dioxide release from pig manure in under-floor deep pits. *Journal of Agricultural Engineering Research* 77(1), 53-66.

NI, J. Q., HEBER, A. J., DIEHL, C. A., LIM, T. T., DUGGIRALA, R. K. & HAYMORE, B. L. (2000b). Burst releases of hydrogen sulfide in mechanically ventilated swine buildings. In: *Conference Prodeedings Odors and VOC Emissions 2000, Session 8-C.* Cincinnati, OH, Water Environment Federation.

NI, J. Q., HEBER, A. J., LIM, T. T., DIEHL, C. A., DUGGIRALA, R. K., HAYMORE, B. L. & SUTTON, A. L. (2000c). Ammonia emissions from a large mechanically-ventilated swine building during warm weather. *Journal of Environmental Quality* 29, 751-758.

NI, J. Q., HEBER, A. J., LIM, T. T., DIEHL, C. A., DUGGIRALA, R. & HAYMORE, B. L. (2002). Hydrogen sulfide emission from two large pig-finishing buildings with long-term high-frequency measurements. *Journal of Agricultural Science* 138, 227-236.

NI, J. Q., VINKIER, C., COENEGRACHTS, J. & HENDRIKS, J. (1999). Effect of manure on ammonia emission from a fattening pig house with partly slatted floor. *Livestock Production Science* 59; 25-31.

NICHOLSON, R. J., WEBB, A. & MOORE, A. (2002). A review of the environmental effects of different livestock manure storage systems, and a suggested procedure for assigning environmental ratings. *Biosystems Engineering* 81, 363-377.

NIOSH (2005). *NIOSH Document: Pocket Guide to Chemical Hazards: Ammonia (2005-149).* US National Institute for Occupational Safety and Health.

OLIVIER, J. G. J., BOUWMAN, A. F., VAN DER HOEK, K. W. & BERDOWSKI, J. J. M. (1998). Global air emission inventories for anthropogenic sources of NO_X, NH_3 and N_2O in 1990. In: *Nitrogen, the Confer-N-s. First International Nitrogen Conference 1990* (Eds K. W. Van der Hoek *et al.*). pp. 135-148. Elsevier, Amsterdam.

OLESEN, J. E. & SOMMER, S. G. (1993). Modelling effects of wind speed and surface cover on ammonia volatilization from stored pig slurry. *Atmospheric Environment. Part A. General Topics* 27(16), 2567-2574.

OSHA (1999). *VI. Health effects discussion and determination of final PEL. In: OSHA Preambles – Air Contaminants (29 CFR 1910.1000).* Occupational Safety and Health Administration, U.S. Department of Labor.

OYETUNDE, O. O. F., THOMSON, R. G. & CARLSON, H. C. (1978). Aerosol exposure of ammnia, dust and *Escherichia coli* in broiler chickens. *Canadian Veterinary Journal* 19, 187-193.

PAIN, B. F., PHILLIPS, V. R., CLARKSON, C. R. & KLARENBEEK, J. V. (1989). Loss of nitrogen through ammonia volatilization during and following the application of pig or cattle slurry to grassland. *Journal of the Science of Food and Agriculture* 47(1), 1-12.

PAIN, B. F., VAN DER WEERDEN, T. J., CHAMBERS, B. J., PHILLIPS, V. R. & JARVIS, S. C., (1998). A new inventory for ammonia emissions from UK agriculture. *Atmospheric Environment* 32(3), 309-314.

PATNI, N. K. & CLARKE, S. P. (1991). Transient hazardous conditions in animal buildings due to manure gas released during slurry mixing. *Applied Engineering in Agriculture*, 7, 478-484.

PATTERSON, P. H. & LORENZ, E. S. (1996). Manure nutrient production from commercial White Leghorn hens. *Journal of Applied Poultry Research* 5, 260-268.
PATTERSON, P. H. & LORENZ, E. S. (1997). Nutrients on manure from commercial White Leghorn hens. *Journal of Applied Poultry Research* 6, 260-268.
PATTERSON, P. H., LORENZ, E. S., WEAVER JR., W. D. & SCHWARTZ, J. H. (1998). Litter production and nutrients from commercial broiler chickens. *Journal of Applied Poultry Research* 7, 247-252.
PELCHEN, A., PETERS, K. J. & HOLTER, J. B. (1998). Prediction of methane emissions from lactating dairy cows. Arch. *Tierz. Dummerstorf* 41(6), 553-563.
PFEIFFER, A. & HENKEL, H. (1991). The effect of different dietary protein levels on water intake and water excretion of growing pigs. In: *Proceedings of the 5th International Symposium on Digestive Physiology iin Pigs. EAAP-publication no. 54, Pudoc Wageningen* (Eds M. W. A. Verstegen, J. Huisman & L. A. den Hartog, L. A.), pp. 126-131.
RANDALL, J. M., ARMSBY, A. W. & SHARP, J. R. (1983). Cooling gradients across pens in a finishing piggery. *Journal of Agricultural Engineering Research* 28, 247-257.
ROBERTSON, A. M. & GALBRAITH, H. (1971). Effects of ventilation on the gas concentration in a part-slatted piggery. *Farm Buildings Research and Development Studies* 1, 17-28.
ROBERTSON, J. F., WILSON, D. & SMITH, W. J. (1990). Atrophic rhinitis: The influence of the aerial environment. *Animal Production* 50, 173-182.
ROELOFFS, J. G. L. & HOUDIJK, A. L. M. (1991). Ecological effects of ammonia. In: *Ammonia and Odour Emissions from Livestock Production* (Eds V. C. Nielson, B. F. Pain & J. Hartung), pp. 10-16. Elvieser Applied Science, London.
ROLLER, W. L. (1965). Need for study of effects on air contaminants on equipment and animal performance. *Transactions of the ASAE* 8(2): 353-357.
SEINFELD, J. H. & PANDIS, S. N. (1998). Atmospheric Chemistry and Physics. Wiley Internscience, New York, p. 1326.
SKARP, S. -U. (1975). Manure gases and air currents in livestock housing. In: *Proceedings, The 3rd International Symposium on Livestock Waste. ASAE Publication PROC-275*, pp. 363-365.
SLANINA, S. (1994). Forest dieback and ammonia – a typical Dutch problem. *Chemistry International* 16, 2-3.
SMITH, J. H., WATHES, C. M. & BALDWIN, B. A. (1996). The preference of pigs for fresh air over ammoniated air. *Applied Animal Behaviour Science* 49, 417-424.
SMITH, R. P., COOPER, R. C., ENGEN, T., HENDRICKSON, E. R., KATZ, M., MILBY, T. H., MUDD, J. B., ROSSANO, A. T. & REMOND JR., J. (1979). *Hydrogen Sulfide*. Baltimore: University Park Press.
SMITS, M. C. J., VALK, H., ELZING, A. & KEEN, A. (1995). Effects of protein nutrition on ammonia emission from cubicle house for dairy cattle. *Livestock Production Science* 44, 147-156.
SOMMER, S. G. (1997). Ammonia volatilization from farm tanks containing anaerobically digested animal slurry. *Atmospheric Environment* 31, 863-868.

SOMMER, S. G., CHRISTIANSEN, B. T., NIELSON, N. E. & SCHJØRRING, J. K. (1993). Ammonia volatilization during storage of cattle and pig slurry: effect of surface cover. *Journal of Agriculture Science* 121, 63-71.
SOMMER, S. G., OLESEN, J. E. & CHRISTIANSEN, B. T. (1991). Effects of temperature, wind speed, and air humidity on ammonia volatilization from surface applied cattle slurry. *Journal of Agricultural Environment* 34, 91-100.
SOMMER, S. G., PETERSEN, S. O. & MØLLER, H. B. (2004). Algorithms for calculating methane and nitrous oxide emissions from manure management. *Nutrient Cycling in Agroecosystems* 69, 143-154.
SRINATH, E. G. & LOEHR, R. C. (1974). Ammonia desorption by diffused aeration. *Journal of the Water Pollution Control Federation* 46, 1939-1957.
STEED JR., J. & HASHIMOTO, A. G. (1994). Methane emissions from typical manure management systems. *Bioresource Technology* 50, 123-130.
STEVENS, R. J., LAUGHLIN, R. J. & FROST, J. P. (1989). Effect of acidification with sulphuric acid on the volatilization of ammonia from cow and pig slurries. *Journal of Agricultural Science* 113, 389-395.
STOMBAUGH, D. P., TEAGUE, H. S. & ROLLER, W. L. (1969). Effects of atmospheric ammonia on the pig. *Journal of Animal Science* 28, 844-847.
SUBAK, S., RASKIN, P. & VON HIPPEL, D. (1993). National greenhouse gas accounts: current anthropogenic sources and sinks. *Climate Change* 25: 15-58.
SVENNSON, L. (1991). Ammonia losses from slurry storages—review of research and measures for reduction. In: *Environmental Challenges and Solutions in Agricultural Engineering*, pp. 64-72. Agricultural University of Norway, As.
TUCKER, D. (1963a). Physical variables in the olfactory stimulation process. *Journal of General Physiology* 46, 453-489.
TUCKER, D. (1963b). Olfactory, vomeronasal and trigeminal receptor responses to odourants. In: *Olfaction and Taste I* (Ed Y. Zotterman). Pergamon, NY.
TUCKER, D. (1971). Nonolfactory responses from the nasal cavity: Jacobson's organ and the trigeminal system. In: *Olfaction* (Ed L. M..Beidler), Springer-Verlag, Berlin, pp. 151-181.
URBAIN, B., GUSTIN, P., PROUVOST, J. & ANSAY, M. (1994). Quantitative assessment of aerial ammonia toxicity to the nasal mucosa by use of the nasal lavage method in pigs. *American Journal of Veterinary Research* 55, 1335-1340.
VAN BREEMAN, N. BURROUHG, P. A., VELTHORST, E. J., VAN DOBBEN, H. F., DE WIT, T., RIDDER, T. B. & REIJNDERS, H. F. R. (1982). Soil acidification from atmospheric ammonium sulfate in forest canopy throughfall. *Nature* 299, 548-550.
VAN DER PEET-SCHWERING, C. M. C., VERDOES, N., VOERMANS, M. P. & BEELEN, G. M. (1996). Effect van voiding en huisvesting op de ammoniakemissie uit vleesvarkensstallen. *Research report P 1,145, Pigs Research Station,* Rosmalen pp. 40.
VISEK, W. J. (1968). Some aspects of ammonia toxicity in animal cells. *Journal of Dairy Science* 51, 1-11.
VOERMANS, J. A. M. & HENDRIKS, J. G. L. (1995). Pit or roof ventilation for growing finishing pigs. Research Institute for Pig Husbandry, Rosmalen, The Netherlands. *Proefverslag nummer* P, 4-9.

VON HOY, S. T., MÜLLER, K. & WILLIG, R. (1994). Zur ammoniak-konsentration bei zwei systemen der mastschweine-haltung auf tiefstrue mit bioaktivator im vergleich zur vollspaltenboden haltung. *Tierärztl Umschau* 49, 414-420.

WARNECK, P. (1988). *Chemistry of the Natural Atmosphere*. Academic Press, New York, pp. 426-441.

WATHES, C. M., JONES, C. D. R. & WEBSTER, A. J. F. (1983). Ventilation, air hygiene and animal health. *The Veterinary Record*, 113: 554-555.

WELTY, J. R., WICKS, C. E. & WILSON, R. E. (1984). *Fundamentals of Momentum, Heat, and Mass Transfer. 3 edn.* New York: Wiley.

WILLIAMS, A. G. & NIGRO, E. (1997). Covering slurry stores and effects on emission of ammonia and methane. In: *Proceedings of the International Symposium on 'Ammonia and Odour Control from Animal Production Facilities Vol. II*, pp. 421-428. Vinlkeloord, Netherlands, 6-10 October, 1997.

WULF, S., JÄGER, P. & DÖHLER, H. (2006). Balancing of greenhouse gas emissions and economic efficiency for biogas-production through anaerobic co-fermentation of slurry with organic waste. *Agriculture Ecosystems and Environment* 112, 178-185

ZHANG, R. (1992). *Degradation of swine manure and a computer model for predicting the desorption rate of ammonia from an under-floor pit*. PhD Thesis. University of Illinois at Urbana-Champaign, 131 pp.

ZHANG, R. H., DAY, D. L., CHRISTIANSON, L. L. & JEPSON, W. P. (1994). A computer model for predicting ammonia release rates from swine manure pits. *Journal of Agricultural Engineering Research* 58, 223-229.

ZHAO, D. W. & WANG, A. P. (1994). Estimation of anthropogenic ammonia emissions in Asia. *Atmospheric Environment* 28(28), 689-694.

ZHU, J., JACOBSEN, L. D., SCHMIDT, D. R. & NICOLAI, R. E. (2000). Daylong variations in odor and gas emissions from animal facilities. *Applied Engineering in Agriculture* 16, 153-158.

Chapter 7. Evaluation of Economic Feasibility and Animal Performance for a Novel Manure Collection and Anaerobic Digestion System at a Commercial Swine Finisher Enterprise

SUMMARY

Anaerobic digestion of manure can provide many benefits at the farm-level. The following case-study at a commercial swine finishing operation near Danville, Pennsylvania, USA was conducted to evaluate the operational and economic feasibility of a novel manure management system in conjunction with an anaerobic digester. Investment capital for the system was provided in part by the producer and by public and private grants. The system utilized under-floor manure storage pits to collect manure for delivery to a digester, and then stored post-digested manure (digestate) in under-floor storage within the same swine houses. Positioning of manure collection pits under swine dunging areas in two large-pen 2200-head buildings allowed for the collection of 75% of total manure volume, which was moved to the digester. Digester-produced biogas content was approximately 28% carbon dioxide and 72% methane. No additional post-digestion manure storage construction was necessary at the farm. The cost-savings of electricity produced from combustion of biogas (monthly value of US $477.10) was nearly equal to the producer's debt service for capital investment required for the construction of the manure handling and digester system (monthly payment of US $478.54). Debt service did not include grant funds. Monitoring of air quality indicators both before and after the introduction of digestate to under-floor manure storage pits in swine housing resulted in no observations of hydrogen sulfide (H_2S) or methane (CH_4) concentrations above critical safety levels. No recorded concentrations of oxygen (O_2) were below critical entry guideline levels. Hourly mean ammonia (NH_3) concentrations at pig level (0.15 m

above the floor) before digestate was present in the buildings were higher ($P<0.05$) compared to when digestate was present (24 ± 2.8 ppm vs. 17 ± 1.0 ppm). During steady-state digester operation the minimum ventilation system for the swine buildings was changed from manure pit ventilation to an end-wall fan on a timer. Hourly mean NH_3 concentrations at pig level were higher ($P<0.05$) after fan removal (37 ± 0.9) than when pit fans were present (17 ± 1.0 ppm). Swine group average daily gain, feed-to-gain ratio and culls and mortalities information from the case farm were compared to that of two other farms. Average daily gain of pigs on the case farm was lower ($P < 0.05$) than that of another farm receiving feeder pigs from the same sow units. Feed efficiency and a combination of culls and mortalities were statistically similar among three farms receiving pigs from the same sow units. We conclude that the novel manure collection system used on this farm can eliminate the need of a post-digester storage facility and reduce the cost for electricity for a commercial swine enterprise. Electric cost-savings made the combined digestion and manure collection system at this location were more affordable than that of a conventional digestion system. External funding and low interest financing were necessary in order for finance payments to be offset by electric cost-savings. Air quality measures did not indicate that the introduction of digestate into under-floor manure pits caused degradations of air quality at pig level. Because of the variations in management no clear effects could be determined from this manure treatment system on the growth performance of pigs in these buildings.

INTRODUCTION

Anaerobic digestion of manure produces biogas that is rich in methane and can be combusted in an engine-generator system to produce electricity. The electricity can be used on the farm to offset electricity purchase. Widespread adoption of anaerobic digestion technology has not occurred because of high capital investment and nominal economic return (Hill *et al.* 1985; Safley & Westerman 1994; Braber 1995). Traditionally there has been minimal incentive for livestock producers to seek alternative energy sources due to historically affordable fossil fuels and electricity. The competitiveness of biogas with other fuels used for heat or combined heat and power (CHP) are limited (Lantz *et al.* 2007). Rising social costs associated with environmental impacts, energy use, and manure odor generation make manure digestion attractive. Many believe that anaerobic digestion will become more affordable as advances in technology, lower capital investment requirements, and rising costs of non-renewable fuels make biogas systems more economically reasonable (Wiese & Haeck 2006).

A traditional component of constructing anaerobic manure digestion systems is that of manure storage facilities for post-digested manure. If construction of digestate storage could be avoided, the implementation of farm-level digestion may be more affordable. One method to avoid additional cost of digestate storage in a standard commercial swine housing unit with under-barn storage would be to segregate the standard under-floor manure storage volume into compartments. Pre- and post-digested manure volumes and the majority of manure from dunging areas would be stored in specified compartments. Separate pen space dedicated to lying and dunging arise in large-pen settings because pigs rest near pen perimeters (Grandin 1980) and dung in open spaces (Fritschen 1975) away from resting areas (Stolba & Wood-Gush 1989).

Storing digested manure under the swine living facility could eliminate the need for a separate post-digester facility, but it may increase the concentration of dangerous gases in the swine living area. Methane (CH_4) is explosive at concentrations between 5-15% (NIOSH 1990). For humans, the Recommended Exposure Limit for gaseous ammonia (NH_3) is 25 ppm (NIOSH 2005) and the US Occupational Safety and Health Administration's Permissible Exposure Limit for gaseous NH_3 is 50 ppm (OSHA 1999). The 10-minute recommended exposure limit for hydrogen sulfide (H_2S) is 10 ppm. The minimum oxygen concentration level for safe human entry is 19.5% (NIOSH 1990). Monitoring of indoor air quality and pig growth efficiencies for indication of negative impacts on growth or mortality in such housing is warranted.

It is not clear whether an under-floor manure storage system designed to segregate raw manure and digested manure will provide sufficient amounts of raw manure for the practical operation of an anaerobic digester. Nor is it clear how this unique manure handling system may affect pig health and performance.

Therefore, the objectives of this study were to:

(1) Quantify the proportion of manure deposited into manure collection pits located under observed dunging areas in a commercial large-pen swine finishing operation.

(2) Evaluate manure constituents before and after digestion.

(3) Quantify manure loading, biogas production and quality, electricity production and engine run-time for a two-year period of steady-state operation of an anaerobic digester.

(4) Evaluate the economic viability of the digestion system based on producer capital costs and electric cost-savings.

(5) Characterize concentrations of CH_4, NH_3, H_2S, and O_2 in swine living space of the swine facility before and after the introduction of digestate to under-floor manure storage space, as well as after digestate introduction when manure pit ventilation was operational and when pit ventilation was not operational.

(6) Evaluate growth performance and mortality and culling with digestate introduction to manure storage located beneath swine living areas, in comparison to pigs at other barns where no manure treatment occurred.

MATERIALS AND METHODS

Dunging Pattern and Manure Pit Depth Analysis

The under-floor storage pits in each of two buildings were modified to advantageously collect manure deposited in dunging areas found in the central location of the eight large pens of the buildings. This was based on previous observations by the building designer that pigs tend to rest along the perimeter and dung centrally in the large pens of similarly designed buildings (Figure 7.1). Each building housed 2200 finishing hogs in four large pens equipped with self-sorting technology on totally slatted floors. The pens were located in two rooms (two large pens per room) within each building (1100 pigs/room, 550 pigs/pen). An interior wall located directly below the centrally located roof peak divided each rectangular building into the two rooms along the longer building axis. The rooms were equal in size and mirrored one another with location of feeders, waterers, scales, and penning; the only exceptions were worker walkways, located along an end-wall. The walkways connected the building's office entrance to the room furthest from the office. The buildings were located adjacent to one another on a farm near Danville, Pennsylvania, USA, and were simultaneously constructed in 2002 by Schick Enterprises (Kutztown, Pennsylvania, USA) from identically dimensioned mirrored designs. The rooms were tunnel-ventilated with static-pressure controlled air inlet curtains on one end-wall and a bank of fans located in the opposite end-wall. Side walls and the central interior wall were equipped with emergency drop curtains that provided natural ventilation during electrical outages. Each building was 85.3 m (280 ft) long and 24.4 m (80 ft) wide. Five manure pit fans, when operational, were spaced evenly along each long wall of the buildings and exhausted air pulled from above the slatted floors through the air space above manure.

Figure 7.1. Photo showing swine lying and dunging pattern in large, totally slatted pens. Pigs lie along the perimeter and dung in the central area of the pens. Because of the rectangular pen shape the dunging areas were long and narrow.

Five under-floor manure pits ran the complete length of each building and were all 1.83 m (6 ft) deep. Across its width each building contained eight commercially sized concrete slats, each 3.05 m (10 ft) wide. Therefore each 12.2 m (40 ft) wide room contained four slats. In the room one slat was located adjacent to both the exterior and interior walls, each above separate manure pits. The two centrally located slats were located above a 6.1 m (20 ft) wide manure pit. The slat along the interior wall was located above a manure pit that was shared with the slat located along the interior wall in the adjacent room; this manure pit was continuous under the central interior wall and under slats in both rooms. The interior wall that divided the building into two was constructed on top of these slats. Thus there were five manure pits in each building; two of the pits were located along the long exterior walls and were each 3.05 m (10 ft) wide, two of the pits were located centrally in each room and were each 6.1 m (20 ft) wide, and the fifth pit was located under the central interior wall and was 6.1 m (20 ft) wide with half of its width located under the floor in either room (Figure 7.2).

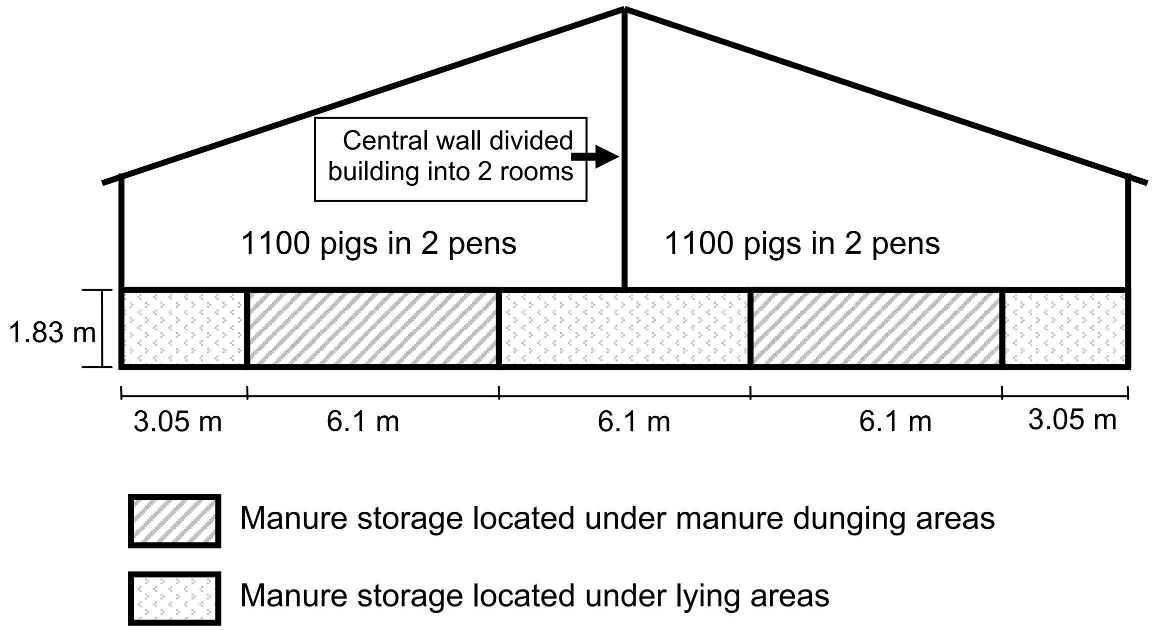


Figure 7.2. End-view of a swine finisher building showing manure storage pit arrangement below swine dunging and lying areas. Each pit was 85.3 m long.

The two pits located centrally in each room were located under observed dunging areas and were connected with a 0.25 m (10 inch) diameter pipe which allowed manure to flow freely between the pits to maintain similar depths between the pits. The pipes were located near the center of the building length. The three pits that were not located below the dunging area, but rather under lying areas along the walls, were connected with piping.

This manure pit configuration was designed to accommodate the collection of a majority of manure deposition in pits located centrally in each room and to remove that manure to an anaerobic manure digestion treatment system. These pits were termed collection pits. Once treated manure was returned from the digestion vessel to the three manure pits (termed return pits) located under the lying areas (along exterior and interior walls) of the buildings. Thus the five manure pits represent two separate manure storage systems within each building. Each of the two systems can hold roughly the same volume of manure as each system was located beneath 12.19 m (40 ft) of building width. Most feeders and swinging waterers in the houses are located above the collection pits to send wasted feed and water to the digester. Compared to swine houses with similar perimeter dimensions and standard deep-pit designs the extra costs of engineering and constructing the new under-floor manure storage structure in these two buildings was $33 410.25 Figure 7.3 demonstrates manure flow schemes of the system.

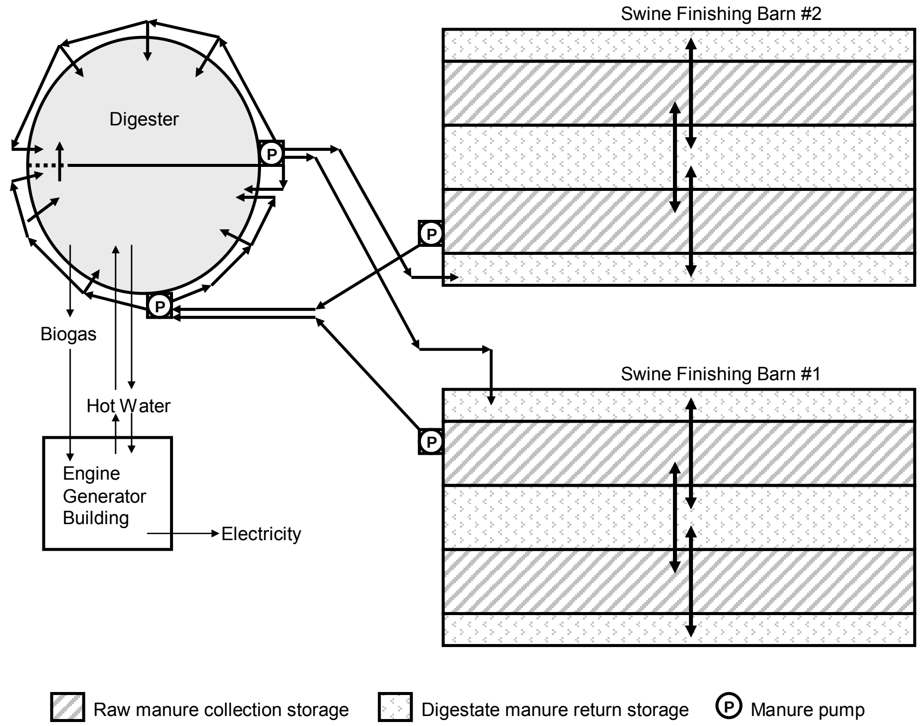


Figure 7.3. Schematic diagram of manure system. Unless labeled, arrows represent manure flow pathways. A pump located at each swine building pumped manure to the digester. The two pumps located at the digester were utilized for mixing of manure within the digester. Digestate returned to under-floor manure storage by gravity. Collection and return pits within the buildings were connected with piping that helped to maintain similar manure depth within the pits of that manure group.

To determine the amount of manure deposited into pits located either below or away from dunging areas, measurements of manure depth in the pits were taken. Because the manure systems were roughly equal in volume and dimensions, changes in depth were used to determine the percentage of manure deposited into each system. Measurements were taken through the slats and at consistent locations to limit possible influence from variations in the concrete pit floors. Depth data measurements were gathered using a pole graduated with 3.18 mm (1/8 inch) markings. These measures were conducted prior to operation of the anaerobic digester. No data were collected when manure volume was influenced by removal of manure from any of the pits for field application. Due to all-in-all-out herd management the buildings were periodically unpopulated after a herd was marketed. The rooms were washed during these unpopulated periods. Because this was representative of normal commercial operation for this swine unit measurements over these periods were maintained and wash water was included in the overall manure volume changes for each pit system. Manure production varied during the stocking and growing cycle. No manure was produced when pens were empty, and manure production increased gradually during the swine growth cycle. Typically, a single room was stocked in a week and the two barns would be populated over a 4-week period. Pigs were removed as market weight was reached. Although pigs in the pens populated first were oldest, younger pigs in pens were removed when market weight was reached. Removal to market typically occurred over a 4-week period. This removal system meant that although a pen was fully populated during a single population event the removal of pigs from that pen occurred over a longer time period. Thus manure production near the end of each growing cycle varied between pens, rooms and buildings. Consideration was not given to variations of manure deposition during various stages of the herd growth cycle.

The swine finishing buildings were each equipped with self-sorting technology. By modifying penning farm management could direct pigs in the large-pen living areas across a scale located at the food court entrance. The food court had a single entry point and two one-way exit gates. Located centrally, one scale serviced both pens within the room but weighed pigs only from one pen at a time. The scale worked in conjunction with an automated gating system and was equipped with a computer that could direct pigs into different pens based on animal weight, and provided the capability to select animals for marketing. As a herd approached market weight additional penning was put in place to segregate a holding area for market-weight pigs. Pigs that entered the scales and weighed greater than a pre-set market weight were directed to the holding area. This holding area contained access to feed and water. Benefits of this type of swine housing system include low capital costs of penning and feed bins, the ability to change feed formulation based on pig weight data, and low labor demands for sorting of market hogs. A schematic diagram of the large-pen swine finishing building with self-sorting technology is presented in Figure 7.4.

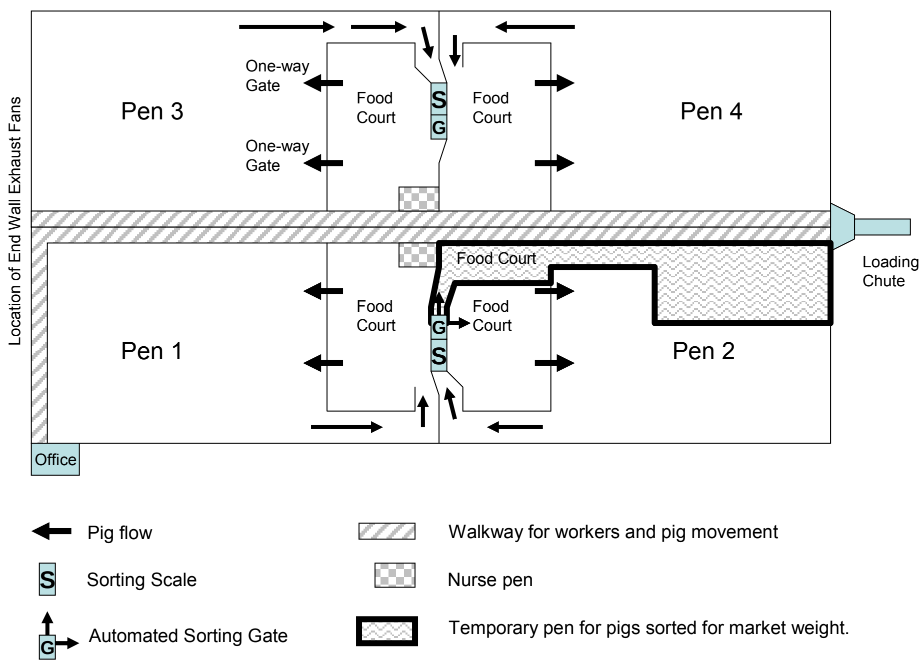


Figure 7.4. Schematic diagram of the large-pen swine finishing pen design. An automated scale and gating system in each room serviced both pens within the room, allowing management to utilize weight to sort pigs between pens and select pigs for market through various pen configurations. In this diagram pigs in Pens 1 and 4 entered the food court without crossing scales, weight data for pigs in Pen 3 was gathered as they crossed the scales without sorting, and market weight pigs in Pen 2 were selected to enter a temporary pen to await movement to market. A similar holding pen could be constructed in Pen 4. Pigs exited food courts through one-way gates. The diagram is not to scale.

Manure Management System

The circular concrete digestion vessel was 13.72 m (45 ft) in diameter and 3.66 m (12 ft) deep. The digester had an operational depth of approximately 2.74 m (9 ft), with a volume of approximately 396 m^3 (13 980 ft^3), or 395 911 liters (104 600 US gal). The two-chamber vessel had an interior concrete wall across its diameter. Manure entered one chamber and exited the digester from the opposite chamber. The top of the concrete central wall was the same height as the exterior wall, except for a spillway that was 1.22 m (4 ft) wide and 1.22 m (4 ft) high at the top of the wall. The bottom of the spillway was about 0.30 m (1 ft) below the depth of manure to permit flow from the first chamber to the second chamber. Manure was first pumped from the under-floor collection pits of the swine houses into a small pre-chamber concrete compartment that was as deep as the digester. Manure entered chamber 1 of the digestion vessel through an opening in the vessel wall between the pre-chamber compartment and the vessel. A similarly designed post-chamber compartment received manure from chamber 2 of the manure vessel. The openings between the pre- and post-compartments and the vessel were 0.91 m (3 ft) wide and 0.61 m (2 ft) deep, and were located 0.61 m (2 ft) above the digester floor. As manure was pumped into the pre-chamber compartment it displaced manure in the digester vessel that eventually flowed by gravity to the under floor return pits in the swine houses. Pump run times were adjusted by settings on automated timers which allowed manure to be pumped from each building 4 times each day. Manure was therefore pumped into the digester eight times daily (every 3 hours) alternately between buildings with each pumping cycle. However, farm management could choose to pump from only one building, at which times pump run-times from that building were increased. The volume of manure entering the digester from each building was adjusted by the producer according to manure availability within the buildings. Likewise, digestate was directed to one or both buildings according to management decisions such as current manure pit depths and desired pit location of manure removal for land application.

The original operational design provided mixing in chamber 1 of the digester, with chamber 2 operating in a plug flow manner. The addition of a mixing pump located at both the pre- and post- digestion compartments enhanced the mixing system, and essentially converted the digester into a complete mix system, by mixing manure within and between each chamber. To mix manure between the chambers manure from chamber 2 was pumped back to chamber 1, which in turn caused manure to flow over the spillway back to chamber 2.

Analysis of Manure Constituents

Manure analyses of influent and effluent manure were performed at the Pennsylvania State University Agricultural Analytical Services Laboratory (University Park, Pennsylvania, USA) and in accordance with the methods described by Peters *et al.* (2003). Samples of manure were obtained from pre- and post-chamber compartments during steady-state digestion operational period under complete mix management. Influent samples were gathered as grab samples directly from influent pipes when the swine building pumps were operating. The pumps were manually operated during data collection. Before sample collection, influent manure was allowed to run from the pipe end for about 3 minutes to minimize influences from manure that may have been located within the pipe since previous pump operation. Samples were quickly placed in an iced cooler for direct transport to the laboratory. Samples were collected on 16 different days between 4 January 2006 and 18 June 2007. On 7 of the 16 dates manure was collected from only one building because management was feeding the digester from only one building due to low manure levels in one building or an out-of-service pump. On 9 of 16 dates manure was collected from the pipe coming from each building with each sample being analyzed individually. The data from the two samples were averaged together for this report. Immediately after influent collection the pump(s) were turned off and without delay effluent manure was collected from the post-digestion chamber with a core sample extraction method using a Coretaker® (Raven Environmental Products, Inc. Saint Louis, Missouri, USA). Core samples were collected from a depth of approximately 2.1 m (7 ft) below the surface of the compartment. To assure that sampled material originated from a consistent depth the effluent was not allowed to flow into the sampler until the sampler's influent end was at the desired depth. This was accomplished by simply sealing the air outlet at the top of the Coretaker®. The Coretaker® was flushed with manure from the sample depth 5 times prior to each sample collection to minimize risk of influence of foreign matter or previously sampled material to sample quality. Analysis included percent solids, pH, percent carbon, total N, ammonia N (NH_3-N), calculated organic N, and nitrate N.

Methane Production and Utilization

The digester was equipped with a flexible polypropylene cover that inflated with gas pressure into a dome-shape. The cover plus digester freeboard space provide a maximum biogas storage capacity of 807 m^3 (28 511 ft^3). The methane-rich biogas was used as a fuel source for a Chevy V-8 engine (5.7 liters displacement), which drove a 220-volt, 3-phase, 60 Hz, electric generator rated at 47 KW for use with biogas (Martin Machinery, Ephrata, Pennsylvania, USA). The electric generation equipment included meters that recorded engine run-time and kilowatts of electricity produced. Volume of biogas was metered prior to entering the engine as it passed through a Roots® meter (model 3M300, Roots Meters & Instruments, Houston, TX, USA). Heat from the engine and its exhaust was captured by a heat exchanging system that transferred heat to the manure in the digester to maintain a mesophilic digestion temperature of approximately 35 ºC (95 ºF). A pressure valve diverted biogas to a flare when the digester cover was inflated to capacity [373.6 Pa (1.5 inches of water)]. The flare was equipped with sparking equipment to assure that all biogas diverted to the flare was combusted. Flared gas volume was not measured. The producer estimated that the flare operated less than 1% of the time. Biogas samples were collected from a petcock valve located on the biogas pipe between the digester and the engine. Fifty-three biogas samples were collected during start-up and steady-state operation, 23 of these samples were collected by the producer during a 2-year steady-state digestion period during which electricity production and biogas production was monitored. The samples were taken during routine system inspections and while engine operation was pulling biogas through the pipe. Because the producer used CO_2 content as a monitoring tool of digester performance, sampling occurred frequently near the beginning of start-up (about once every 3 days) and less frequently near the end of this period (about once every 45 days) as comfort level with system management increased and biogas content variation decreased. The carbon dioxide content of these biogas samples was measured with a Bacharach Fyrite® Gas

Analyzer to monitor biogas quality (manufacturer reported accuracy ±0.5%). Two biogas samples from this location were subjected to gas chromatography to provide a comparison to the values observed with the Bacharach Fyrite® Gas Analyzer. Gas samples were collected into a Tedlar® bag and transported directly to the Pennsylvania State University; Energy Institute, Gas Chromatograph Lab (Shimadzu GC17A Chromatograph) and analyzed within 2 hours of collection. Electricity produced at this farm-level digester was used to supplement the demand of the swine barns. Excess electricity was metered as it entered the utility grid and the producer received a credit against farm usage from the utility.

Swine Housing Gas Concentrations and Observations of Impacts on Herd Health

Air quality concerns arose due to possible gas emissions into the swine housing area from the introduction of digestate into the return manure pits, where the digestate could mix with raw manure. To minimize the potential negative impact on air quality due to manure surface disturbance the 0.25 m diameter pipes that deliver digestate to the manure pits were located at the bottom of the storage pits, ensuring that manure entered the buildings beneath the manure surface, unless the manure was less than 0.25 m deep. In addition, manure also entered the pit on the end of the tunnel-ventilated buildings that were closer to exhaust fans. Pipe locations are illustrated in Figure 7.5.

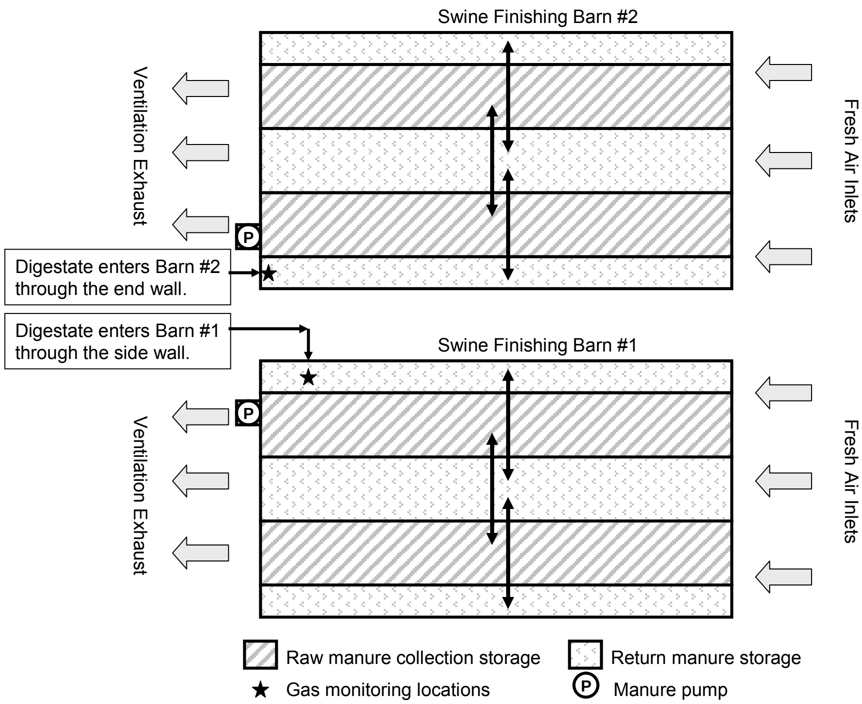


Figure 7.5. Digestate entered the building manure storage pits at the bottom of the pits and near exhaust fans. Digestate entered Barn #1 on its side wall approximately 6.4 m (21 feet) from the end-wall fans. Digestate entered Barn #2 through an end-wall, approximately 2.4 (8 feet) from the building corner and directly below exhaust fans.

A QRAE Plus four-gas monitor (RAE Systems, San Jose, California, USA) was used to collect data from several locations in each of the buildings. The monitor was equipped with ammonia, hydrogen sulfide, combustible gas [Lower Explosive Limit (LEL)], and oxygen sensors. The monitor was calibrated in accordance with product recommendations and sensors were replaced twice during the monitoring period. The unit was calibrated to fresh air, away from the swine housing area, prior to each monitoring event. The LEL sensor was calibrated using CH_4 at 50% LEL. Measurement accuracy of the monitor reported by the manufacturer was as follows: ±3 ppm or 10% of reading for NH_3 (range 0-200 ppm), ±2 ppm or 10% of reading for H_2S (range 0-100 ppm), ±3% LEL <50% LEL and ±5% LEL ≥ 50% LEL (range 0-100% LEL), and ±0.4 Vol % or 2% of reading for O_2 (range 0-30%). The monitor measured the concentration of each gas every second. These measurements were averaged over a preset time interval and recorded as a single data point. For example, when the time interval between measurements was set at 60 seconds the monitor recorded a single data entry based on the average of 60 readings. In the present study, the preset time interval ranged from 60 to 600 seconds. Because data were recorded using varying time intervals (60 to 600 seconds) the data were grouped into 1-hour blocks from which means and standard errors were calculated for NH_3, H_2S, and CH_4 concentrations at each monitoring location. These hourly concentrations were also used to identify the frequency in which NH_3 concentrations exceeded 25, 50, and 100 ppm.

Gas concentrations in the swine house living space were recorded prior to digester operation (between 11 November 2003 and 11 October 2005). Gas concentration measurements were repeated after digester operation commenced and at times when digestate was draining into the manure pits (between 20 April 2005 and 30 April 2007). Sample time period overlap was possible when farm management directed all digestate to one building during digester start-up, while no digestate was directed toward the other building. These measurements were taken in the swine living area at one of several locations; (1) ceiling height; (2) 1.22 m (4 ft) above the slatted floor; (3) pig level [0.15 m (6 inches) above the floor]; and (4) just below the slatted floor. To protect the monitor from tampering from pigs, samples at pig level and below the floor were taken through Teflon tubing that ran through the center of a steel pipe. The pipe was permanently secured to the floor and ceiling and the tubing inlet locations remained in fixed positions. The monitor was placed at one of the sampling locations and retrieved at a later time.

Data collection periods ranged from 50 minutes to about 14 days, which are termed monitoring events. The monitoring events occurred during the following months (number of monitoring events for the month): January (4), February (3), March (2), April (2), May (2), June (1), August (1), October (1), November (2), December (1). After digester operation commenced measurements were collected during the following months: February (1), March (2), April (4), June (3), July (2), October (1) at both pig level and below the slatted floor. Thus, 19 and 13 monitoring events were conducted before and after the digester became operational, respectively. No manure transfers other than those resulting from feeding of the digester occurred during these periods.

All air measurements from pig level and below floor level were taken near the pipe emptying digestate into the return pit where gas concentrations were expected to be highest. In Barn #1 the measurements were taken 1.8 m (6 feet) away from the side wall, and in Barn #2 readings were taken 0.76 m (2.5 feet) from the end-wall (Figure 7.5). Digestate entered under-floor manure storage near a corner of the buildings and near exhaust fans. For this reason, influence from the digestate was expected to impact only a small area. The pits that receive digestate were both along an outside wall. For digestate to enter other pits it would need to move to 25 cm diameter pipes located near the center of the pit and then move through the pipe. Only 1 of 4 pens in each building was located where pigs could occupy space near the introduction of fresh digestate.

In this observational case-study, building ventilation parameters were not monitored during gas monitoring events. The ventilation systems in each building was controlled by farm management and consisted of a computerized controller, static pressure meter, automatically controlled inlets, and end-wall fans. At the onset of data recording minimum ventilation was provided through continuous operation of pit ventilation fans. On 5 June 2006 (after 8 measurement periods with digestate present), the integrator directed the producer to remove the pit fans and provide minimum ventilation with a 127-cm end-wall fan placed on a timer [5 monitoring events were conducted after this change June (2), July (2), April (1)]. The ventilation change occurred during the period after digestate was introduced to under-floor manure storages. Data is reported for periods before and after the ventilation change.

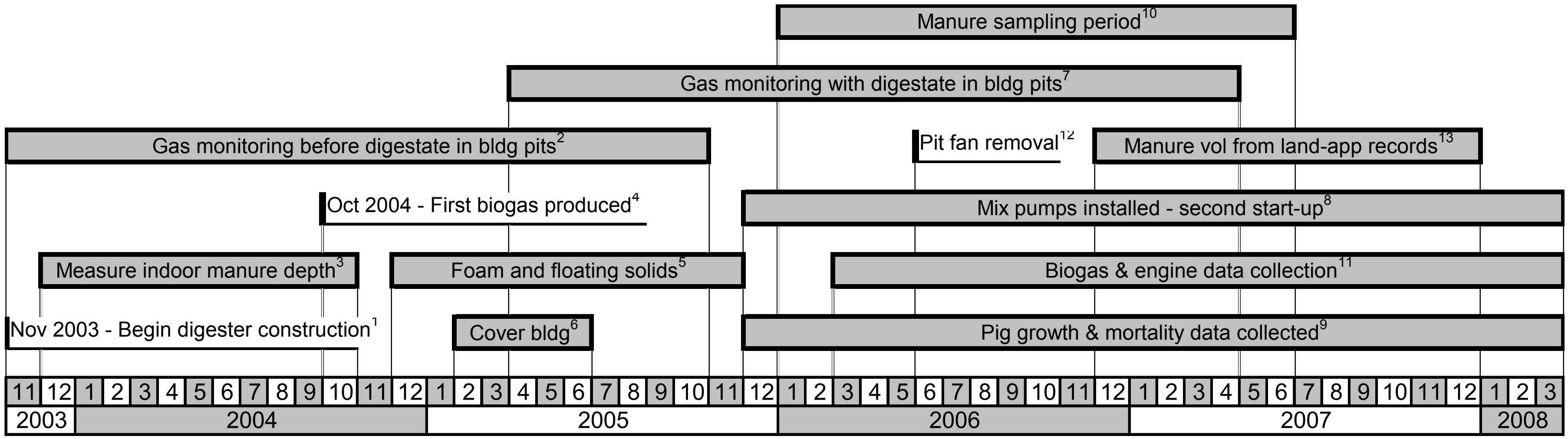
Evaluation of Swine Performance

During the period when digestate from steady-state digester operation was being introduced to the under-floor manure storage pits of the swine houses two different sow farms supplied feeder pigs to this commercial finishing operation. Sow Source 1 supplied pigs from 17 December 2005 to 26 November 2007, a total of 24 393 pigs representing 10 groups. A group of pigs occupies a single building and the integrator supplied information on growth and performance of each group in a report known as a “close-out”. Sow Source 2 supplied a single group of pigs to this farm during steady-state digestion (29 October 2007 to 27 February 2008). Closeout information from two standard grow-finish farms was gathered to allow comparison of pig performances, although the number of pigs in the groups is not large and many factors beyond the introduction of digested manure into the under-floor manure pits of the study farm may have impacted pig performance at the farms. One of the farms, Farm 2, reared 4 groups (8789 pigs) of pigs from Sow Source 1. This farm and a second farm (Farm 3) each reared 2 groups of pigs from Sow Source 2 (4439 pigs at Farm 2, 4489 pigs at Farm 3).

Event Timeline

Because this case-study involved several data collection stages which occurred at various times, a timeline representative of relative times of notable events is provided in Figure 7.6. Construction of digester system components occurred over several phases. Design and construction of the novel under-floor manure storage system found within the swine housing facilities was completed during construction of the housing units (November 2002). Digester construction did not commence until after the barns had been populated for over one year. The digestion vessel was covered and biogas production began in October 2004. However, long-term, steady-state digestion was hindered because foaming action within the vessel created floating solids which inhibited flow of manure across the central spillway of the digester, and clogged the biogas collection pipe, which was responsible for biogas pressure beneath the cover so great that the cover ruptured. It appeared that some contributing factors to floating solid build-up were 1) an insulation panel that floated on top of the digester manure, which acted to press foam together and inhibit movement of the manure surface, 2) inadequate mixing system, which did not target surface solids, and 3) a faulty thermocouple temperature probe system, which incorrectly displayed temperatures of about 35ºC, that were actually lower (around 29°C). The digester was shut down and floating insulation was removed, a more effective manure mixing system was implemented, and temperature was monitored by placing a hand-held bulb thermometer attached to a wire into two 3 m (10 ft) long, 1.25 cm (0.5

inch) inside diameter PVC pipes. The PVC pipes were capped on their bottom end and filled with water. One pipe was hung in the influent compartment and one in the effluent compartment, each of these angled into the opening between the compartment and the digester chambers. A cover building was installed over the digester to protect it from weather in 2005. A second start-up phase began in December 2006.



[1]Construction of the digester began in November 2003. The swine buildings, constructed with special under-floor manure collection pits, were populated in October 2002.
[2]Gas monitoring was conducted at pig level, below the floor, 4 feet above the floor, and at ceiling height prior to digestion operation.
[3]Under-floor manure storage depth was measured prior to digestion to determine the distribution of manure into collection system components located beneath both dunging and lying areas of the swine finishing buildings.
[4]Original digester operation commenced in October 2004.
[5]The original digestion period was hampered by the production of foam and floating solids, which were removed from the digestion vessel at the end of this period.
[6]A canvas cover building was installed over the digestion vessel to provide weather protection.
[7]Gas monitoring was conducted at pig level and below the floor at a location above where digestate entered the under-floor manure storage pits of the swine houses. Three monitoring events occurred during the initial digester operation and ten events occurred after a second start-up.
[8]New mixing pumps were installed in the digester and a second start-up phase began in December 2005.
[9]Pig growth and mortality data was collected during steady-state digester operation while digestate was introduced to manure storage located under swine living areas. Data from this farm were compared to farms with finishing swine stocked from the same sow source as these barns. Sow source switched in 2007.
[10]Manure was sampled from digester influent pipes, and also from the effluent compartment of the digester.
[11]Biogas and electric production was monitored over a 24-month period (coinciding with utility billing cycles) when the digester operated at steady-state.
[12]The swine integrator requested removal of minimum ventilation manure pit exhaust fans and converted minimum ventilation to an end-wall fan on a timer in June 2006.
[13]Land application records were used to estimate volumes of manure digested and produced at the farm.

Figure 7.6. Timeline representation of case-study events at a swine finishing farm with novel manure collection and anaerobic digester systems. Times are represented on a monthly scale.

Statistical Analysis

Manure analysis data and swine house air quality measurements were analyzed via Proc GLM (Ver. 9.1.3, SAS Inst., Inc., Cary, NC) as completely randomized designs. For manure analysis data, comparisons of nutrient concentrations in influent and effluent manure were made using day of collection as the experimental unit. For swine house air quality measurements, comparisons of gas concentrations were made before and after the beginning of digester operations, using mean hourly gas concentrations as the experimental unit. Records of swine performance were analyzed via Proc Mixed (Ver. 9.1.3, SAS Inst., Inc., Cary, NC) , using the closeout group average for each measure of performance within a building as the experimental unit. The source of pigs was included in the model as a random effect. A probability value of 0.05 was used to declare statistical differences.

RESULTS

Manure Management System

The amount of manure deposited into the collection pits of the two buildings combined over a 9-month measuring period prior to digester operation was 75% [73% in Building 1 (73 measurements) and 77% in Building 2 (82 measurements)].

Comparisons of solids, carbon, sulfur and nitrogen fractions (total N, ammonium N, organic N, and Nitrate N) of influent and effluent manure sample sources are presented in Table 7.1. Except for ammonium N and nitrate N all of these components decreased significantly ($P < 0.05$). Digestion caused a rise in pH ($P < 0.05$).

Table 7.1. Comparison of analyses (LS means) from manure collected before (influent) and after (effluent) anaerobic digestion on a commercial swine finisher enterprise.

Manure Component	Influent n = 16	Effluent n = 16	S.E.
Solids (%)	7.2^a	3.8^b	0.47
pH	7.3^a	8.1^b	0.07
Carbon (%)	3.5^a	1.6^b	0.34
Total N (g/liter)	5.9^a	5.1^b	0.23
Ammonium N (g/liter)	3.7	3.9	0.13
Calculated Organic N (g/liter)	2.2^a	1.2^b	0.18
Nitrate-N (mg/liter)	0.17	0.26	0.077
Sulfur (g/liter)	0.58^a	0.36^b	0.03

^{a,b} Values within a row with different superscripts are different ($P < 0.05$).

According to farm manure application records 6 633 441 liters (1 673 300 US gal) of manure were removed from the return storage pits over a 340-day period (13 February 2006 through 18 January 2007). During this time 2.86 groups of pigs occupied the barns, which included the tail end of a group, two full groups and the beginning of a fourth group. Using average pig weights and daily manure production, calculated manure production was 46.97 liters/d per 500 kg of animal weight (11.36 gal/d per 1000 lbs of animal weight). The amount of manure collected in the pits located under the dunging area (75% combined average for both building) was used to estimate the volume of manure directed to the digester. Based on a total digester vessel volume of 395 911 liters (104 600 gal) and an estimated loading rate of 14 160 liters/d (3741 gal/d), the calculated hydraulic retention time of the digester was therefore an estimated 28 days.

Methane Production and Utilization

At this farm a CO_2 monitor was used to indirectly estimate content of CH_4 in biogas under the assumption that nearly 100% of biogas content was composed of CO_2 and CH_4. Based on the average of CO_2 readings during steady-state operation, the estimated CH_4 concentration was about 72% and within the expected range of 60 to 80% (Roos *et al.* 2004). Two biogas samples both of which with a CO_2 content of 28%, as measured by the Bacharach Fyrite® Gas Analyzer, were subjected to gas chromatography and had CO_2 concentrations of 30.5 and 28.8% and CH_4 concentrations of 72.4 and 68.3%, respectively.

Steady-state monitoring of the digester and electricity production systems was conducted over a 727-day period (6 March 2006 through 2 March 2008). This period coincided with

24 monthly billing cycles from the utility supplier. During the same period CO_2 content averaged 28.9% (range 24-32%, S.E.=0.3), and 147 138 m^3 (5 195 437 ft^3) of biogas was directed to the engine. Total engine run-time was 9058.33 hours (avg. 12.46 h/d), producing 170 140 kWh (avg. 234.0 kWh/d) of electricity. The average hourly electricity production was 18.8 kWh/h of engine operation. The amount of gas required to produce a unit of electricity was 0.86 m^3/kWh (30.54 ft^3/kWh). Additional energy production created heat that was transferred from the engine to the digestion vessel to maintain mesophilic temperatures. These values are summarized in Table 7.2.

Table 7.2. Summary of biogas production and engine operation information from a 727-day steady-state operational period.

Efficiency Parameter	Units	Value
Average biogas CO_2 content (n = 23, S.E.=0.3)	%	28.9
Range of biogas CO_2 content	%	24-32
Biogas produced	cu m	147 138
Hours of engine operation	hrs	9058.33
Daily hours of engine operation	hrs/d	12.46
Electricity production	kWh	170 140
Daily electricity production	kWh/d	234
Electricity produced per hour of engine operation	kWh/hr	18.8
Biogas per unit of electricity	cu m/kWh	0.86
Estimated manure volume directed to digester	liter	10 308 383
Estimated biogas produced per unit of manure	cu m/1000 liter	13.21
Estimated manure units per unit of electricity	liter/kWh	60.6

Although engine operation averaged 12.46 h/d, standard management of the system allowed biogas to accumulate under the digester's flexible storage cover [up to 807 m^3 (28 511 ft^3)], which allowed the engine to operate for longer periods, and in turn remain shut-off for longer period. For example the engine may have operate for 30 continuous hours and then been shut down for 30 hours while biogas accumulated in storage. Shut-down was conducted manually by the producer when biogas in storage was low.

Information provided by the electric utility supplier reflected the electricity purchases for the swine and digester operation. The information begins and ends according to 24 monthly utility billing periods and overlaps with the steady-state digester operation and farm electric generation period above (727-day period from 6 March 2006 through 2 March 2008). Total electricity purchased from the utility was 168 378 kWh (7015.8 kWh/mo). The total costs for purchased electricity during this time was $19 285.61 ($803.57/mo), of which $11 333.94 ($472.25/mo) was for energy and $7951.67 (331.32/mo) was for charges associated with distribution and transmission of electricity on the utility grid. The average retail purchase price of electricity was $0.0673/kWh ($11 333.94/168 378 KWh). When produced by the digester system, electricity was first directed to the digester system needs (e.g. manure pumps) and the swine facilities. If electric demand of these two systems was below the amount produced by the generator the excess electricity was directed to the utility grid. Net metering was used to subtract the amount of electricity that entered the grid from the monthly utility invoice. Thus all electricity produced represented an electricity cost-savings by reducing electricity purchase. During this 24-month period farm-level electric generation had a value of

$11 450.42, or $477.10/mo (based on a value of $0.0673/kWh). The values of monthly electric purchase and monthly electric production were added to calculate the total value of monthly electric demand for the farm of $937.35. The digester system's electricity production accounted for 50.3% of the total electricity demand in the swine barns and digester system.

Total project costs were $242 033.83. This included all costs associated with barn manure pit reconfiguration, digester construction, electric generation equipment, digester cover building, and improvements of manure mixing equipment. Grant funds, from five separate sources, totaled $199 613.53, leaving a producer balance of $42 420.30. Besides electrical savings, the farm was enrolled in a carbon trading program. From January 2005 through May 2007 the destruction of methane at the farm generated 500 credits. The value of credits at issue was $1590.00, although the producer retained the credits in a market pool where value accrual was possible.

Swine Housing Gas Concentrations

A summary of hourly mean concentrations of NH_3 and H_2S in animal housing are presented in Table 7.3 according to air sampling location and whether the reading was attained before or after the introduction of digestate to under-floor manure storage pits. Pit ventilation was operational for all of the data in the table. After digestate was present no observations were made 1.22 m above the slatted floor or at ceiling height. Methane concentrations averaged 0.17 ± 0.03 ppm and there were no statistical differences ($P > 0.05$) between samples collected before and after (data not shown).

Table 7.3. Concentrations of NH_3 and H_2S gases in swine housing before and after the introduction of digestate into the under-floor storage pits at various measurement locations.

Hourly Ammonia Concentrations (ppm)	Before Digestate	S.E.	After Digestate	S.E.	Probability
Below Floor	40	4.7	34	1.3	NS
Pig Level	24	2.8	17	1.0	*
1.22 m above floor	19	1.1	NA	NA	NA
Ceiling Height	11	1.0	NA	NA	NA

Hourly Hydrogen Sulfide Concentrations (ppm)	Before Digestate	S.E.	After Digestate	S.E.	Probability
Below Floor	0.50	0.094	0.41	0.027	NS
Pig Level	0.63	0.057	0.02	0.021	***
1.22 m above floor	0.10	0.023	NA	NA	NA
Ceiling Height	0.47	0.020	NA	NA	NA

* $P<0.05$

*** $P<0.0001$

NS = not significant

NA = not available

A summary of hourly mean concentrations of NH_3, H_2S, and LEL gases in animal housing are presented in Table 7.4 according to air sampling location and whether the reading was attained when minimum ventilation pit fans were operational or not. All data in the table were collected when digestate was present in under-floor manure storage pits. Collection points were at pig level and below the slatted floor.

Table 7.4. Concentrations of NH_3, H_2S, and Lower Explosive Limit (LEL includes CH_4) gases in swine housing before and after the introduction of digestate into the under-floor storage pits and whether minimum ventilation was provided with under-floor manure pit fan ventilation after digestate introduction.

Hourly Ammonia Concentrations (ppm) After Digestate					
	With Fan	S.E.	No Fan	S.E.	Probability
Below Floor	34	1.8	6	1.6	***
Pig Level	17	1.4	37	0.9	***
Hourly Hydrogen Sulfide Concentrations (ppm) After Digestate					
	With Fan	S.E.	No Fan	S.E.	Probability
Below Floor	0.41	0.032	0.02	0.029	**
Pig Level	0.02	0.025	0.30	0.016	***
Hourly LEL Concentrations (%) After Digestate					
	With Fan	S.E.	No Fan	S.E.	Probability
Below Floor	0.3	0.07	0.2	0.06	**
Pig Level	0.0	0.06	0.8	0.04	***

** $P<0.01$
*** $P<0.0001$

Prior to digestate introduction to under-floor manure storage areas the maximum mean hourly concentration level of H_2S from all sampling locations was 0.6 ppm, while no LEL gas was detected. After digestate introduction maximum measured hourly concentration at any location was 2.5 ppm for H_2S and 2.8% for LEL (data not shown). Oxygen (O_2) levels were also recorded. The minimum mean hourly value recorded for O_2 before and after digestate introduction were 20.6 and 20.0%, respectively (data not shown). The occurrence of NH_3 levels above 25, 50 and 100 ppm after the introduction of digestate appear in Table 7.5.

Table 7.5. Frequency of hourly NH_3 concentrations exceeding 25, 50 and 100 ppm after digestate introduction to under-floor manure storage for sample locations at pig level and below floor level.

Location of NH_3 monitoring	Percent of hourly NH_3 concentration means within range*
After digestate, With pit fan	
Pig Level, n = 271	
>25 ppm	28.3
>50 ppm	1.5
>100 ppm	0.0
After digestate, With pit fan	
Below Floor, n = 161	
>25 ppm	45.3
>50 ppm	19.3
>100 ppm	6.8
After digestate, No pit fan	
Pig Level, n = 654	
>25 ppm	61.6
>50 ppm	24.3
>100 ppm	2.6
After digestate, No pit fan	
Below Floor, n = 197	
>25 ppm	6.6
>50 ppm	2.0
>100 ppm	0.0

*Percents represent the number of hours in which hourly mean NH_3 concentrations were above the listed level.

During the period when digestate from steady-state digester operation was being introduced to the under-floor manure storage pits of the swine houses two different sow farms supplied feeder pigs to this commercial finishing operation. Performance indicators provided by the swine integrator from closeout information for groups of pigs reared in the Case Farm buildings and pigs reared in two standard grow-finish farms, Farm 2 and Farm 3, are presented in Table 7.6. The Case Farm raised 10 groups of pigs (22 055 pigs) from the first sow farm (Sow Source 1) and 1 group (2338 pigs) from the second sow farm (Sow Source 2). One of the swine finishing farms, Farm 2, reared 4 groups (8789 pigs) of pigs from Sow Source 1. Farm 2 and Farm 3 each reared 2 groups of pigs from Sow Source 2 (4439 pigs at Farm 2, 4489 pigs at Farm 3). A group of pigs are all of those that enter a building at the beginning of a single growth cycle. In this record keeping system culls and mortalities are grouped together because, due to economic considerations, some pigs were euthanized instead of being sent to a cull market. Average daily gain of pigs on the case farm was lower than that of Farm 2, but statistically similar ($P > 0.05$) to that of Farm 3. Feed to gain ratios and a combination of culling and mortality were similar among the three farms included in the analysis ($P > 0.05$).

Table 7.6. LS means of growth performance and culls and mortality in groups of pigs housed above manure pits receiving digested manure compared to that of pigs housed in standard grow-finish facilities.

	Case Farm	SE	Farm 2	SE	Farm 3	SE
No of groups	11	-	6	-	2	-
ADG, kg	0.77^{a}	0.01	0.89^{b}	0.03	0.81^{ab}	0.02
F/G	2.63	0.05	2.34	0.11	2.47	0.06
Culls & mortalities, %	5.92	0.99	3.56	1.21	3.66	1.21

^{ab} Means within a row with different superscripts are different ($P < 0.05$).

DISCUSSION

Modifying the under-floor manure structure at this commercial operation showed that the behavioral characteristics of the finishing swine in large-pen housing can be utilized to eliminate the necessity of additional manure storage capacity for post-digested manure. By collecting approximately 75% of the raw manure we successfully operated a mesophilic anaerobic digester. Estimates of manure production for this system of 46.97 liters/d per 500 kg of animal weight (11.4 gal/d per 1000 lbs of animal weight) were consistent with local industry expectations (Penn State Agronomy Guide 2006).

Accurate monitoring of changes in mass balance of nutrients was beyond the scope of this case-study but concentrations of influent and effluent manure components were monitored. Mean percent solids decreased from 7.2 to 3.8% ($P<0.05$), but only part of this reduction can be explained by the removal of carbon-based biogas from the system (Table 7.1). The percentage of carbon (C) in manure samples decreased from 3.5 to 1.6% ($P<0.05$). Using the laboratory density standard for manure (1 kg/liter) (Peters *et al.* 2003) and assuming that the calculated annual loading and effluent volumes were equal, the average percent of solids and C from paired influent and effluent manure samples was used to estimate the total weight of these manure components entering and exiting the digester (calculations not shown). The annual volume of biogas directed to the engine, average biogas content, and molecular weights of CH_4 and CO_2 were used to estimate the weight of CH_4 and CO_2 exiting the system in biogas form. The annual weight of CH_4 (29 104 kg) and CO_2 (74 820 kg) removed in biogas was 103 924 kg, equivalent to a total of 56 910 kg of C. The volume of biogas directed to the flare was not considered in this estimate because the flare operated less than 1% of the time, based on producer estimates. The amount of C removed during digestion, based on estimates of mass flow weights of manure and changes in carbon content of influent and effluent samples, was 90 270 kg. Therefore, we cannot account for 33 360 kg C (90 270 kg – 56 910 kg), and we believe it remained in the digester, as part of settled or floating solids.

Others report that anaerobic digestion of manure has little effect on total nutrient content but it can change the form of nutrients (Lantz *et al.* 2007; Salminen *et al.* 2001; Shih 1987, 1993; Vermeulen *et al.* 1992). In this digester, a significant decrease in total N concentration (5.9 to 5.1 g/liter, $P<0.05$) occurred. Similarly, the concentration of organic N decreased (2.2 to 1.2 g/liter, $P<0.05$), which was expected as the digestion process converts some of the N in organic matter to ammonium (NH_4^+) (Börjesson & Berglund 2003, 2007). Sommer *et al.* (2001) reported that the amount of NH_4^+ representing 70 and 85% of total nitrogen content in digested and undigested manure, respectively. Although we observed numeric increase in the concentration of ammonium N (3.7 to 3.9 g/liter) the change was not significant ($P>0.20$).

Another manure constituent of interest is sulfur (S), the concentration of which decreased (0.58 to 0.36 g/liter, $P<0.05$) during anaerobic digestion. While concentrations of S compounds, including H_2S, were not measured in biogas it appeared that S may leave this system as a constituent of biogas. Concentrations of biogas H_2S as high as 609 ppm from swine manure digestion have been reported (Pagilla *et al.* 2000).

The monthly value of electric produced at the farm was \$477.10. The farm secured a low interest loan through an environmental fund of 2% interest for 8 years. The producer costs of construction were \$42 420.30, which equates to a monthly payment of \$478.54 under the loan parameters. Electrical costs and usage include electricity demands of the manure management and digestion systems. For this system the savings in electrical costs were approximately equal to the costs of debt service over the two-year period of evaluation. Without grant funding the monthly financial responsibility for a loan for the entire project (\$242 033.83), using the current low interest loan structure, would be \$2730.35/mo. If an interest rate were supplied through a commercial lender instead of a special environmental fund for the same 8-year term the rate would be much higher. For example, if a 6% interest rate were applied to the proportion of total cost that the producer was currently responsible for (\$42 420.30) the monthly payment would be \$557.46, while the monthly payment for the entire project (\$242 033.83) at this interest rate would be \$3180.67/mo. Based on electric cost-savings none of these alternative scenarios would service construction debts. External funding and a low-interest loan were both necessary for economic feasibility of this anaerobic manure digester on the basis of electricity cost-savings for the first 8 years based on steady-state production. However, this digester did not attain long-term steady-state until a second start-up phase began about 14 months after construction ended, which extended the time period in which incurred debt will be serviced by future electric cost-savings. Current increases in energy costs will reduce loan payback term.

One of the benefits of this system was the elimination of post-digestion manure storage. A manure storage comparable in size to the total manure storage capacity of the two buildings combined would be about 3785 m^3 (1 000 000 gal). A standard industry alternative storage system for this volume of manure would be a high-density polyethylene (HDPE) lined, 3.7 m (12 ft) deep (including freeboard), sloped-bank earthen storage basin that measured 47.5 m (150 ft) square at the top of its banks. If the basin were constructed with a concrete agitation ramp, leak detection system, and chain-link perimeter fence the costs, including excavation, in Pennsylvania would be about $65 000 (W. H. Latshaw, personal communication; V. Brubaker, personal communication). The costs associated with construction of the compartmentalization of under-floor manure storage at these two barns was $33 410. The difference between the estimated cost of a conventional post-digester storage and the unique configuration in the present study was $31 590. If external grant funding remained constant and this amount were added to the current borrowed sum the producer would be responsible for $74 150 of financed funds and a monthly payment of $836.48 using the interest rate (2%) and term (8-year) for borrowed funds at this location. In this scenario the 25% manure volume that previously was not digested would be available for digestion. Assuming the same biogas and electricity production efficiencies for the additional manure the value of electricity produced would be equal to $634.54/mo, or 67.5% of the monthly total electric usage value of $940.20. Thus, the increased construction costs of the post-digestion manure storage basin would make anaerobic digestion less financially desirable at this farm.

Additional costs associated with this digester involved the labor and time of the producer. Labor cost-savings were realized by the producer during construction because the producer was able to self-perform the work for several phases of construction. Solid accumulation issues during the initial start-up period forced the producer to dedicate a large amount of time to repairs and upgrades of the digestion system. During steady-state operation the producer spent 4-5 h/mo on routine oversight of the digester, which includes daily visits to the digester to review biogas production and operation of the engine and heat exchanger. Additionally, digester maintenance, such as changing oil, required 1-2 h/mo of the producer's time. The producer was not directly compensated for this time, nor was this time considered in economic evaluation of the project. In addition, we acknowledge inevitable accumulation of manure solids in the digester, which will necessitate future removal of the digester cover and subsequent removal of all manure and manure solids. This process would entail a shut-down of the system, several days of work, and be followed by another start-up period.

Additional financial benefits at this farm were realized from marketable carbon credits. When collected CH_4 is combusted and used for heat or energy purposes, a replacement of fossil fuel use occurs, thereby reducing carbon dioxide emissions from the fossil fuels (Salminen & Rintala 2002). A single carbon credit was awarded for each US ton of CH_4 combusted in the engine. Replacing fossil fuels with biogas reduces the emissions of greenhouse gases, nitrogen oxides, hydrocarbons, and particulates (Börjesson & Berglund 2006). The value of the carbon credits received for this operation at issue was $1590.00. In addition, anaerobic digestion decreases the emission of nitrous oxide (N_2O) from land-applied manure compared to non-digested manure (Petersen 1999) because the organic matter remaining in digested manure is less likely to undergo microbial decomposition than that found in untreated manure. Furthermore the smaller organic molecules found in the decomposed organic matter provide less energy to support the growth of N_2O forming microorganisms in soil (Sommer *et al.* 2000). The global warming potential of CH_4 and N_2O (compared to that of CO_2 on a 100-year time horizon) is 23 and 296, respectively (IPCC 2001), thus any reduction in the emission of these two gases may also help reduce negative atmospheric change. Finally, social benefits of odor reduction are realized when swine manure is anaerobically digested with hydraulic retention times over 20 days (Fischer *et al.* 1984). It is difficult to assign monetary value to these collective benefits, however livestock producers are under increased pressure to reduce impacts on the local community and society in general; anaerobic digestion represents an important step in environmental sustainability.

Manure biogas systems that flare gas instead of utilizing the gas for energy production can be much less expensive to implement. In the present study the costs associated with installing a generator set were about \$90 000. Without public funding the producer cost of construction in this scenario would have been about \$142 000. Using current loan guidelines the monthly payment for this amount would be \$1600. If electric generation equipment were not installed at this location the only monetary offset for debt service would be the sale of carbon credits. The generation of carbon credits would not provide adequate debt service in this scenario.

The adoption of this manure collection system raised justifiable concerns of potential negative impacts to swine health and production stemming from the introduction of biologically active digestate to manure storage located directly beneath swine lying areas. An additional concern was that methane concentrations in confined areas both below and above the slatted floor may lead to explosive concentrations of the gas. Ventilation of swine barns is important to remove pollutants, provide fresh air and regulate temperature (Wathes *et al.* 1983). Gas concentrations vary along the length of a building (Robertson & Galbraith 1971) as well as within the height the building (Skarp 1975). Four gases, NH_3, H_2S, LEL (including CH_4), and O_2, were monitored at various heights in the swine units at locations where the potential for dangerous levels of these gases was considered greatest.

Ammonia and hydrogen sulfide are known to couple with dust in etiology of disease for both humans and pigs (Zhang *et al.* 1994) and have negative physiological effects on livestock, such as respiratory stress (Anderson *et al.* 1987). For humans, the Recommended Exposure Limit for gaseous ammonia (NH_3) is 25 ppm and the US Occupational Safety and Health Administration's Permissible Exposure Limit for gaseous NH_3 is 50 ppm (NIOSH 2005). The US Occupational Safety and Health Administration's Permissible Exposure Limit for gaseous ammonia is 50 ppm (OSHA 1999). According to the United States Department of Health and Human Services' Agency for Toxic Substances & Disease Registry (ATSDR 2007) the effects of gaseous ammonia for humans at various concentrations are as follows:

- 25-50 ppm - Detectable odor; unlikely to experience adverse effects

- 50-100 ppm - Mild eye, nose, and throat irritation; may develop tolerance in 1-2 weeks with no adverse effects thereafter
- The maximum airborne concentration below which it is believed that nearly all individuals could be exposed for up to 1 hour without experiencing or developing irreversible or other serious health effects or symptoms which could impair an individual's ability to take protective action is 200 ppm for NH_3.

For growing pigs NH_3 levels of 10 ppm can depress growth and feed intake (Jones *et al.* 1996). Ammonia concentrations of 50 ppm can inhibit pig growth, lead to mild respiratory disorders, and is a gastrointestinal irritant (Drummond *et al.* 1980). Acute exposures, defined as single, non-repetitive exposures of not more than 8 hours, to the highest NH_3 levels recorded in commercial piggeries (100-200 ppm) may result in irritation and secretion from ocular, conjunctival, nasal, and pharyngeal membranes (e.g. Stombaugh *et al.* 1969; Curtis *et al.* 1975; Donham *et al.* 1989). Reported ranges of NH_3 concentrations in swine houses include 1-30 ppm (Do Boer & Morrison 1988), and 20-75 ppm (Skarp 1975), with average concentrations below 20 ppm reported from exhaust locations (Ni *et al.* 2000). The 10-minute recommended exposure limit for hydrogen sulfide is 10 ppm (NIOSH 2005). Methane seems to have no negative health impact unless the concentration is so high that it displaces oxygen. Low oxygen levels can have negative impacts on health. The minimum oxygen concentration level for safe human entry is 19.5% (NIOSH 1990). In addition, methane can be flammable or explosive at 5-15% gaseous concentration by volume (Muehling 1969; Lioa 1996; NIOSH 1990). Under the conditions of this case-study, dangerous concentrations of H_2S, CH_4, and O_2 were not recorded at any time or sample location.

Mean hourly ammonia concentrations at pig level were 24 ppm during 36 hours of monitoring prior to the introduction of digestate to under-floor manure storage pits at this farm, compared to 17 ppm after digestate introduction during 271 hours of monitoring (Table 7.3). At pig level when digestate was present and pit fans were operational mean hourly NH_3 concentration readings exceeded 25, 50, and 100 ppm 28.3, 1.5, and 0.0% of the time, respectively. After pit fan removal the levels of 25, 50, and 100 ppm were exceeded during 61.6, 24.3, and 2.6% of readings, respectively (Table 7.5).

There are many factors that may contribute to variation in the readings of NH_3 concentrations at these locations. Ammonia concentration in manure depends on three parameters of the manure: total ammonia concentration, temperature, and pH (Hansen *et al.* 1998). Flux rates and the amount of NH_3 that enters the atmosphere from manure are dependent upon two thermodynamic equilibria: ammonia gas/liquid equilibrium and ammonia dissociation equilibrium in the liquid. These characteristics are dependent on pH and temperature, and NH_3 losses increase with increases in either or both of these conditions (Bonmatí & Flotats 2003). Other factors which influence NH_3 emissions from manure include emitting surface area (Aarnink *et al.* 1996) air temperature (Muck & Richards 1983), and air speed (Olesen & Sommer 1993; Zhang *et al.* 1994).

In this commercial farm case-study a factor that favored NH_3 release include a significant increase of manure pH during digestions (from 7.3 to 8.1, S.E.=0.07). Manure temperature was also increased by supplemental heating means during digestion, which may have increased NH_3 emission rates. However, the temperature of digestate likely decreased when mixed with manure in the storage pits beneath swine lying areas, which may have helped to inhibit activity of mesophilic bacteria found in the digestate. Furthermore, decreasing temperature will decrease gas flux rates (Aneja *et al.* 2001). Mixing digestate with fresh manure may have lowered pH of the digestate after it entered under-floor manure storage, although the amount of mixing between manure volumes was not known. Factors that influence release and in-house concentrations of NH_3 such as temperature and ventilation rates were not monitored at this commercial enterprise. We believe that after pit fans were removed that background NH_3 concentrations of sampled air were greater because no exhaust occurred along the length of the building after pit fans were removed. Therefore, all NH_3 emissions from the pits and floors of the houses were pulled to the end of the building near monitoring locations. Other factors that may have influenced gas concentration data include animal stocking density, age or weight of pigs, feed ingredients, inside temperature settings, outside temperatures, ventilation rates, level of manure in the under-floor pits, and amount of urine and feces found on top of the slatted floor. Results from air quality monitoring at this farm should be viewed with caution due to the factors listed here, and others, which may have influenced the data.

Growth performance and cull and mortality rates for groups of pigs stocked after digestate was present in under-floor manure pits were compared to groups raised from the same sow sources at other farms. Groups of pigs from the case farm were housed in the buildings during times when pit fans were both operating and not operating. Comparison of growing pig performance between the case farm and two other swine finishing farms, as assessed by ADG and F/G, revealed that performance at the case farm was poorer ($P < 0.05$) than that at Farm 2, but similar to that of Farm 3 ($P > 0.05$). The feed to gain ratios and the percentage of combined culls and mortalities were similar among the three farms ($P > 0.05$) (Table 7.6). Many factors can influence swine finishing performance at commercial enterprises. Group closeout data must be examined with care. The sample size of reported closeout information is small. Only one of four pens in each building housed pigs above the area where digestate entered under-floor manure storage and the location where the risk for negative gas concentrations considered greatest was near exhaust fans. Some factors beyond air quality that likely had impact on pigs reared at the different locations may include swine health at entry, differing management skills, and different building designs. Additional research concerning effects of this manure treatment system on swine health is warranted.

As with most novel systems, the producer encountered challenges with the digestion system. If such a system were to be built again several changes are recommended by the researchers and managers of this operation. For safety reasons it is recommended that minimum ventilation never be turned off, even when buildings are not populated. A second recommendation is to utilize a designer, builder, or consultant that has successfully implemented a manure digestion system similar to the proposed system. The producer should evaluate and accept financial risk associated with construction of a new digestion system. The producer must be willing and capable to learn about the new system and its operation. Specific to this farm, penning modifications would likely increase the percent of manure collected for digestion in central pits and increase biogas production. The largest penning improvement opportunity likely came during periods when the hogs were being automatically sorted for marketing. This period occurred when pigs were at their largest and producing more manure than at other periods of the growth cycle. During sorting penning at this farm was oriented longitudinally in the barns (Figure 7.4) and caused the observed dunging patterns to shift towards the exterior wall with part of the dunging pattern located above pits designated to hold digestate. Figure 7.7 shows a suggested orientation for temporary penning of pigs sorted for market that may increase the amount of manure deposited into central manure collection pits, compared to the longitudinal temporary pen orientation used at this facility.

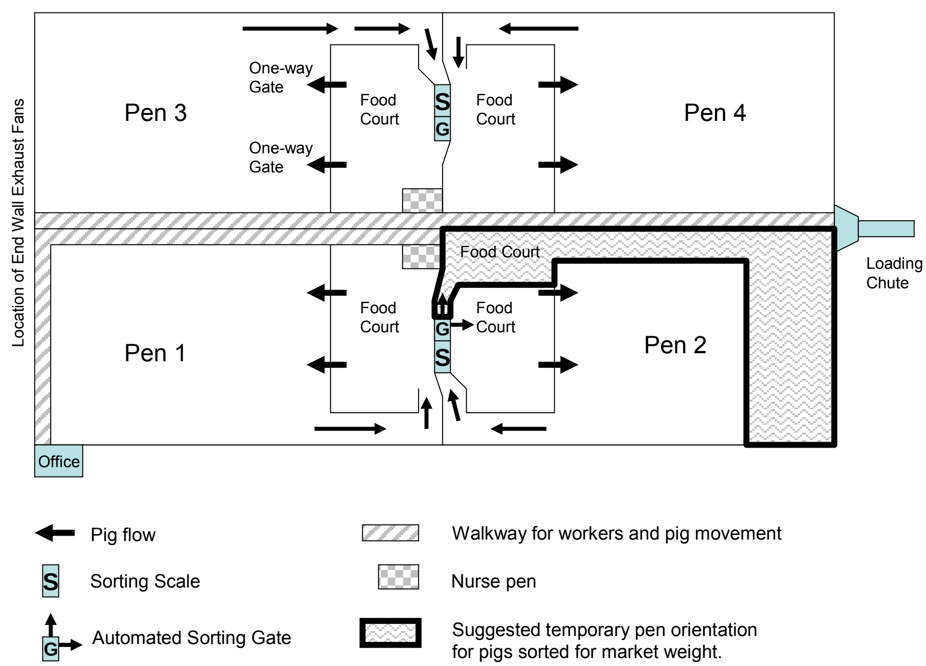


Figure 7.7. Suggested pen orientation for finishing swine sorted for movement to market for a facility in which raw and digested manure is stored beneath the facility. The orientation of temporary penning for pigs sorted to move to market in Pen 2 may increase the amount of manure collected in central under-floor manure pits whose manure is directed to the farm digester and thereby lead to a greater percentage of total manure production that is directed to the digester, as compared to current temporary penning orientation. The pen utilizes walkway space to widen the movement alley for pigs traveling to or from the food court.

Another recommendation involves modification of the under-floor manure collection system. The modification would eliminate the piping that allows manure to move between storage pits in the current design (Figure 7.3) by reconfiguring the pits into two compartments instead of five as illustrated in Figure 7.8.

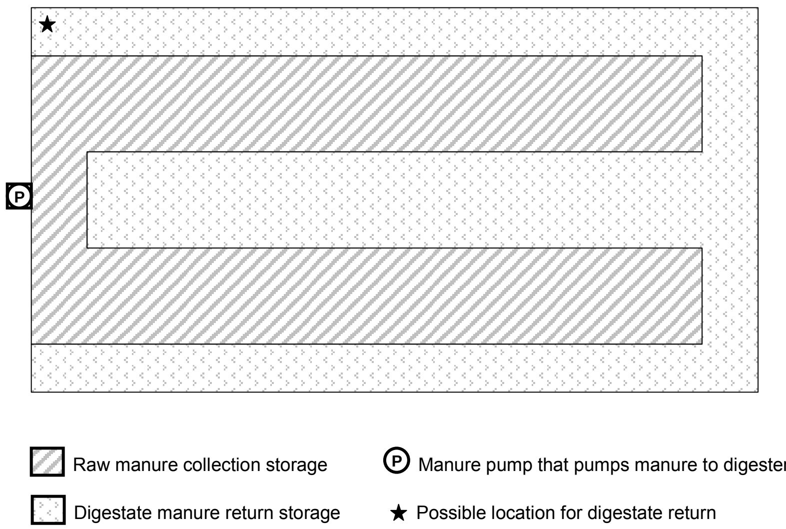


Figure 7.8. Suggested configuration for under-floor manure storage pit at a large-pen swine finisher building that utilizes dunging pattern to collect a majority of manure for direction to an anaerobic digester. The suggested manure pit configuration would eliminate the piping that allowed movement of manure between manure pits of the same manure group (Figure 7.3).

CONCLUSIONS

The novel manure collection system at this innovative farm collected 75% of the total manure from below finishing swine dunging areas and allowed an anaerobic digester to operate without the need for the development of a separate post-digestion manure storage facility. Analyses of influent and effluent manure indicated a reduction in manure percentages of solids and carbon, of which 59 and 63% of the removal was accounted for in biogas diverted to the engine, respectively. The remaining solids and carbon likely remained in the digester as settled or floating solids. Digestate contained lower concentrations of total N and organic N, while NH_3-N and nitrate-N did not increase significantly. It is likely that some N remained in the digester in solid form. The content of sulfur also decreased. Some sulfur may have remained in the digester in solid form and some may have been lost in the form of H_2S in biogas. Biogas content consisted of about 28% CO_2 and 72% CH_4. The gas was utilized to produce 170 140 kWh of electricity over a 24-month steady-state period. The electricity had a value of $11 450 and supplied 50.3% of the electric demand for the swine buildings and manure treatment system.

When the economic and electric production factors from this farm were applied to various alternative digestion systems the current system provided the most favorable economic feasibility. While the system design improved the economic feasibility at this commercial swine farm, the economic benefits associated with sale or cost-savings of electric production did not justify the costs of implementing a digestion system without external grant funding. In order for farm-level energy production from manure to become widely adapted the economics of system implementation must change. Several factors could help with this including: 1) external funding of digester construction through public or private grants, 2) increased purchase rates for farm-level electric production, in which case the difference in payments provided to the farm from the utility will likely be distributed somehow to other consumers of the utility, 3) advances in energy and environmental policy and credit markets which favor small-scale energy production and limit barriers to connectivity, 4) improved technology and increased digester system efficiencies, and 5) increased energy costs that may make the economics of digester implementation more attractive.Finally, benefits of anaerobic manure digestion beyond energy production may prove to be economic drivers of system implementation. For instance, in Pennsylvania, regulations are under development that will require new or expanding livestock facilities, which meet certain animal density definitions when populated, to have a state approved Odor Management Plan in place prior to stocking the facility. One of the proposed odor management tools that producers can adopt in adherance to odor requirements is anaerobic digestion (PA SCC 2007).

The introduction of digestate to manure storage areas under the swine living space did not cause levels of H_2S or Lower Explosive Limit gas to rise to concerning concentrations. The mean hourly concentration levels of NH_3 and H_2S at pig level each significantly decreased after ($P<0.05$) digestate was introduced to under-floor pits and ventilation remained unchanged. When digestate was present the removal of pit fan ventilation increased ($P<0.05$) the hourly pig level concentrations of NH_3, H_2S, and LEL. The level of NH_3 (37 ppm) was above recommended exposure levels. Gas monitoring was conducted at the location considered to be the most likely to attain negative air quality due to introduction of digestate to under-floor storage space. Air near the sample locations was close to exhaust fans and likely had no effect on pigs housed in other locations of the buildings. During times when minimum ventilation was provided by an end-wall fan on a timer, no air was exhausted along the length of these tunnel ventilated buildings, therefore background gas concentrations were likely higher when they reached the monitoring location. Both the manure storages and urine and feces on top of slats will emit NH_3 in swine houses (Aarnink *et al.* 1996).

It is unclear whether gas concentration levels had negative impacts on pig performance or mortality. Comparison of finishing swine performance at the study farm and at two conventional finishing farms revealed numerically poorer measures of performance, although the only statistical difference detected was between the case farm and Farm 2. It can be safely stated that no large obvious negative etiological effects from gases were observed in these houses by the producer or swine integration staff. Gas concentrations and animal performance should be viewed with caution as many factors likely influenced their results beyond those reviewed in this observational study.

IMPLICATIONS

The novel system concept applied to manure treatment at this farm demonstrates promise. Compared to traditional manure digestion systems the elimination of post-digestion manure storage improved the economic practicability of manure digestion at this location. External grant funding and a low interest loan were both necessary for economic feasibility. More work is warranted concerning possible consequences of changing in-house gas levels and animal performance. As environmental and social demands of livestock producers increase this system design, or components of this design, may prove beneficial in extensive adoption of anaerobic manure digestion systems.

REFERENCES

AARNINK, A. J. A., VAN DEN BERG, A. J., KEEN, A., HOCKSMA, P., & VERSTEGEN, M. W. A. (1996). Effect of slatted floor area on ammonia emission and on the excretory and lying behaviour of growing pigs. *Journal of Agricultural Engineering Research* 64, 299-310.

ANDERSON, G. A., SMITH, R. J., BUNDY, D. S. & HAMMOND, E. G. (1987). Model to predict gaseous contaminants in swine confinement buildings. *Journal of Agricultural Engineering Research* 37, 235-253.

ANEJA, V. P., BUNTON, B., WALKER, J. T. & MALIK, B. P. (2001). Measurement and analysis of atmospheric ammonia emissions from anaerobic lagoons. *Atmospheric Environment* 35, 1949-1958.

ATSDR (2007). *Ammonia (NH_3) CAS 7664-41-7; UN 2672 (between 12% and 44% solution), UN 2073 (>44% solution), UN 1005 (anhydrous gas or >50% solution).* United States Department of Health and Human Services, Agency for Toxic Substances and Disease Registry.

BONMATÍ, A. & FLOTATS, X. (2003). Air stripping of ammonia from pig slurry: characterization and feasibility as a pre- or post-treatment to mesophilic anaerobic digestion. *Waste Management* 23, 261-272.

BÖRJESSON, P. & BERGLUND, M. (2003). *Environmental analysis of biogas systems. Report nr. 45.* Department of Environmental and Energy Systems Studies, Lund University, Lund Sweden, (Report in Swedish with English Summary).

BÖRJESSON, P. & BERGLUND, M. (2006). Environmental systems analysis of biogas systems--Part I: Fuel-cycle emissions. *Biomass and Bioenergy* 30, 496-485.

BÖRJESSON, P. & BERGLUND, M. (2007). Environmental systems analysis of biogas systems--Part II: The environmental impact of replacing various reference systems. *Biomass and Bioenergy* 31(5), 326-344.

BRABER, K. (1995). Anaerobic digestion of municipal solid waste: a modern waste disposal option on the verge of breakthrough. *Biomass Bioenergy* 9, 365-376.

BRUBAKER, V. 2008. Personal communication. Manada Construction Company, Meyerstown, Pennsylvania, USA.

CURTIS, S. E., ANDERSON, C. R., SIMON, J., JENSEN, A. H., DAY, D. L. & KELLY, K. W. (1975). Effects of aerial ammonia, hydrogen sulfide and swine-house dust on rate of gain and respiratory tract structure in swine. *Journal of Animal Science* 41, 735-739.

DO BOER, S. & MORRISON, W.D. (1988). *The effects of the quality of the environment in livestock buildings on the productivity of swine and safety of humans. A literature review*. Department of Animal and Poultry Science, University of Guelph, Guelph, Ontario, Canada.

DONHAM, K., HAGLIND, P., PETERSON, Y., RYLANDER, R. & BELIN, L. (1989). Environmental and health studies of farm workers in Swedish confinement buildings. *Journal of Industrial Medicine* 46, 31-37.

DRUMMOND, J. G., CURTIS, S. E., SIMON, J. & NORTON, H. W. (1980). Effects of aerial ammonia on growth and health of young pigs. *Journal of Animal Science* 50(6), 1085-1091.

FISCHER, J. R., IANOTTI, E. L. & PORTER, J. H. (1984). Anaerobic digestion of swine manure at various influent solids concentrations. *Agricultural Wastes* 11, 157-166.
FRITSCHEN, R. D. (1975). *Toilet Training of Pigs On Partly Slatted Floors*. University of Nebraska, Lincoln, NebGuide G 74-40.
GRANDIN, T. (1980). Livestock behavior as related to handling facilities design. *International Journal for the Study of Animal Problems* 1, 33-52.
HANSEN, K. H., ANGELIDAKI, I. & AHRING, B. K. (1998). Anaerobic digestion of swine manure inhibition by ammonia. *Water Resource* 32(1), 5-12.
HILL, D. T., HOLMBERG, R. D. & BOLTE, J. P. (1985). Operating and performance characteristics of scraped swine manure as a thermophilic anaerobic digestion substrate. *Agricultural Wastes* 14, 37-49.
IPCC (2001). *Climate change 2001: the scientific basis. Contribution of Working Group I to the Third Assessment Report on the Intergovernmental Panel on Climate Change.* Cambridge University Press, Cambridge, UK.
JONES, J. B., BURGESS, L. R., WEBSTER, A. J. F. & WATHES, C. M. (1996). Behavioural responses of pigs to atmospheric ammonia in a chronic choice test. *Animal Science* 63, 437-445.
LANTZ, M., SVENSSON, M., BJÖRNSSON, L. & BÖRJESSON, P. (2007). The prospects for an expansion of biogas systems in Sweden – incentives, barriers and potentials. *Energy Policy* 35, 1830-1843.
LATSHAW, W. H. (2008). Personal communication. Assistant State Conservation Engineer, United State Department of Agriculture, National Resources Conservation Services, Harrisburg, Pennsylvania, USA.
LIAO, C. -M. (1996). Model to predict human and pig exposure to gaseous contaminants emitted from stored manure. *Applied Mathematical Modelling* 20, 731-740.
MUEHLING, A. J. (1969). Gases and odors from stored swine waste. *Journal of Animal Science* 30, 526-531.
MUCK, R. E. & RICHARDS, B. K. (1983). Losses of manorial nitrogen in free-stall barns. *Agricultural Wastes* 7, 65-79.
NI, J. Q., HEBER, A. J., DIEHL, C. A. & LIM, T. T. (2000). Ammonia, hydrogen sulphide and carbon dioxide release from pig manure in under-floor deep pits. *Journal of Agricultural Engineering Research* 77(1), 53-66.
NIOSH (1990). *NIOSH Publication No. 90-103: Preventing deaths of farm workers in manure pits*. US National Institute for Occupational Safety and Health, May 1990.
NIOSH (2005). *NIOSH Document: Pocket Guide to Chemical Hazards: Ammonia (2005-149)*. US National Institute for Occupational Safety and Health.
OLESEN, J. E. & SOMMER, S. G. (1993). Modelling effects of wind speed and surface cover on ammonia volatilization from stored pig slurry. *Atmospheric Environment. Part A. General Topics* 27(16), 2567-2574.
OSHA (1999). *VI. Health effects discussion and determination of final PEL. In: OSHA Preambles – Air Contaminants (29 CFR 1910.1000).* Occupational Safety and Health Administration, U.S. Department of Labor.
PAGILLA, K. R., KIM, H. & CHEUNBARN, T. (2000). Aerobic thermophilic and anaerobic mesophilic treatment of swine waste. *Water Resource* 34(10), 2747-2753.

PA SCC (Pennsylvania State Conservation Commission) (2007). *Proposed Rulemaking; State Conservation Commission [25 PA. CODE CH. 83], Facility Odor Management [37 Pa.B. 4780].* The Pennsylvania Bulletin, September 1, 2007.

PENN STATE AGRONOMY GUIDE (2006). Average daily production and total nutrient content of manure. The Pennsylvania State University, College of Agricultural Sciences. Agronomy Guide 2007-2008, 37.

PETERS, J., COMBS, S.M., HOSKINS, B., JARMAN, J., KOVAR, J.L., WATSON, M.E., WOLF, A.M. & WOLF, N. (2003). *Recommended Methods of Manure Analysis.* Board of Regents of the University of Wisconsin. Cooperative Extension Publishing. Madison, WI, USA.

PETERSEN, S. (1999). Nitrous oxide emissions from manure and inorganic fertilsers applied to spring barley. *Journal of Environmental Quality* 28, 744-751.

ROBERTSON, A. M. & GALBRAITH, H. (1971). Effects of ventilation on the gas concentration in a part-slatted piggery. *Farm Buildings Research and Development Studies* 1, 17-28.

ROOS, K. F., MARTIN, JR, J. H. & MOSER, M. A. (2004). *AgSTAR Handbook; A manual for developing biogas systems at commercial farms in the United States; Second Edition.* U.S. Environmental Protection Agency. EPA-430-B-97-015.

SAFLEY, JR., L.M., & WESTERMAN, P. W. (1994). Low-temperature digestion of dairy and swine manure. *Bioresource Technology* 47, 165-171.

SALMINEN, E. & RINTALA, J. (2002). Anaerobic digestion of organic solid poultry slaughterhouse waste – a review. *Biosource Technology* 83, 13-26.

SALMINEN, E., RINTALA, J., HÄRKÖNEN, J., KUITUNEN, M., HÖGMANDER, H. & OIKARI, A. (2001). Anaerobically digested solid poultry slaughterhouse wastes to be used as fertilizer on agricultural soil. *Bioresource Technology* 78, 81-88.

SHIH, J. C. H. (1987). Ecological benefits of anaerobic digestion. *Poultry Science* 66, 946-950.

SHIH, J. C. H. (1993).Recent development in poultry waste digestion and feather utilization – a review. *Poultry Science* 72; 1617-1620.

SKARP, S. -U. (1975). Manure gases and air currents in livestock housing. In: *Proceedings, The 3rd International Symposium on Livestock Waste. ASAE Publication PROC-275*, pp. 363-365.

SOMMER, S., PETERSEN, S. & SÖRGARD, H. (2000). Greenhouse gas emission from stored livestock slurry. *Journal of Environmental Quality* 28, 1610-1618.

SOMMER, S. G., MØLLER, H. B. & PETERSEN, S. O. (2001). *Reduktion af drivhusgasemission fra gylle og organisk affald ved biogasbehandling [The reduction of greenhouse gases from manure and organic waste using digestion and biogas production]. DJF-rapport nr 31*, Tjele, Denmark: Husdyrbrug, Danish Institute of Agricultural Sciences (in Danish).

STOLBA, A. & WOOD-GUSH, D. G. M. (1989). The behavior of pigs in a seminatural environment. *Animal Production* 48, 419-425.

STOMBAUGH, D. P., TEAGUE, H. S. & ROLLER, W. L. (1969). Effects of atmospheric ammonia on the pig. *Journal of Animal Science* 28, 844-847.

VERMEULEN, J., HUYSMANS, A., CRESPO, M., VAN LIERDE, A., DE RYCKE, A. & VERSTRAETE, A. (1992). Processing of biowaste by anaerobic composting to

plant growth substrates. In: *Proceedings of the International Symposium on Anaerobic Digestion of Solid Waste, Venice (14–17 April, 1992)*, pp. 147–157.
WATHES, C. M., JONES, C. D. R. & WEBSTER, A. J. F. (1983). Ventilation, air hygiene and animal health. *The Veterinary Record*, 113: 554-555.
WIESE, J. & HAECK, M. (2006). Instrumentation, control and automation for full-scale manure-based biogas systems. *Water Science and Technology* 54(9), 1-8.
ZHANG, R. H., DAY, D. L., CHRISTIANSON, L. L. & JEPSON, W. P. (1994). A computer model for predicting ammonia release rates from swine manure pits. *Journal of Agricultural Engineering Research* 58, 223-229.