7 Sustainable Manure Management

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Introduction

Manure management has risen to the fore of livestock production concerns in the developing world, with implications to farming system design, function and profitability, as well as to environmental quality and human health. Sustainable manure management, i.e. manure management that balances the production, economic and environmental concerns of manure generation, handling, processing and end use, requires concerted investment of resources and time as well as a strategic approach to livestock production that often extends well beyond the farm gate. This chapter seeks to elucidate the concerns underlying manure management and the considerations of sustainable manure management. While we provide a broad review of salient issues, particular emphasis is placed upon emerging areas of concern and innovation in manure management.

To appreciate the challenges of manure management in modern livestock operations, it is instructive to consider the evolution of livestock production from traditional, diversified systems with a high degree of integration between feed and livestock production, to modern, specialized systems in which feed and livestock production are sometimes completely separate. The advent of commercial fertilizers largely replaced the role of livestock manure as a primary source of nutrients for crop production. Networks for transporting feed and forages have enabled the specialization and intensification of animal production, with the number of animals used for production of meat, milk and eggs increasing from ~10 billion in 1930 to 68 billion in 2002 (FAOSTAT, 2012). By 2000, it was estimated that livestock were generating 101 and 17 Tg of manure nitrogen (N) and phosphorus (P), respectively, per year (Bouwman *et al.*, 2009). While the quantity of manure nutrients generated over this period increased, the growing disintegration of livestock and crop production has left most modern livestock producers with a surplus of on-farm nutrients.

Even greater concentrations of manure nutrients are apparent at regional scales (e.g. Maguire *et al.*, 2005). Potter *et al.* (2010) determined the geographic distribution of manure N and P produced from the main livestock groups (cattle, buffalo, goats, sheep, pigs and poultry) and reported that the highest rates of N and P in manures produced are found in the USA, parts of South America, western Europe, East Africa, northern India, eastern China and New Zealand. Regions with concentrated animal production that also use fertilizers in crop production have

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high levels of excess N, P and potassium (K). In Europe, de Walle and Sevenster (1988) determined N balances for 11 EU countries and reported that all countries had a surplus of N (22–320 kg ha^{-1}) with five having a surplus greater than 100 kg ha^{-1}. A national nutrient balance for Korea, in 1995, showed that excess nutrients were in the order of 331, 386 and 406 thousand kg for N, P and K, respectively (Richard and Choi, 1999). In the USA, the number of confined livestock farms declined 50% from 1982 to 1997, while the number of confined animal units increased 10% (Gollehon *et al.*, 2001). This shift in production led to more excess on-farm nutrients (734,000 t N and 462,000 t P in 1997) and the separation of land from livestock. Approximately 20% of farm-level excess N and 23% of farm excess P exceeded the land assimilative capacity at the county level. To solve these nutrient distribution problems, substantial amounts of manure may need to be transported off farm and in some cases out of the region in which they are produced.

For many livestock producers, manure has become one of the most problematic areas of management under a growing litany of off-farm concerns. Concerns over long-term self-sufficiency related to inefficient recycling of manure nutrients are mounting. More acute are nuisance, environmental and health concerns associated with environmental emissions and runoff. Both large and small operations face these issues. Current major strategies for sustainable manure management include: (i) on-farm; and (ii) off-farm recycling of nutrients for crop production; (iii) recycling of nutrients as an animal feed ingredient; and (iv) export to non-agricultural uses (energy production, fibreboard, plastics).

Sustainable manure management implies more than improved use of manure nutrients in crop production, ranging from the effective use of the energy potential associated with manures, to the minimization of off-site transport of potential contaminants. How manure is handled, stored and land-applied affect its quality, including nutrient value and potential to pollute the environment. Manure treatment can also play a central role in changing manure quality to enhance its beneficial properties and minimize its potentially adverse properties. Therefore choosing the appropriate manure management system is essential in order to continue to have large-scale animal production that balances food production priorities with negative environmental and social consequences.

On-farm Manure Management Systems

Manure collection and handling systems enable livestock producers to utilize all the components in their manure management system efficiently. Unconfined livestock operations, i.e. grazing operations where dung is excreted on pastures or rangeland, are not included in this discussion. A typical manure management system will include some or all of the following components: (i) area where manure is produced (i.e. feedlot/dry lots, barns, other confinement buildings); (ii) manure treatment area including recycling of useable manure by-products (i.e. solids separator, digester, composting); (iii) manure transport (i.e. transfer of manure from collection to storage or treatments areas); and (iv) manure storage facility (i.e. manure tank, holding pond, stackhouse or other storage area). The purpose of manure collection and handling systems is to gather and move manure among the components of a manure management system efficiently and safely. This system can also incorporate technologies to fractionate the manure to improve its utilization and derive more value from it.

The type of equipment and procedures used to collect and handle manure depends primarily on the solids content of the manure, with the quantity of solids in manure varying with species and production system. Classification of manures on the basis of solids content varies, but most conventional definitions classify 'solid manures' as those with greater than 20% solids, including 'litters', which are predominantly generated by poultry operations and tend to contain greater than 70% solids, 'semi-solid manures' that contain 12–20% solids, 'slurries' that contain 4–12% solids and 'liquid manures' that contain less than 4% solids. The solids content of excreted manure is often changed by processes such as adding bedding, drying manure on a lot surface, adding washwater or dewatering the manure by solids separation.

Solid and semi-solid manures are usually collected using scrapers, box scrapers, blades, front-end or skid-steer loaders, or similar devices.

These manures are typically transported by truck and directly land applied, stacked for storage, or composted. Slurry manure is typically generated where little bedding is added to the excreted manure/urine. The simplest manure collection arrangement is slotted or perforated flooring over a collection tank. Slurry manure can also be collected using scrapers or vacuums. Slurry is usually pumped and stored or treated, in some cases it is directly land applied. Liquid manures generally result from the addition of washwater or rainwater to manure. Flush systems are common where manure is flushed from the confinement building using either fresh or recycled water. Runoff from lot surfaces can be treated and stored in holding ponds. In most cases, the liquid is blended with clean water and used as irrigation water.

The characteristics of the manures collected in these systems will vary with species and production system and determine the nutrient value of the manure, the potential for gaseous emissions, the potential for energy generation, and the extent of treatment/processing needed to transform the manure into value added products. Tables 7.1 and 7.2 provide some manure characteristics as excreted by livestock and poultry, while Table 7.3 provides 'as removed' manure characteristics for the main livestock and poultry groups. It is important to remember that these values can change greatly based on animal feeding (see Chapter 6, this volume) and manure handling and storage. As demonstrated in these tables, manures contain valuable nutrients that can be recycled in the crop–animal system and solids that can be converted to energy and other valuable manure by-products.

Land Application of Manures for Crop Production

Land application of manure to support on- and off-farm crop production is common worldwide and a fundamental attribute of sustainable manure management systems. The positive contributions of manure to soil fertility and soil tilth are well established (e.g. Williams and Cook, 1961; Bationo and Mokwunye, 1991). Crop response to recent application of manure is generally positive, with yields stimulated by macro- and micronutrients in manure. Over the longer term, manure can substantially augment soil organic matter and soil structural properties that stabilize aggregates, increase water-holding capacity and improve rainfall infiltration. Despite these benefits, land application of manure can result in adverse impacts, many of little immediate concern to crop production, which complicate land application and must be considered in devising sustainable manure management strategies. Westerman and Bicudo (2005) listed eight challenges for integration of manures into agricultural production, which included:

- Regional imbalances of nutrients (e.g. not enough land on the farm to apply nutrients produced by animals on the farm);
- Imbalances of nutrients in manure compared with crop needs;
- The relatively low nutrient concentration compared with chemical fertilizers;
- The variability in nutrient content, difficulty in quickly determining nutrient content, and predicting the availability of nutrients to growing crops;
- The often bulky nature of manures making it more difficult to haul and spread consistently;
- Possible transfer of weed seed;
- Satisfying environmental regulations on application amounts, application timing, and application methods; and
- Possible environmental concerns, such as emission of ammonia (NH_3) and greenhouse gases (GHG), odour, pathogens and pharmaceutically active compounds (PAC).

Given these concerns and the growing regulatory and paperwork burden of land-applying manure in many areas of the developed world, manure's value can readily turn from an agronomic resource to a perceived liability (Kleinman *et al.*, 2012). Manure's bulky nature, heterogeneous nutrient forms, concentrations and ratios and adverse qualities (e.g. weed seed, odour, pathogens) make manures imperfect fertilizer substitutes. As a result, pound for pound, manure nutrients are more costly to transport than commercial fertilizers, often making the economic value of manure greatest near the point of generation, i.e. the livestock barn.

Land application of manure for crop production involves an array of potential environmental and human health concerns, the latter of which are discussed in greater detail below. Sustainable

Table 7.1. Estimated typical manure characteristics as excreted by meat producing livestock and poultry. (Source: ASABE Standard D384.2.)

Animal type and production grouping	Total manure (kg per finished animal)	Moisture (% w.b.)	Total solids	Volatile solids	COD	BOD	N	P	K	Assumed finishing time period (days)
			(kg per finished animal)							
Beef										
Finishing cattle	4500	92	360	290	300	67	25	3.3	17.1	153
Swine										
Nursery pig (12.5 kg)	48	90	4.8	4.0	4.4	1.5	0.41	0.068	0.16	36
Grow-finish (70 kg)	560	90	56	45	47	17	4.7	0.76	2.0	120
Poultry										
Broiler	4.9	74	1.3	0.95	1.05	0.30	0.053	0.016	0.031	48
Turkey (male)	36	74	9.2	7.4	8.5	2.4	0.55	0.16	0.26	133
Turkey (females)	17	74	4.4	3.5	4.0	1.1	0.26	0.074	0.11	105
Duck	6.5	74	1.7	1.0	1.4	0.28	0.062	0.022	0.031	39

COD, chemical oxygen demand; BOD, biochemical oxygen demand.

Table 7.2. Estimated typical manure characteristics as excreted by all other livestock and poultry. (Source: ASABE Standard D384.2.)

Animal type and production grouping	Total manure (kg per day per animal)	Moisture (% w.b.)	Total solids	Volatile solids	COD	BOD	N	P	K
			(kg per day per animal)						
Beef									
Cow (confinement)	–	88	6.6	5.9	6.2	1.4	0.19	0.044	0.14
Growing calf (confinement)	22	88	2.7	2.3	2.3	0.52	0.13	0.025	0.085
Dairy									
Lactating cow	68	87	8.9	7.5	8.1	1.30	0.45	0.078	0.103
Dry cow	38	87	4.9	4.2	4.4	0.626	0.23	0.03	0.148
Heifer – 440 kg	22	83	3.7	3.2	3.4	0.54	0.12	0.020	–
Swine									
Gestating sow – 200 kg	5.0	90	0.50	0.45	0.47	0.17	0.032	0.009	0.022
Lactating sow – 192 kg	12	90	1.2	1.0	1.1	0.38	0.085	0.025	0.053
Boar – 200 kg	3.8	90	0.38	0.34	0.27	0.13	0.028	0.0097	0.0176
Poultry									
Layer	0.088	75	0.022	0.016	0.018	0.005	0.0016	0.00048	0.00058

COD, chemical oxygen demand; BOD, biochemical oxygen demand.

Table 7.3. As removed characteristics by all other livestock and poultry. (Source: ASABE Standard D384.2.)

	Moisture	TS	VS	TKN	TAN	P	K	Ca	Na	Mg	S	Zn	Mn	Cu
	(% w.b.)		(%TS)	(% w.b.)					(ppm w.b.)					
Beef														
Earthen lot	33.1	67.2	30.2	1.180	0.10	0.50	1.25	1.21	3012	3650	2841	85	393	14
Dairy														
Scraped earthen lot	54	46	–	0.70	–	0.25	0.67	0.45	311	100	–	1.2	1.4	0.2
Scraped concrete lot	72	25	–	0.53	–	0.13	0.40	0.31	32	9	–	0.4	0.7	0.1
Lagoon effluent	98	2	52	0.073	0.08	0.016	0.11	0.04	7	3	–	0.9	1.4	2
Swine														
Finisher – slurry wet dry feeders	91	9	–	0.7	0.5	0.21	0.24	0.25	380	–	400	85	–	50
Slurry storage dry feeders	93.9	6.1	–	0.47	0.34	0.18	0.24	0.25	380	–	180	68	–	30
Flush building	98	2	–	0.2	0.14	0.07	0.17	0.04	300	290	155	33.6	14.4	31.2
Agitated solids & water	97.8	2.2	–	0.10	0.05	0.06	0.06	0.08	215	300	180	44.4	15.6	19.2
Lagoon surface water	99.6	0.40	–	0.06	0.04	0.02	0.07	0.01	215	55	37	3.6	1.2	2.4
Lagoon sludge	90	10	–	0.26	0.07	0.25	0.07	0.04	191	132	79	22	80	90
Poultry														
Broiler litter	31	70	70	3.37	0.75	0.60	1.37	1.82	–	–	–	–	–	–
Turkey litter	30	–	–	2.18	–	0.33	1.23	5.0	–	–	–	–	–	–

TS, total solids; VS, volatile solids; TKN, total Kjeldahl nitrogen; TAN, total ammoniacal N.

management of manures requires concerted effort to avoid the unintentional, adverse consequences of a seemingly prudent agronomic practice. In many areas of the developed world, land application of manure is regulated. Regulations range from international directives aimed at improving environmental quality (e.g. EU Nitrates Directive; European Commission, 2010) to local rules aimed at preventing nuisance complaints (e.g. municipal odour ordinances; SRF Consulting, 2004). Site selection and manure application method, rate and timing are the primary factors controlling these adverse impacts (Fig. 7.1).

Site selection

Site selection represents the first step in decision making when land-applying manures. The potential for nuisance concerns is often a primary site selection factor. In general, nuisance concerns (e.g. odour, flies) can be readily addressed by simply avoiding sites that are in proximity to or upwind of potential sources of offence (e.g. housing tracts). Site selection for manure application must weigh potential trade-offs (e.g. sites that conserve manure N may be prone to P loss in surface runoff) and reflect farming system concerns (e.g. field availability, crop requirement, manure storage and handling capabilities) for a particular physiographic context (site selection in sloping upland landscapes will feature different priorities than site selection in flat coastal plain landscapes). An array of nutrient management decision support tools have been developed over the past several decades and are now in widespread use across the USA, Europe and South Pacific. These tools generally consider local climate, site hydrology (e.g. seasonal high water table), soil properties (e.g. erodibility, leaching potential), field management (e.g. tillage system) and delivery factors (e.g. field buffers) as indicators of the potential

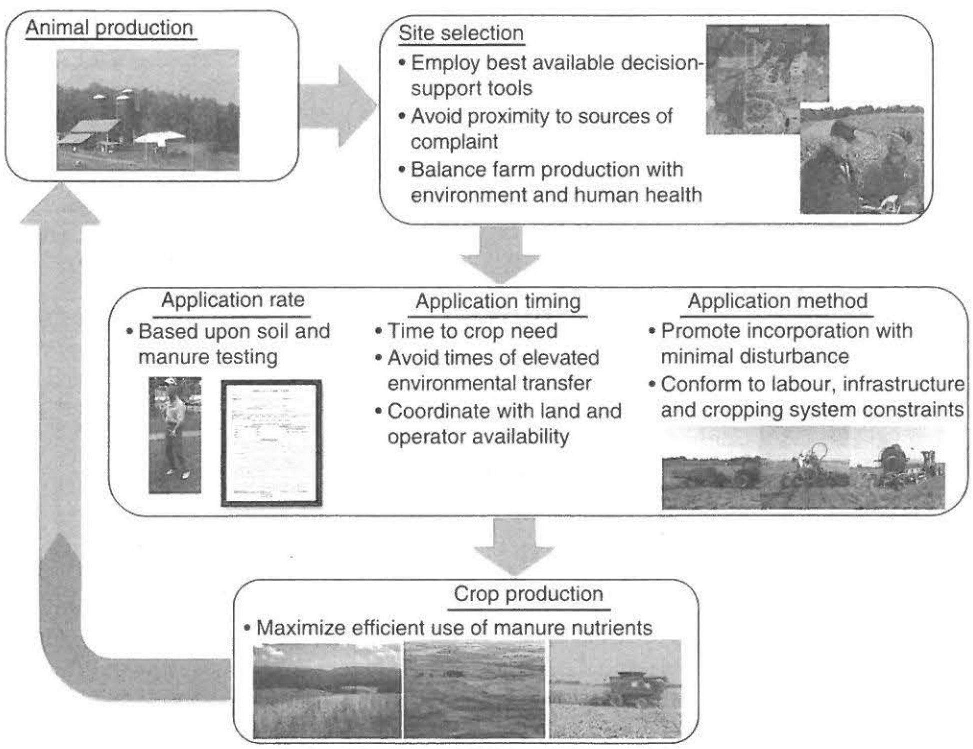

Fig. 7.1. Linking manure and crop production through the central components of a successful land application programme.

off-site transport of manure nutrients (Sharpley *et al.*, 2003; Delgado and Follett, 2011). Such tools offer strategic decision support for land application of manure. More recently, advances in the weather forecasting models have been used to develop prototype tools that provide daily decision support on whether sites are suitable to receive manure based upon pollution potential (Dahlke *et al.*, 2008; Melkonian *et al.*, 2008; Buda *et al.*, 2013).

Manure application methods

The fate of manure constituents and the impact of land application on crop, soil, air and water quality can be profoundly affected by manure application methods. Broadcast application of manure is the most ubiquitous application method of both liquid and dry manures alike. Much of the reported inefficiency in delivering manure nutrients to crops stems from the prevalence of broadcast application. Generally, broadcasting manures exacerbates emissions of NH_3 (Dell *et al.*, 2011; Pfluke *et al.*, 2011) and odour (Brandt *et al.*, 2011) and losses of P in runoff (Johnson *et al.*, 2011), and promotes severe vertical stratification of nutrients in soil. Incorporation of manure into soil at time of application generally improves nutrient use relative to broadcast application. However, incorporation has historically been accomplished by tillage, conflicting with no-till and perennial forage management systems. In recent years, an array of low disturbance technologies has emerged from Europe and the USA that place manure below the canopy of a perennial crop or into the subsoil with minimal disturbance to surface cover. Low disturbance technologies range from those that band manure below the canopy of perennial forages (trailing hose, trailing shoe) to those that inject manure into the soil (chisel injection, disc/coulter injection, high pressure injection) to those that lightly till or perforate the soil to improve infiltration of applied manure into the surface (aeration, low-disturbance vertical/strip tillage). Many configurations of these technologies exist, such that there are considerable options to adapt low-disturbance applicators to handle the specific requirements of existing machinery/manure handling systems, cropping systems and soil conditions. Barriers to the adoption of these specialized technologies often revolve around the capacity and speed of application relative to broadcast application and perceived cost. However, Rotz *et al.* (2011) demonstrated that the cost of adopting some low disturbance applicators on small dairy farms in the north-eastern USA was either neutral to slightly profitable in comparison with traditional broadcast methods. A comprehensive review of low-disturbance manure application technologies and their associated impacts on surface residue, nutrient loss, erosion, odour and crop response is offered by Maguire *et al.* (2011).

Poor accounting for manure nutrients often results in excess application of manures for crop production. *Manure application rate* is strongly tied to fugitive losses of nutrients to air and water over the short term (Thompson *et al.*, 1990; Kleinman and Sharpley, 2003). Over the long-term, repeated applications of manure in excess of crop requirement can overwhelm the soil's natural buffering capacity, resulting in chronic contributions that cannot be readily controlled with manure management options (Kleinman *et al.*, 2011). Sustainable manure management must consider realistic yield expectations and credit existing sources of nutrients (past manure application, recent legume crops). Testing of manure and soil are necessary to optimize manure application rates, although use of book values may represent a significant improvement over the status quo in many instances. Application rates should be adjusted to account for the significant differences in manure nutrient use efficiency of different application methods. For instance, immediate incorporation of manure may conserve more than 60% of the plant available N in manure that would be lost with broadcast application (Beegle, 2012).

Timing of manure application

Timing of manure application represents one of the most difficult logistical aspects of land application programmes for manure recycling. Application of manure shortly before crop planting is recommended to ensure that manure nutrients are employed by the growing crop. Long periods between manure application and crop growth lower the nutrient use efficiency as manure nutrients are removed by environmental processes. Avoiding manure application during periods of high potential for runoff or drainage is a key aspect of preventing losses of manure constituents to water (Walter *et al.*, 2001). Timing can also be used to optimize nutrient delivery; broadcast application immediately prior to a light rain can significantly translocate manure nutrients into the soil, providing many of the benefits of the incorporation techniques described above. However, a variety of factors prompt farmers to apply at other times, and these factors must be considered if manure application timing is to be improved. The absence of sufficient manure storage is most often cited as a cause of poor timing. Older, smaller confinement facilities sometimes lack the infrastructure to store manure temporarily and continue to rely upon daily application of manure, year round (Dou *et al.*, 2001). Even when adequate storage exists, severe weather conditions (e.g. extreme precipitation associated with hurricanes) may overwhelm open storage structures and drive farmers to land-apply manure when conditions are poor. Alternatively, site access may push farmers to land-apply during periods when crops have been recently harvested, soils are trafficable (waterlogged soils are frozen), or no other fields are available to receive manure. Ultimately, farming education, expanded options for land application of manure (including off farm export), well integrated farming systems and accurate, short-term decision support (e.g. the forecasting tools described above for site selection) are all key to ensuring prudent timing of manure application for crop production.

Manure as a Livestock Feed Ingredient

As manures contain valuable nutrients and trace minerals, one option for utilization of manures (normally collected in dry systems) is the recycling of nutrients as an animal feed and as a nutrient source in aquaculture. Smith and Wheeler (1979) reported that animal excreta products contain 48% to 73% total digestible nutrients and 20% to 31% crude protein and therefore the nutrient content of manure has been shown to be three to ten times more valuable as animal feed than as plant nutrients. Utilizing animal manures as feed nutrients has many benefits including decreases in potential pollution and feed costs and better utilization of essential mineral sources. Ruminants are particularly ideal for the feeding of manures due to rumen microbiology and their ability to utilize fibre, non-protein N and nucleic acids. The most valuable manure for protein supplement in feedstock is poultry litter, due to the high concentration of nutrients. When processed by an acceptable method, poultry litter is an economical and safe source of protein, minerals and energy for beef cattle and swine (Carter and Poore, 1995; Akinfala and Komolafe, 2011). The most common methods for processing animal manures for producing feed are: drying, composting, ensiling, deep stacking, chemical treatment and extrusion-pelletizing (Arndt *et al.*, 1979; Carter and Poore, 1995). If used as a feedstock, manures must be collected frequently to reduce losses of valuable N, as NH_3 volatilization from manures and litters happens quickly after excretion. In addition, the manure must be treated (composted, ensiled, chemical or heat treatment) to destroy pathogens and reduce odours to improve animal acceptability. It is also important to obtain accurate nutrient composition of manures if used as an animal feed as there is great variability in manure nutrient contents and they may differ significantly from published values (Zinn *et al.*, 1996).

Manure can also be used in aquaculture systems, not directly as a feed, but as a fertilizer to enhance algae and other aquatic plant growth, which then serves as a feedstock. In China, animal manures have traditionally been used as fertilizer for fishponds and integrated fish farming and livestock production is common (Edwards, 1980). One of the more common systems is integrated poultry–fish farming which combines poultry production with fish culture where the spilled feed and manure from the poultry system are inputs into the fish subsystem (Sinha, 1985). The recycling of nutrients in the system allows for intensification of production and reduction of the environmental impact of production (Costa-Pierce, 2002).

Despite compelling reasons for recycling manure as a feedstock in livestock and fish production, food safety (e.g. bovine spongiform encephalopathy) and animal welfare concerns are the basis for campaigns, and laws (Canada, EU), targeting the feeding of processed manures to livestock. Ultimately, public acceptance of this strategy will determine its contribution to sustainable manure management.

Manure Treatment Technologies and Non-agricultural Uses

In order to address some of the challenges related to use of manures for agricultural production and, in some cases, to produce more value added products from manures, there is a variety of manure treatment technologies available for on-farm use. Many of these manure treatment technologies have been used in one form or another for many years, while others are recent solutions. Often, technologies are used in combination to create a system that can be tailored to the manure management plan of the farm, climate or other factors (i.e. potential alternative end uses or by-product generation). Therefore, treatment technologies are generally selected to meet specific treatment goals on the farm. These treatment goals may include nutrient reduction or capture (primarily N and P), emissions reduction (including GHG, bioaerosols, NH_3, odours), volume reduction, energy recovery, pathogen reduction, and adding value to the manure. Manure treatment technologies are often linked together to address several treatment goals and challenges faced by animal producers such as excessive nutrients on farm, manure runoff and odour. Burton and Turner (2003) provide an excellent detailed discussion of many of these practices. Figure 7.2 presents a flow diagram of potential treatments and products that can be utilized in and derived from a manure management system. As some of the treatments/products are specific to solid versus liquid systems, they will be discussed separately below.

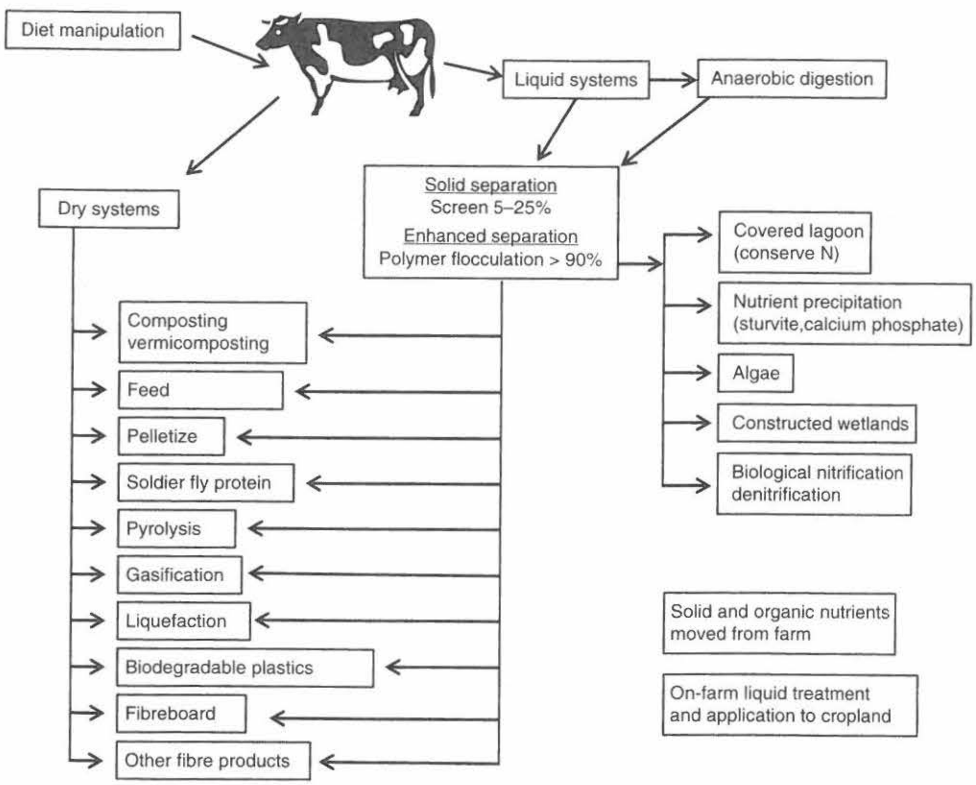

Fig. 7.2. Manure management treatments and technologies. Adapted from Szögi and Vanotti (2003).

Solid manure systems

Depending on the type of livestock raised and the production system, a large percentage of on-farm manure may be handled as a solid. Production systems that produce mainly solid manures are broiler and turkey operations, beef feedlots and dry-lot dairies. In addition, solid manures can enter the system via solid separation of slurries and liquid manures. As the moisture content in these manures tends to be low they are good substrates for composting, pelleting and for use in thermochemical conversion.

The need to move nutrients off farm has resulted in much interest in *composting* manures to reduce bulk, concentrate nutrients, reduce odour, kill pathogens and weed seeds, and have a stabilized product for transport (Westerman and Bicudo, 2005). In addition, composting manure has also been shown to degrade antibiotics effectively thereby reducing the potential for transport following land application (Kim *et al.*, 2012; Selvam *et al.*, 2012). The composted material is more uniform and easier to handle than raw manure, providing a source of slow release nutrients and therefore has commonly been used for years on many production facilities. There are several methods for composting manures: passive composting, aerated composting, windrow composting, in-vessel composting and vermicomposting. Passive composting is probably the most common method used today because it involves simply stacking manure (and other feedstock) and leaving them to compost over a long period. Very little, if any, activity is performed on the pile once it has been constructed. Aerated static pile composting modifies the passive composting technique by using blowers to supply air to the composting manure. This process does not involve turning and/or agitation of the piles. Electronic feedback controls are often used to monitor the pile temperature and control the operation of aerating blowers. Windrow composting is similar to passive composting although the piles of manure are turned or aerated by mechanical equipment to maintain optimum conditions. Manures are placed in long rows and are mechanically turned at frequent intervals in the composting process. In-vessel composting refers to any type of composting that takes place inside a structure, container or vessel. Each type of vessel system relies upon mechanical aeration and turning to enhance and decrease the duration of the composting process. All of these systems require a greater investment in manure management as manure/compost is moved several times and needs to be mechanically turned or aerated in some way; in some cases these costs can be prohibitive for on-farm adoption.

One of the major disadvantages of composting raw manures is the loss of N as NH_3 – a valuable nutrient for crop production and an air quality concern, as well as a loss of carbon (C), as carbon dioxide (CO_2) and methane (CH_4), which is a valuable soil conditioner. During the thermophilic phase (high temperature) of composting, much of the manure N is lost mainly as NH_3. Nitrogen losses can range from 3% to 60% of total initial N (Bernal *et al.*, 2009) with the majority of the N lost during the first 4 days of composting (Jiang *et al.*, 2011). The loss of organic matter or C has been shown to range from 9–81% of initial OM depending on the manure type and bulking agent used (Bernal *et al.*, 2009) with the greatest loss of C occurring later in the composting process (Jiang *et al.*, 2011). Countermeasures to reduce the loss of N during the composting process can improve the fertilizer value of composts, and several management options are available. The addition of a readily available C source such as molasses has been shown to reduce NH_3 losses as more N is stabilized in the microbial biomass (Liang *et al.*, 2006). Additives such as zeolite and biochar have been shown to reduce N losses by up to 52% (Steiner *et al.*, 2010; Luo *et al.*, 2011). Fukumoto *et al.* (2011) demonstrated that the use of struvite precipitation and nitration promotion in the composting process of swine manure reduced total N losses by 60%.

Vermicomposting is a process that relies on earthworms and microorganisms to help stabilize active organic materials and convert them to a valuable soil amendment and source of plant nutrients. As the process is mesophilic (moderate temperature) less N is lost during the process leaving a lower C:N ratio, which improves its value for agricultural uses (Lazcano *et al.*, 2008). Due to the lower composting temperature, manures do not undergo thermal stabilization that eliminates pathogens. Therefore, one potential drawback to the use of vermicomposting for treating animal manures is the presence of human pathogens, which could restrict the use of vermicompost as an organic fertilizer (Aira *et al.*, 2011). To circumvent this problem, thermophilic composting as a pretreatment to vermicomposting is also being used to reduce pathogens. Mupondi *et al.* (2011) reported that a pre-composting period of 1 week was found to be ideal for the effective vermicomposting of dairy manure.

In order to increase the bulk density, nutrient density and particle size uniformity of manures there has been interest in *pelletization*. By pelletizing manures, a more nutrient-dense product is available for transport, thereby enabling a larger land area to be utilized for land application (Hammac *et al.*, 2007). Pelletization of manures can be done with dry manures such as poultry litter or liquid manures with the addition of dry substances (Heinze, 1989). In 2001, the world's largest pelletization plant, Perdue-AgriRecycle Poultry Manure Pelletization Plant, was opened on the Delmarva Peninsula to process 95,000 t of manure a year. This was a joint effort between Perdue, one of the largest US poultry producers, and the State of Delaware to help address regional nutrient accumulation issues. The product produced in the pelletization process is shipped around the world for use as fertilizer and fish feed. In Ireland, technology was developed to blend composted biodegradable farm wastes such as pig manure, spent mushroom compost and poultry litter with dried blood or feather meal with mineral supplements, which was then pelletized to produce an organo-fertilizer with specific N:P:K target ratios that was pathogen free (Rao *et al.*, 2007). These designer organo-fertilizers are one way to add additional nutrient value to manure and increase their marketability.

Solid manures can also be used for *thermochemical conversion* to produce biogas. Technologies that burn manure to produce energy or treat manure to produce fuels are classified as 'thermochemical conversion', and include direct combustion (burning with excess air to produce heat), pyrolysis (thermal treatment in the absence of air, resulting in the production of pyrolysis oil and a low-BTU gas), gasification (thermal treatment at higher temperatures in an oxygen-restricted environment to produce a low- to medium-BTU gas) and hydrothermal liquefaction (thermal conversion of solids in a liquid stream to oils and char for separation and use as a fuel). The fuels that are products of pyrolysis and gasification can be used in boilers and engines. By-products from pyrolysis, such as biochar, are also being investigated as a soil amendment for C storage. Cost estimates for these technologies vary widely and do not always include the costs of pretreatment, drying and fuel preparation, or post-treatment, gas clean-up, electrical generation, and emissions controls. In general, it appears the value of the heat and energy alone does not provide sufficient financial incentive for a thermochemical conversion facility. Additional income streams that might make the technology more economically appealing do not currently exist but could include a combination of tipping fees collected for accepting manure solids, renewable power production tax incentives and the recovery of value from the ash. Liquid manures can also be used in thermochemical conversion to produce energy via direct liquefaction, aqueous-phase gasification and combined pyrolysis/gasification (Cantrell *et al.*, 2007).

Slurry and liquid manure systems

Livestock production facilities that house animals in confinement buildings typically generate large amounts of both slurry and liquid manure. In many cases, slurry and liquid manure undergoes some form of *solid–liquid separation* (removal of organic and inorganic matter) prior to storage. Objectives for removing solids include removal of nutrients for transport off-site, removal of larger particles to make liquid transfer more efficient, and removal of organic material to reduce volatile emissions. Separation efficiency depends on the particle size distribution in the influent, the characteristics of the treatment technology and the treatment time. Separation devices can utilize gravity flow, have few moving parts and require little management effort, or they can utilize pumps and motors and require intensive management. Mechanical separators include: stationary inclined screen; vibrating screen; rotating flighted cylinder; rotating cone; piston; liquid cyclone; and roller, belt, screw or filter presses. Gravity separators include settling basins, ponds and weeping walls.

Chemical flocculants have also been used in conjunction with solid–liquid separation systems in order to improve separation efficiencies. For example, solid separation using screens has low efficiency with solids removal rates of 5 to 15% (Vanotti and Hunt, 1999), whereas solid separation using screens with a flocculant agent can remove >90% of total and volatile solids and >70% of chemical oxygen demand (COD) and total N and greater than 50% of TP (García *et al.*, 2009; Paz Pérez-Sangrador *et al.*, 2012). Precipitation or flocculation in a treatment cell where the material can be harvested is beneficial. Although there are many separation techniques available for use on livestock farms, there is still a need to improve the cost effectiveness of these technologies. Once manure has gone through a solid separator, the remaining liquid is either transferred to a storage pond where it is kept until land application, or it can be further processed to produce recycled cleaner water, energy and other by-products.

Once the liquid fraction is transferred to a storage pond, basin or lagoon, there can be additional losses of N due to NH_3 volatilization as well as the generation of CH_4 and volatile organic compounds (VOCs; responsible for odour) due to the development of anaerobic conditions. One way to minimize these emissions is to utilize a *cover*. Guarino *et al.* (2006) tested several permeable covering systems (maize stalks, wood chips, vegetable oil, expanded clay, wheat straw) to reduce emissions from livestock slurry tanks and lagoons. They reported reductions of NH_3 emissions from swine and dairy slurry in the range of 60–100% with 140-mm solid covers or 9-mm liquid covers. Miner *et al.* (2003) reported that a permeable polyethylene foam lagoon cover reduced NH_3 emissions by approximately 80% on an anaerobic swine lagoon. Floating an impermeable cover over the surface of a lagoon or pond can also capture up to 80% of methane and reduce odours. The trapped gas can be flared or used to produce heat or electricity. Craggs *et al.* (2008) reported that placing a floating polypropylene cover on anaerobic swine and dairy ponds yielded biogas recoveries of 0.84 m^{-3} m^{-2} day^{-1} and 0.032 m^{-3} m^{-2} day^{-1}, respectively. They estimated that this could produce 1650 and 135 kWh day^{-1} from fully covered anaerobic swine and dairy ponds. Permeable surface covers not only act as a physical barrier to gas transport, but can also support microbial communities that are capable of utilizing reduced gases emitting from the slurry (Petersen and Miller, 2006) providing additional mitigation benefits.

When land for application of liquid manures is limited, treatment systems that remove N via *biological nitrification/denitrification* have been employed. Uncovered anaerobic lagoons have commonly been used to treat livestock wastewaters with the main treatment focusing on N volatilization and reduction of solids. While the goal is to reduce N by conversion to N_2 gas there is still a large amount of NH_3 volatilized into the atmosphere. These lagoons are also large sources of CH_4 as the solids are broken down in anaerobic conditions, and they have been identified as one of the major GHG emitting sources in livestock production systems (Leytem *et al.*, 2011). Systems that capture the CH_4 and use it for energy generation or flare the CH_4 help mitigate this impact (see below). The design of batch reactors to convert NH_4-N to N_2 has also been an area of research (Loughrin *et al.*, 2009; Wang *et al.*, 2010). Related to this is the anaerobic ammonium oxidation technology where ammonium (NH_4^+) is oxidized to N_2 and has been shown to remove up to 92% of NH_4-N from swine manure effluent (Molinuevo *et al.*, 2009). Another cost effective and passive method for treating wastewaters is the use of constructed wetlands which are primarily designed to remove N prior to land application through plant uptake and denitrification. These constructed wetlands also have the added benefit of reducing the total suspended solids (TSS), COD and P, which are important from a water quality standpoint. A marsh-pond-marsh constructed wetland was shown to remove up to 51% of TSS, 50% of COD, 51% total N and 26% total P from swine wastewater (Poach *et al.*, 2004). Constructed wetlands can also release N as NH_3, although this has been shown to be a relatively small portion of total N loss (Poach *et al.*, 2002). The main limiting factor for denitrification in these systems is the conversion of N to nitrate (Hunt *et al.*, 2006). While these management strategies can reduce the potential for water quality impairment from over-application of N, manure management strategies that are designed to waste valuable N are difficult to justify as replacing the lost N via chemical fertilization requires considerable expenditure of energy.

One of the more common energy production/capture systems for liquid manures is *anaerobic digestion*. Anaerobic digestion is a natural biological process by which bacteria break down organic matter in an oxygen-free environment with moisture content of 85% or higher. The process produces 'biogas', inorganic salts and residual organic material. The biogas consists of CH_4, CO_2 and trace amounts of other gases including hydrogen sulphide (H_2S). Biogas can be burned to produce heat or to power an electric generator. Zaks *et al.* (2011) estimated that anaerobic digesters have the potential to generate 5.5% of US electricity and mitigate 151 Mt CO_2e, mostly from CH_4 abatement. Rico *et al.* (2011) reported that anaerobic digestion of liquid dairy manure at 37°C produced 1.47 m^3 biogas (m^{-3} day^{-1}) and 1 m^3 CH_4 (m^{-3} day^{-1}) and could provide 2% of the total electrical power in the region of Cantabria, Spain. Marañón *et al.* (2011) reported that anaerobic digestion of cattle slurry on dairy farms in northern Spain could provide enough CH_4 to fulfil the farms' energy requirements and in some cases provide a surplus that could be used for heating and that annual GHG emissions savings ranged from 978 to 1776 kg carbon dioxide equivalents (CO_2e) per year due to reductions in CH_4 emissions during slurry storage. The amount of biogas produced and the percentage of residual organic matter depends on the duration of the anaerobic digestion process and factors such as temperature, moisture, nutrient content and pH. The residual organic material can be used for animal bedding, a soil amendment, or value added products such as fibreboards and other building materials. Additional benefits of anaerobic digestion are the breakdown of VOCs responsible for odour, and the destruction of weed seeds and pathogens. Digestion can occur in anaerobic lagoons or in engineered systems. The types of anaerobic digester technology available include: covered anaerobic lagoons, plug-flow digesters, completely-stirred tank reactor, upflow anaerobic sludge blanket and anaerobic sequencing batch reactor. Due to the large capital investment, initial set-up costs and expense of running a digester, they are not always economically feasible, particularly in areas with low energy prices. Yet, co-digestion of manure and other biomass is a potential way to improve the economics of digesters. The co-digestion of dairy manure and food processing wastes increased biogas production by 110% and tripled gross receipts on a commercial dairy (Frear *et al.*, 2011). The addition of vegetable waste in the anaerobic digestion of swine manure increased methane yield up to threefold (Molinuevo-Salces *et al.*, 2012).

Once manure has gone through a solid separation process or through a digester, the effluent can be treated to capture valuable nutrients in the liquid stream and concentrate them to generate a more valuable fertilizer source. A common technology to capture P, NH_4^+ and K is the use of *struvite* (magnesium ammonium phosphate) precipitation or P capture can be accomplished with hydroxylapatite (calcium phosphate) formation as well. Struvite precipitation has been found to remove 70–85% of P, 56–95% of NH_4^+ and <10% of K (Zeng and Li, 2006; Song *et al.*, 2011; Yilmazel and Demirer, 2011). However, a large amount of NH_4-N can be lost via volatilization (Song *et al.*, 2011). The use of hydrated limes to remove P from wastewaters has been shown to remove >90% of P (Vanotti *et al.*, 2003; Szögi and Vanotti, 2009).

The use of *algae* and other photo-bioreactors can also remove significant amounts of N and P, although these require light and the end-product may need significant processing before the nutrients can be re-used efficiently. The culturing of microalgae for biofuels production using wastewater as a nutrient source is also an area under investigation (Lam and Lee, 2012). Chen *et al.* (2012) demonstrated that non-filamentous green algae were able to tolerate high nutrient loads and could recover nutrients from wastewater from anaerobic digestion. It has been reported that up to 98% of N and 76% of P can be removed from wastewaters (Kebede-Westhead *et al.*, 2006; Chen *et al.*, 2012). Singh *et al.* (2011) reported a maximum biomass productivity of 76 mg l^{-1} day^{-1} for microalgae grown on poultry litter anaerobic digester effluent with a 60% and 80% removal rate of total N and P, respectively, from the effluent. The algae contained 39% protein, 22% carbohydrates and <10% lipids, making it a good animal feed supplement. The processed algae have also been tested as a slow release fertilizer (Mulbry *et al.*, 2005, 2007).

In addition to nutrient capture, other products can be obtained or made from solid separated materials, liquid effluent, digester effluent and post-digester solids. *Fibreboard* and building materials can be manufactured utilizing digested solids in place of sawdust (Winandy and Caia, 2008). *Seed pots* are being manufactured from manures (CowPots). Manures can be used to generate *granular active carbon*, which can be used for water treatment (Lima and Marshall, 2005). Extracted proteins and amino acids from manures can be utilized as feed ingredients; also *black soldier fly* prepupae meal, which are raised on manures, can be a valuable feed ingredient (Bondari and Sheppard, 1981; Sheppard *et al.*, 1994; Meyers *et al.*, 2008). Carbohydrate material from manure can be utilized to make *biodegradable plastics* and other products (ABCNEWS.com, 2001). While there are many potential value-added products that can be generated from manure, none of these are currently in mainstream production.

Barriers to Manure Treatment Technology Adoption

Although there are a great number of commercially available systems for manure treatment and fractionation including solid–liquid separation, generation of biogas, nutrient extraction and value added products, these are still not common practices on most livestock farms. There are two main driving forces that prevent wide scale adoption of many manure treatment and fractionation technologies. The biggest deterrent to the adoption of many technologies is that they are not economically viable. While the technologies may be available, they do not produce enough value to make them attractive to most producers. This is due in part to cheap energy prices, relatively low costs of fertilizers, and a lack of demand for the different products derived from manure fractionation. Another large barrier to technology adoption is the on-farm management of some of these technologies. Most producers are focused on managing livestock production and do not have the time or interest to operate and maintain other equipment on site. For many of these technologies to be successfully adopted and operated, third-party collaborators will have to be involved that will be willing to install, operate and maintain the equipment on site without interfering with animal/farm management. Both of these hurdles will need to be overcome in the future in order for these innovative technologies to become common on the average livestock farm. One other issue of concern, in some cases, is the issue of scale. Some technologies are proven on a very small scale and would need to be scaled up to make them economical on larger livestock operations. On the other hand, for some technologies such as biogas production, there may need to be a larger source of feedstocks for the technology to be cost effective, which would eliminate the possibility of utilizing these technologies on smaller farms or would require a central production site that received manure from surrounding farms.

Future Research Needs

While many technologies for deriving more value from manure exist there needs to be more research done to improve the efficiencies of these technologies and make them more cost effective for the average producer. Some examples of areas needing further consideration are listed below, although this list is not exhaustive.

- Improve the N retention in composting practices.
- Improve solid–liquid separation and make it more cost effective.
- Improve thermal conversion technologies to make them cost effective.
- Improve anaerobic digestion technologies to make them more cost effective.
- Improve upon nutrient extraction technologies to improve economics.
- Take proven technologies and scale them up to make them economical on farm.
- Evaluate existing and new technologies for their potential to generate environmental credits (sale of C offset and nutrient credits).
- Develop new technologies to capture and concentrate nutrients.

In addition to research, success of many of these technologies would be dependent on support from government agencies or local communities in order to make them economically viable.

Pathogens and Veterinary Antibiotics in Livestock Manures

While historical concerns related to livestock production have traditionally focused on nutrient pollution of waters and air quality, increased awareness of zoonotic pathogens and veterinary pharmaceuticals in animal manures is now recognized as a public health concern (Venglovsky *et al.*, 2009). Domesticated livestock, as well as wildlife, harbour a variety of bacterial, viral and protozoal pathogens, some of which are endemic and cannot easily be eradicated (Sobsey *et al.*, 2006). As a result, the pathogens can be found in fresh animal manures at production facilities and off farm when inadequately treated manures are used as soil conditioners and fertilizers (US EPA, 2005). Some pathogens commonly found in cattle, swine and poultry manures are *Campylobacter* spp., *Escherichia coli* O157:H7, *Salmonella* spp., hepatitis E virus, *Cryptosporidium parvum* and *Giardia lamblia* (Kraus *et al.*, 2003). The titre of these zoonotic pathogens can exceed thousands per gram of faeces, with infection causing temporary illness or mortality, especially in high-risk individuals (Hutchison *et al.*, 2005a; Klein *et al.*, 2010; Létourneau *et al.*, 2010). Exposure of humans to pathogens can occur through occupational exposures, ingestion of contaminated food and water, or aerogenic routes (Matthews, 2006; Dungan, 2010).

Use of antimicrobials in livestock production may also intensify the resistance of pathogens to antibiotics, reducing the ability to treat infected individuals (Boxall *et al.*, 2003; Bahe *et al.*, 2006). In Europe, the USA and other countries, antibiotics are used therapeutically (high doses) in livestock production to treat specific diseases or sub-therapeutically (low doses) by incorporating into feed to improve growth efficiency (Sarmah *et al.*, 2006). It is also common practice to administer multiple classes of antibiotics to livestock simultaneously at the production facility (Song *et al.*, 2007). Because not all antibiotics are absorbed in the gut of animals, they are excreted via urine and faeces in unaltered form and as metabolites (Halling-Soresen *et al.*, 1998; Boxall *et al.*, 2004). It has been estimated that as much as 80% of orally ingested antibiotic can be excreted in urine and faeces (Elmund *et al.*, 1971; Levy, 1992; Halling-Sørensen *et al.*, 2002). Several classes of veterinary pharmaceuticals and antibiotics, including coccidiostats, ionophores, lincosamides, macrolides, sulfonamides and tetracyclines, have been detected in surface waters adjacent to livestock operations (Campagnolo *et al.*, 2002; Hao *et al.*, 2006; Song *et al.*, 2007). In addition, the practice of land-applying livestock manure for its fertilizer value provides for the introduction of veterinary antibiotics (VAs) over large areas in the environment, which have been detected in soils and waters worldwide (Hamscher *et al.*, 2002; Christian *et al.*, 2003). For more detailed discussion on antibiotics in animal agriculture and other emerging issues, see Chapter 18, this volume.

Fate of pathogens in manure

Prior to land application, manures are generally stockpiled, stored in lagoons or pits, or anaerobically digested or composted, all of which can influence the ultimate survival of pathogens (Sobsey *et al.*, 2006; Ziemer *et al.*, 2010). In untreated liquid manures, pathogens may persist for a long time depending upon the storage conditions and temperature, type of slurry and pathogen type. In general, low temperature, optimal moisture and solids content and no aeration have been shown to enhance the survival of pathogens in manures (Jones, 1976; Kudva *et al.*, 1998; Venglovsky *et al.*, 2006). For example, *E. coli* O157, *Salmonella*, *Listeria* and *Campylobacter* were shown to survive for up to 6 months in dairy manure slurries (2% and 7% DM); however, in manure heaps (both turned and unturned) where temperatures were >55°C, the pathogens survived less than a few days (Nicholson *et al.*, 2005). In inoculated sheep manure, Kudva and co-workers (1998) found that *E. coli* O157:H7 survived for 21 months under varying climactic conditions when not aerated, but only 4 months when aerated, with the difference being attributed to drying during aeration. In sheep and cattle faeces (and cattle slurries), *E. coli* O157:H7 survived the longest without aeration at temperatures <23°C. In swine slurry, viable *Salmonella* spp. were recovered up to 300 days when stored at 4°C, while at 37°C none was detected after 7 days (Arrus *et al.*, 2006).

In contrast to bacteria, protozoan parasites (e.g. *Cryptosporidium* and *Giardia*) can survive in livestock manures for an extended period, which is likely due to their ability to form cysts and oocysts. The *Cryptosporidium* oocyst can resist die-off over a wide temperature range while remaining infective (Fayer *et al.*, 2000). In unstirred swine slurries spiked with *Cryptosporidium* oocysts, a 1-log reduction in oocysts was calculated to occur at 270 and 345 days during cooler and warmer temperatures, respectively (Hutchison *et al.*, 2005b). In contrast, the survival of *Giardia* cysts is highly temperature dependent. In a mixed human and swine manure, 90% of *Giardia lamblia* cysts were non-viable at 130 and 4 days when the respective incubation temperatures were 5° and 25°C (Deng and Cliver, 1992a).

Viruses are obligate intracellular parasites that are unable to replicate outside their host and, as a result, their numbers do not increase once released into the environment. A variety of physical, chemical and biological factors are responsible for the stability of viruses in animal manure management and treatment systems. Virus survival in manures is likely influenced most by temperature, pH (very high or low), NH_3, microbial activity, aggregation (virus clumping), encapsulation or embedding in membranes or particles, and indirectly through solids association (Deng and Cliver, 1995a; Sobsey *et al.*, 2006). In various animal manures, D_{90} values (time, in days, required for a 90% reduction of virus titre) ranged from <7 days for herpesvirus to more than 180 days for rotavirus (Pesaro *et al.*, 1995). With a bovine parvovirus and porcine enterovirus, D_{90} values in animal manures were 200–300 days when the viruses were kept at 5°C (Srivastava and Lund, 1980; Lund and Nissen, 1983). In mixed swine and human waste, poliovirus type 1 was more stable at 14°C than at 21°C, with respective D_{90} values of 52 and 19 days (Deng and Cliver, 1992b). When dairy manure was mixed with human waste, D_{90} values for hepatitis A virus at 5°C were 35 days compared with 8 and 7 days at 25° and 37°C, respectively (Deng and Cliver, 1995b). As with bacteria and protozoan parasites, livestock manures represent a potential viral hazard when applied to agricultural land without being treated (Sobsey *et al.*, 2006).

Fate of pathogens in soil

It has been demonstrated that pathogenic bacteria survive longer when manures are immediately incorporated into soils than when left on the surface for some time (Hutchison *et al.*, 2004). Pathogens on the surface may be exposed to UV irradiation, temperature fluctuations and desiccation that can potentially decrease their ability to survive compared with soil-incorporated pathogens. In soils, however, indigenous soil microorganisms have been shown to increase the inactivation rate of pathogens (Dowe *et al.*, 1997; Jiang *et al.*, 2002). The pH and temperature of soil also influences the survival of bacteria, which is limited by low soil pH and higher temperatures (Gerba *et al.*, 1975). Soil texture may also enhance the survival of pathogens, as *E. coli* O157 was reported to survive up to 2 months longer in loam and clay soils than in sandy soil (Fenlon *et al.*, 2000).

In sandy and clay loam soils amended with various manures, *Campylobacter, E. coli* O157, *Listeria* and *Salmonella* were found to survive as long as a month or longer (Nicholson *et al.*, 2005). Similarly, Stanley *et al.* (1998) detected *Campylobacter* for up to 20 days after the application of contaminated dairy slurry, while Jones (1986) reported survival times for *Salmonella* up to 259 days in soils amended with animal faeces. In addition, it was found that *Salmonella* may persist in soils for a longer period in a viable non-culturable state, thus avoiding detection via use of traditional culture-based techniques (Turpin *et al.*, 1993). *Listeria* are ubiquitous in the rhizosphere, making them well adapted to survive for extended periods in soils (Van Renterghem *et al.*, 1991). In dairy manure-amended soil, *Listeria monocytogenes* survived for up to 43 and 14 days when incubated at 5° and 21°C, respectively (Jiang *et al.*, 2004). Dowe *et al.* (1997) showed that chicken manure promoted better growth of *L. monocytogenes* than did liquid hog manure, but only when the competitive bacterial flora was reduced by autoclaving.

In soil, *Giardia* cysts were inactivated after incubation for 1 week at −4° and 25°C; however, cysts were recoverable from soils for 2 months when maintained at 4°C (Ziemer *et al.*, 2010). *Cryptosporidium* oocysts are more environmentally resistant and remained infective >3 months in soil at −4° and 4°C, but at 25°C degradation of the oocysts was accelerated and samples were infective for a shorter period (Olson *et al.*, 1999). While low and high temperatures definitively affect the survival and inactivation of oocysts, changes in soil moisture content had little or no effect on their inactivation (Jenkins *et al.*, 2002). In contrast, Kato *et al.* (2002) found that inactivation rates of oocysts were greater in dry soils than in moist and wet soils that were subjected to freeze–thaw cycles. Jenkins *et al.* (2002) also reported that soil texture may influence the inactivation of oocysts, but it could not be ruled out if this effect was related to soil pH. However, in a field study later conducted by Kato *et al.* (2004), oocyst inactivation could not be correlated with soil pH, moisture and organic matter content.

Removal of viruses in soils occurs largely by adsorption, with viruses surviving about as long as pathogenic bacteria (Gerba *et al.*, 1975; Gilbert *et al.*, 1976; Sobsey *et al.*, 1980). Hurst *et al.* (1980) found that the survival of enterovirus, rotavirus and bacteriophage in amended soils was influenced by temperature, soil moisture, presence of aerobic microorganisms, degree of adsorption, level of extractable P, exchangeable aluminium and soil pH. Overall, however, adsorption and temperature had the greatest effect on virus survival in soil, with virus survival decreasing with increasing temperature. At 37°C, no enterovirus infectivity was recovered from soil after 12 days, but at 4°C the virus persisted for at least 180 days (Yeager and O'Brien, 1979). Due to the adsorption of the virus by soils and influence of temperature on their survival, the land application of sewage effluent during warm and dry months has been documented as a viable disposal option to minimize the off-site transport and survival of viruses (Bitton *et al.*, 1984; Straub *et al.*, 1993).

Transport of pathogens in soil

Application of livestock manures on soils, particularly surface application of manure, can result in the transport of manure pathogens to surface or ground waters (Abu-Ashour *et al.*, 1994; Jamieson *et al.*, 2002; Tyrrel and Quinton, 2003). The overland transport of microorganisms is also called horizontal movement, while the leaching of microorganisms through soil and other porous subsurface strata is referred to as vertical movement. Unless a soil is saturated or contains an impermeable barrier, vertical movement of microorganisms will occur (Mawdsley *et al.*, 1995).

Despite the existence of bacterial, viral and protozoal pathogens in manures, few studies to date have examined their vertical and horizontal movement in soils under field conditions (Thurston-Enriquez *et al.*, 2005; Close *et al.*, 2010). As a result, knowledge of pathogen transport in soils has largely been inferred from studies of faecal indicator organism movement at grazed pastures, feedlots and manure-amended soils (Doran and Linn, 1979; Young *et al.*, 1980; Edwards *et al.*, 2000; Soupir *et al.*, 2006) or soil column or block studies amended with pathogen-containing livestock manure (Gagliardi and Karns, 2000; Davies *et al.*, 2004; Kuczynska *et al.*, 2005; Semenov *et al.*, 2009). Some physical and chemical properties that influence the vertical movement of microorganisms are soil type, water content and water flow, microbe and soil particle surface properties, cell motility, pH, plant roots, temperature, and presence of micro- and meso-faunal organisms (Mawdsley *et al.*, 1995; Unc and Goss, 2004).

Rapid horizontal transport of microorganisms to surface waters can occur when either the rainfall intensity exceeds the soil's infiltration rate or when the soil becomes so saturated that no rainfall can percolate (Tyrrel and Quinton, 2003). Factors that influence the level of microbiological contamination in runoff from agricultural lands are organism die-off rates, quantity and type of manure applied, sloping terrain, rainfall intensity and water infiltration rate (Evans and Owens, 1972; Doran and Linn, 1979; Baxter-Potter and Gilliland, 1988; Abu-Ashour and Lee, 2000; Jenkins *et al.*, 2006; Ramos *et al.*, 2006). Methods to mitigate the offsite transport of microorganisms in runoff from manure-amended soils and livestock feedlots include use of vegetative filter strips (Coyne *et al.*, 1995; Fajardo *et al.*, 2001) or vegetative treatment systems with a settling basin for solids collection and a vegetated area (Koelsch *et al.*, 2006; Berry *et al.*, 2007). Alternatively, livestock manures could be treated (e.g. composted, anaerobically digested) prior to land application, thus reducing subsequent risks associated with pathogens (Lund *et al.*, 1996; Tiquia *et al.*, 1998).

Aerosolization of pathogens

Pathogens can potentially become aerosolized during the land application of liquid and solid manures, representing a potential risk if inhaled in sufficient quantities or ingested after deposition on food crops and fomites (Brooks *et al.*, 2004; Dungan, 2010). When bioaerosols are released from a source, they can be transported short or long distances, eventually being deposited (Brown and Hovmøller, 2002; Jones and Harrison, 2004). Unlike zoonotic agents in manures, soils and waters, aerosolized microorganisms are highly susceptible to meteorological factors such as relative humidity, solar irradiance and temperature (Cox and Wathes, 1995). In general, both laboratory and field studies have shown that the viability of aerosolized microorganisms decreases with decreases in relative humidity and increases in ambient temperature and solar irradiance (Mohr, 2007). Despite the potential for bioaerosol formation during these activities, very few studies have investigated the risk of human exposure to pathogens during the land application of animal manures.

During the land application of swine and cattle slurries by tanker and high-pressure spray guns, airborne bacterial counts steadily decreased with distance from the application site and pathogenic bacteria such as *Salmonella* spp. and *Klebsiella pneumoniae* were not detected (Boutin *et al.*, 1988). During the spray irrigation of swine slurry, a marker strain of *E. coli* was detected 125 m downwind in aerosols, but not at 250 and 500 m downwind (Hutchison *et al.*, 2008). Using polymerase chain reaction (PCR) to amplify 16S ribosomal RNA genes in air samples collected immediately adjacent to the spreading of swine and dairy cattle slurry, pathogens having an aerogenic route of infection were not identified (Murayama *et al.*, 2010; Dungan, 2012). While results from these and other studies suggest a low risk for exposure to pathogens (Brooks *et al.*, 2005; Tanner *et al.*, 2005), significant knowledge gaps still exist with respect to the fate and transport of bioaerosols, making it difficult to predicate the health risks associated with aerosolized pathogens accurately (Pillai and Ricke, 2002).

Fate, transport and negative impacts of veterinary antibiotics in soil

Once in the soil, antibiotics can be transported to surface and ground waters in a dissolved phase or sorbed to soil particles and colloids (Kay *et al.*, 2004; Song *et al.*, 2010). Tetracyclines have been shown to strongly sorb to soils, while macrolides, such as tylosin, have a weaker tendency to sorb (Rabølle and Spliid, 2000; Allaire *et al.*, 2006). In contrast, sulfonamides are likely the most mobile of the antibiotics and have been detected in groundwater at relatively high concentrations (Hamscher *et al.*, 2005; Batt *et al.*, 2006). Despite such knowledge, there is still little known about the occurrence, fate and transport of VAs in the soil–water environment. Recent research has shown that the addition of pig manure to soil caused a temporary increase in tetracycline resistance genes soon after manure application (Sengeløv *et al.*, 2003). When manures are land applied, resistant bacteria are also transferred, creating the possibility of horizontal transfer of resistance genes to the indigenous soil bacteria. The addition of nutrients to soils has been shown to enhance horizontal transfer to bacteria by providing nutrients for activation of transfer as well as mobilizing genetic elements (Top *et al.*, 1990; Heuer and Smalla, 2007). Furthermore, antibiotics and their metabolites in the manures might give resistant bacteria a selective advantage after being land applied (Halling-Sørensen *et al.*, 2001).

In addition to concerns over the proliferation of antibiotic-resistant bacteria, other concerns over antibiotics in the environment are related to negative impacts on water quality and soil microbial communities. While detectable levels of antibiotics have been observed in natural waters throughout the USA, much needs to be learned about the chronic effects of low-level exposures of pharmaceuticals on human and environmental health (Kolpin *et al.*, 2002; Focazio *et al.*, 2008). Currently, there are no provisions within the Safe Drinking Water act to monitor or regulate antibiotics. With respect to impacts on soil microorganisms, sulfonamides were found to reduce enzymatic activity and have a prolonged effect on microbial community structure and diversity (Schmitt *et al.*, 2005; Thiele-Bruhn and Beck, 2005; Hammesfahr *et al.*, 2008; Gutiérrez *et al.*, 2010; Toth *et al.*, 2011). In contrast, chlorotetracycline was shown to have no effect on soil respiration and bacterial community structure, which can be attributed to sorption of the antibiotic to the soil matrix (Zielezny *et al.*, 2006). However, effects of sulfonamides were only observed when soils were amended with a C source (e.g. manure, glucose), which was responsible for stimulating bacterial growth and activity. This is an important implication, as use of manure as a fertilizer source may not only enhance the horizontal transfer of antibiotic resistance genes, but exacerbate the effect of antibiotics on the soil microbial community. As a result, there is great interest in understanding the effect of sustained applications of antibiotic-containing manures on the long-term health and function of agronomic soils. Of additional concern is the ability of some plant species to absorb antibiotics into their tissues, creating another route for exposure of humans to antibiotics (Kumar *et al.*, 2005).

Effect of manure treatment technologies on pathogens

To reduce risk factors associated with the land application of manures, various physical, chemical and biological treatment technologies could be used to reduce or eliminate the presence of pathogens (Heinonen-Tanski *et al.*, 2006). While there are advantages and disadvantages with these methods, some can provide additional benefits, such as the production of compost that can be used to enhance the properties of agricultural soils (Tester, 1990) or biogas for energy generation (Holm-Nielsen *et al.*, 2009). As mentioned previously in this chapter, there is a wide variety of technologies available to treat livestock manures; however, the only processes with a documented record of cost-effective pathogen reduction are composting and anaerobic digestion (Sobsey *et al.*, 2006; Martens and Böhm, 2009). Windrow composting was shown to eliminate *Salmonella* in pig manure when temperatures were maintained between 64° and 67°C for up to 3 weeks (Tiquia *et al.*, 1998). In a bench-scale study, *E. coli* O157:H7 and *Salmonella enteritidis* were not detected after 72 h of composting at 45°C (Lung *et al.*, 2001). In another laboratory study, Grewal *et al.* (2006) reported that *E. coli*, *Salmonella* and *Listeria* were not detectable after 3 days in dairy manure mixed with straw or sawdust and incubated at 55°C. When cattle slurry was fed into a mesophilic anaerobic digester for 24 days at an operating temperature of 28°C, only moderate reductions in *Salmonella typhimurium*, *Yersinia enterocolitica*, *L. monocytogenes* and *Campylobacter jejuni* were reported (Kearney *et al.*, 1993). Pathogenic reductions, however, are generally greater at higher temperatures used in thermophilic digesters (Lund *et al.*, 1996; Burtscher *et al.*, 1998). As an added benefit, anaerobic digestion and composting have also been found effective in significantly reducing the level of VAs in livestock manures (Arikan, 2008; Ramaswamy *et al.*, 2010; Wu *et al.*, 2011).

Summary

The sustainability of modern manure management is far from certain, with many demonstrating significant limitations from the standpoint of efficient use of manure resources and protection of environmental quality and human health. As demonstrated in this chapter, for manure management to be sustainable, a broad array of issues must be considered and addressed, all in the context of highly competitive modern livestock production systems that largely seek to minimize costs to the consumer. In the past decade, there have been major innovations in the areas of land application, manure treatment and processing and in the science of understanding the impact of manure management. As a result, major opportunities exist to improve the components of manure management. To be sustainable, these optimized components must work within the constraints of the broader livestock production system.

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