

Managing waste from confined animal feeding operations in the United States: the need for sanitary reform

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ABSTRACT

Confined food-animal operations in the United States produce more than 40 times the amount of waste than human biosolids generated from US wastewater treatment plants. Unlike biosolids, which must meet regulatory standards for pathogen levels, vector attraction reduction and metal content, no treatment is required of waste from animal agriculture. This omission is of concern based on dramatic changes in livestock production over the past 50 years, which have resulted in large increases in animal waste and a high degree of geographic concentration of waste associated with the regional growth of industrial food-animal production. Regulatory measures have not kept pace with these changes. The purpose of this paper is to: 1) review trends that affect food-animal waste production in the United States, 2) assess risks associated with food-animal wastes, 3) contrast food-animal waste management practices to management practices for biosolids and 4) make recommendations based on existing and potential policy options to improve management of food-animal waste.

Key words | agriculture, food animal, livestock, manure, policy, public health

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INTRODUCTION

The current practice of intensive, high-throughput methods for producing food animals, often referred to as industrial food-animal production (IFAP), has both benefits and risks. The latter include a range of public health and ecological consequences that are to a great extent associated with the challenge of managing the massive quantities of animal wastes. Historically, this waste was valued in agriculture as a source of nutrients for land amendment at a time when the amount of excreta produced by livestock and poultry could be applied to land at levels consistent with agronomic sustainability (Ellis & McCalla 1978; Sims 1995). However, as the numbers of livestock and poultry raised for human consumption have grown and production has become more intensive, in terms of both housing and regional location, this practice is no longer feasible. Moreover, in many areas, the land available for waste application has been reduced as a result of other uses, such as increased residential development. Taking these trends together, the adverse

impacts of land disposal of animal wastes now often outweigh their benefits (Mallin & Cahoon 2003; Cicanec 2004; Chapin *et al.* 2005).

The purpose of this paper is to: 1) assess trends that have affected food-animal waste production in the United States, 2) evaluate human health risks associated with food-animal wastes, 3) compare current practices, regulations and guidelines for the management of food-animal wastes with those required for management of human biosolids (hereafter referred to as 'biosolids') and 4) make policy and technology recommendations to improve management of animal waste.

While this review is intended to address issues relevant to waste from all terrestrial food animals, much of the relevant public health literature addresses swine and poultry production. Consequently, the waste from these animals receives relatively more attention, though, where supported by literature, available information on waste from

doi: 10.2166/wh.2010.075

dairy and beef operations is included. It should also be noted that, while beyond the scope of this review, aquaculture wastes are of significant environmental and public health concern (Sapkota *et al.* 2008; Cole *et al.* 2009).

Changes in food-animal production

Most dairy cows, chickens, pigs and turkeys in the US are now housed in high-density, confined spaces. A single operation can generate millions of gallons of liquid animal waste annually and/or large quantities of manure. These animals are supplied manufactured feeds rather than access to forage. As a result, food-animal production is now independent of geographical conditions, since these operations no longer require access to fertile land for growing feeds or space for animals to live outside (Aho 2003). The consequence is a 'separation of land from livestock' (Golleson *et al.* 2001).

United States

In the US, the change in food-animal production practices began over the past 60 years, first in the broiler poultry industry and then in other livestock production. Since the 1950s, the number of operations in the US producing livestock and poultry for commercial markets has decreased by 80% (Miner *et al.* 2000), while production has more than doubled (United States Department of Agriculture 2009b). Figure 1 demonstrates increases in domestic meat

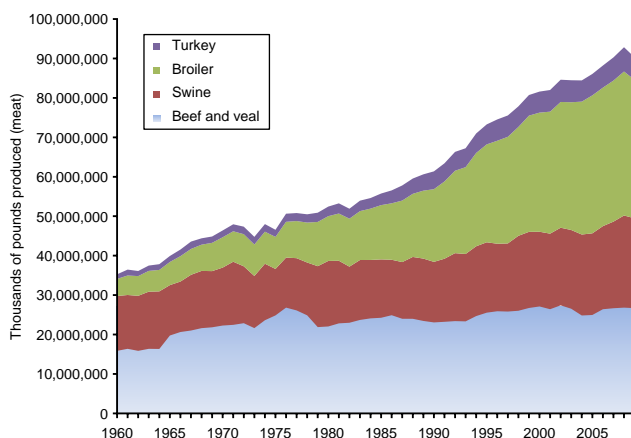


Figure 1 | Domestic production trends, by animal sector, 1960–2009.

production (in terms of beef, swine, broiler chicken and turkey meat) from 1960 to 2009 using USDA data (United States Department of Agriculture 2009b). All animal sectors have shown substantial increases in production over the past 50 years, though the most striking change is in broilers, which increased by a factor of eight.

The geographic distribution of food-animal production has also changed. Poultry production and processing provides an example that is illustrative of geographic distribution trends of food-animal production, as demonstrated in Figure 2. Prior to 1950, poultry production took place in smaller scale operations over many states in the US, and it could be effectively mapped in terms of facilities processing at least 50,000 birds annually. However, as displayed in the second map from 2007, poultry production has become concentrated in regions of the south (known as the broiler belt) and along the eastern seaboard, and now requires visual notation at the level of one million or more birds. This transformation is attributed to many factors, including intensified production methods and vertical integration of the industry.

The industrial system of raising and processing large numbers of animals in confinement was first developed by the broiler poultry industry in the 1930s, and supported a change in production from a seasonal cycle to continuous production. During World War II, the US army was the largest consumer of broiler chickens and, following the war, the integrated model was increasingly adopted by the industry (United States Department of Agriculture 1999). Integrators (i.e. companies that perform most of the production activities including: hatcheries, breeder flocks, feed mills and processing plants) maintain strict controls over much of the farmers' activities. In the current system, farmers or growers lease the chickens from the integrator, never taking ownership except in the case of mortalities, and then receive pay for the marketable weight of the birds at the time of processing. Integrators are responsible for all aspects of the raising of the animals, including formulation and provision of the feed (Boyer *et al.* 2006) and for veterinary treatments. It is estimated that nearly three-quarters of poultry farmers earn below poverty-level wages and most farmers have significant debt from capital investments that reduce their bargaining power with integrators (Farmers' Legal Aid Group 2001). Consequently,

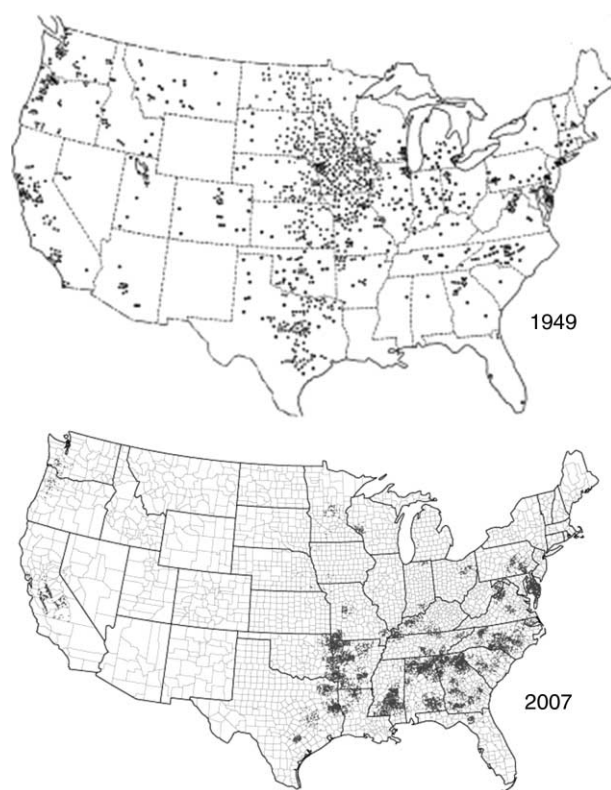


Figure 2 | Changing demographics of the poultry industry (1949 and 2007) (Aho 2003; United States Department of Agriculture 2009a). In Map 1, a point represents a plant processing 50,000 or more chickens annually. In Map 2, each point represents 1,000,000 chickens produced annually (8.9 billion total).

farmers generally do not treat the waste because of added costs (Boyd 2001).

After 1970, other sectors of food-animal production adopted practices similar to the poultry industry. At present, most animals raised for human consumption in the US are grown in confinement under controlled lighting, temperature and feeds. Swine production demonstrated a 574% increase in the number raised in confinement in the US between 1982 and 1997 (Golleson *et al.* 2001; Ribaud *et al.* 2003b). It has been estimated that food animals produced in confinement account for 80 billion US dollars of the 106 billion in annual sales resulting from US animal agriculture (Golleson *et al.* 2001); this represents greater than half the revenue from all US farms in 2001. Table 1 demonstrates the percentage of food animals raised by size category of animal feeding operation (AFO) and percentage of land base occupied by each AFO category. Nearly half of all food

animals are produced in the largest sized AFOs, and these animals occupy less than 5% of the land base used for animal production.

Globalization of industrial food-animal production

Globally, food-animal production has increased more than five-fold in the last 50 years, and the industrialized model is the fastest growing system of animal production worldwide (Halweil & Nierenberg 2004). Confined food-animal production provides 74% of poultry, 50% of pork and 43% of beef (Halweil & Nierenberg 2004). It is growing rapidly in Asia, Africa and Latin America and in many countries with relatively weak veterinary and public health infrastructure (Halweil & Nierenberg 2004).

Demand is increasingly driven by urban consumer populations at home as well as exports for expanding economies, and this demand is met by both multinational corporations and national entities. As an example, at the start of the 1980s, the Chinese government began a concerted effort to guarantee the provision of meat and poultry to urban residents, and intensified, mechanized and confinement rearing was introduced to achieve this (Wolf *et al.* 2003). Growth of the food-animal sector in China has resulted in waste management problems in multiple regions, including the Beijing municipality, where the rapidly expanding industry is generating more nutrients than can be supported by agricultural land, resulting in discharges of nitrogen and phosphorous into surface and ground waters (Wolf *et al.* 2003). In 2003, animal waste generated in China was estimated to be 3.2 billion tons—three times the amount of industrial solid waste produced in that same year (Wang *et al.* 2005).

The European Union has also adopted industrialization of food-animal production. In Bretagne, a region in western France, swine production increased from 1.1 M to 8.8 M between 1955 and 1997 and currently accounts for 55% of swine production in the country (Petit & Vanderwerf 2003). Similarly, in the Netherlands, between 1950 and 1980, the number of swine increased from 2 to 10 million, poultry from 41 to 81 million and dairy cows from 1.4 to 2.4 million (Westhoek *et al.* 2004).

Access to new markets as well as the increase in standard of living and changes in food preferences is driving

Table 1 | Percentage of animals produced in the US for different size categories of animal feeding operations (AFOs), and percent of land base and average acres per animal unit per AFO category (one animal unit (AU) is approximately 1,000 pounds of animal live weight) (Gollehon *et al.* 2001)

Animal type	Very small AFOs <50 AUs*	Small AFOs 50–299 AUs	Medium AFOs 300–999 AUs	CAFOs† >1,000 AUs
<i>Feedlot beef</i>				
% of total AUs	5%	8%	7%	80%
% of land base	66%	25%	6%	4%
<i>Dairy</i>				
% of total AUs	6%	54%	19%	22%
% of land base	12%	75%	10%	3%
<i>Swine</i>				
% of total AUs	7%	32%	26%	35%
% of land base	43%	45%	9%	2%
<i>Poultry</i>				
% of total AUs	3%	40%	27%	30%
% of land base	36%	45%	14%	4%
<i>Total</i>				
% of total AUs	5%	33%	19%	43%
% of land base	33%	53%	10%	4%
Average acres per animal unit	14.91	3.50	1.20	0.18

*1 animal unit (AU) is approximately 1,000 pounds of animal live weight (Gollehon *et al.* 2001).

†CAFOs—Concentrated Animal Feeding Operations. A CAFO has approximately 1,000 AUs.

multinational companies to expand their operations overseas. Between 1993 and 2000, the US-based pork company Smithfield went from owning no foreign subsidiary companies to owning companies in Canada, France, Mexico, Brazil and Poland (Cummings 2000). In countries where US companies have operations, it is expected that meat and poultry production will experience the same level of expansion and integration that has been established in the US (Cummings 2000). Production in some countries is dominated by foreign companies; in Vietnam, for example, it is estimated that 75% of large-scale poultry production is carried out by foreign companies (Moi 2006).

Globalization of industrial food-animal production companies is concerning, due to potential differences in enforcement of legal mechanisms to prevent exposures to and environmental degradation resulting from waste-borne contamination. In addition, expansion of these companies across borders has strong environmental justice implications, similar to those experienced domestically (Mirabelli *et al.* 2006; Donham *et al.* 2007), in that persons residing near foreign-owned animal operations

endure disproportionate exposures to generated waste as compared to those benefiting from the production of these food animals.

WASTE PRODUCTION AND MANAGEMENT

Quantity of waste generated

According to the US Department of Agriculture, confined food animals produce roughly 335 million tons of waste (dry wt.) per year (United States Department of Agriculture 2005). In contrast, biosolids generated by publicly owned treatment works was estimated to be 7.6 million tons (dry wt.) in 2005. Biosolids are comprised of waste streams from residential and commercial sources, stormwater runoff and residual compounds from the wastewater treatment process (National Research Council 2002). Approximately 66% of generated biosolids (5.0 million tons) are applied to land only after treatment to meet certain classification standards (United States Environmental Protection Agency 1999).

For food animals, the USEPA estimates 'nearly all' of the waste produced, including manure, litter and process

wastewater, is applied to land without any required pretreatment or classification (United States Environmental Protection Agency 2004). For example, for poultry litter (i.e. excreta, spilled feed, feathers, soil and bedding material), over 90% is applied to land (Moore *et al.* 1995). In many states, poultry litter is also used as a feedstuff in beef cattle production (Martin & McCann 1998). The US Food and Drug Administration estimates that up to 20 to 25% of broiler litter in some states is fed to cattle (Lu *et al.* 2003); in the state of Florida, it is estimated that 35% of the 1 million tons of litter produced in 2003 was used in feed (Sapkota *et al.* 2007b).

Handling of animal waste and biosolids

The shift towards animal confinement has affected the methods and options for handling the waste prior to disposal. With insufficient space available for each animal to freely excrete and for natural systems to absorb and decompose these wastes, confined swine and cattle operations have developed water-based slurry systems that essentially flush waste from the floors where the animals are housed, and channel the liquid slurry into large ponds for storage (Figure 3). With the exception of poultry operations, industrial food-animal operations use water or slurry-based systems, requiring large pits for storing the increased volumes of waste (United States Environmental



Figure 3 | Swine waste lagoon (Georgia, USA). Photo courtesy of Natural Resources Conservation Service.

Protection Agency 2007a,b, 2008a). These systems also require large volumes of water—in dairies where flush cleaning is practised, water use can reach 150 gal/d/cow, and a 5,000-swine CAFO can use up to 340 million gallons of drinking water and flushing water each year (Kridler 1999). Consequently, the use of flush systems increases the volume of waste necessitating treatment and can augment odours associated with anaerobic conditions of the wastewater. Waste ponds, often referred to as ‘lagoons’, are the most common waste handling system in US swine production (Wing *et al.* 2002). Leaks and ruptures associated with poor management or weather are frequently observed (Mallin & Cahoon 2003). In Iowa, 30 separate spills occurred between 1992 and 1997; other spills have been documented in Missouri, Nebraska, Ohio, Maryland and New York (Mallin & Cahoon 2003; United States Environmental Protection Agency 2003; Hodne 2005).

Treatment standards are required for biosolids prior to land disposal. Effluent from wastewater treatment plants must be stabilized through defined processes to reduce pathogen levels, odour and content of volatile solids, vector attraction and metal content. After treatment, biosolids are classified as either Class A or Class B, based on the concentration of faecal coliforms and the treatment process applied (National Research Council 2002). Class A biosolids are assumed to be pathogen-free and can be land applied without any pathogen-related restrictions. Class B biosolids must have a faecal coliform count less than 2×10^6 per gram (dry wt.) or have been treated by a ‘process to significantly reduce pathogens’ (National Research Council 2002). According to the Part 503 rule, Class B biosolids cannot be packaged and sold or given away or used on lawns or in home gardens, but can be applied in bulk quantities on agricultural, forest and mine reclamation sites (National Research Council 2002). There are also specific pollutant concentrations for nine metals. However, if biosolids do not meet these limits, they can still be approved by EPA for land amendment as long as they are accompanied by an information sheet specifying maximum annual application rate.

Despite the fact that levels of pathogens in animal waste can be equal to or even higher than levels found in human wastewater and sludge (Table 2) (National Research Council 2002; United States Environmental Protection

Agency 2003), there are no treatment or other processing requirements, nor any prescribed pathogen or metals criteria required for animal waste. Swine waste, for example, can contain 28 times the density of faecal streptococci found in

human waste (Miner *et al.* 2000). Moreover, many organisms pathogenic to humans, such as *Campylobacter* spp. and *E. coli* O517:H7 are endemic in animal feeding operations (AFOs) (Hutchison *et al.* 2004).

Table 2 | Contents of animal waste and biosolids by average number of selected microbes*

	Mean concentration in waste (reference)	
	Animal waste	Wastewater and sludges
<i>Bacteria</i>		
Faecal coliforms	3.3×10^6 swine (Miner <i>et al.</i> 2000) ^{†,‡}	8.98×10^8 (Soares <i>et al.</i> 1992) [†]
	1.3×10^6 poultry (Miner <i>et al.</i> 2000) ^{†,‡}	8.5×10^8 (Gibbs <i>et al.</i> 1994) [†]
	2.3×10^5 cattle (Miner <i>et al.</i> 2000) ^{†,‡}	3.4×10^4 (Payment <i>et al.</i> 2001) [§]
Faecal streptococcus		3.6×10^7 (Dahab & Surampalli 2002) [†]
		3.4×10^6 (Lasobras <i>et al.</i> 1999) [†]
	10^6 – 10^7 swine (Pourcher <i>et al.</i> 1991; Miner <i>et al.</i> 2000) [†]	1.5×10^7 (Soares <i>et al.</i> 1992) [†]
	10^5 – 10^8 poultry (Pourcher <i>et al.</i> 1991; Miner <i>et al.</i> 2000) [†]	5.0×10^7 (Gibbs <i>et al.</i> 1994) [†]
	10^5 – 10^6 cattle (Pourcher <i>et al.</i> 1991; Miner <i>et al.</i> 2000) [†]	5.1×10^5 (Payment <i>et al.</i> 2001) [§]
<i>Salmonella</i> spp.		2.1×10^7 (Dahab & Surampalli 2002) [†]
		1.5×10^5 (Lasobras <i>et al.</i> 1999) [§]
		1.58×10^4 (Moce-Llivina <i>et al.</i> 2003) [§]
	9.6×10^5 swine (Hutchison <i>et al.</i> 2004) ^{†,‡}	2.9×10^5 (Gibbs <i>et al.</i> 1994) [†]
<i>E. coli</i> O157	10^1 – 10^7 poultry (Sims & Wolf 1994; Gast & Holt 1998; Whyte <i>et al.</i> 2001; Hutchison <i>et al.</i> 2004) [†]	
	10^4 – 10^7 cattle (Morisse <i>et al.</i> 1992; Hutchison <i>et al.</i> 2004) [†]	
	6.9×10^4 swine (Hutchison <i>et al.</i> 2004) ^{†,‡}	1.5×10^4 (Payment <i>et al.</i> 2001) [§]
<i>Campylobacter</i> spp.	ND poultry (Hutchison <i>et al.</i> 2004) ^{†,‡}	
	10^2 – 10^6 cattle (Cassin <i>et al.</i> 1998; Hutchison <i>et al.</i> 2004) [†]	
	3.1×10^2 swine (Hutchison <i>et al.</i> 2004) ^{†,‡}	
<i>Giardia</i> spp.	2.6×10^2 poultry (Hutchison <i>et al.</i> 2004) ^{†,‡}	
	3.2×10^2 cattle (Hutchison <i>et al.</i> 2004) ^{†,‡}	
	5.3×10^4 swine (Hutchison <i>et al.</i> 2004) ^{†,‡}	1.1×10^2 (Chauret <i>et al.</i> 1999) ^{**}
<i>Cryptosporidium parvum</i>	ND poultry	4.65 (Payment <i>et al.</i> 2001) [§]
	2.2×10^2 cattle (Hutchison <i>et al.</i> 2004) ^{†,‡}	5.6×10^4 (Gibbs <i>et al.</i> 1994) [†]
	3.0×10^2 swine (Hutchison <i>et al.</i> 2004) [†]	1.7×10^5 (Soares <i>et al.</i> 1992) [†]
	ND poultry [†]	
	10^1 – 10^4 cattle (Hutchison <i>et al.</i> 2004) [†]	4.77×10^1 (Chauret <i>et al.</i> 1999) ^{**}

*Includes only feedlot cattle, dairy cattle, poultry and swine. BDL—below detection limit, ND—non-detect, Biosolids data adapted from Sidhu & Toze (2009).

[†]g⁻¹ dry weight.

[‡]Reference provides geometric mean only.

[§]g⁻¹ wet weight.

**Sludge L⁻¹.

RISKS ASSOCIATED WITH FOOD-ANIMAL WASTE

Because of the large amount of untreated animal waste that is disposed of on land, AFOs have emerged as a significant source of water pollution. The problems are exacerbated when operations over-apply waste, a common practice among larger operations that lack sufficient land for waste application (Jerger 2004). Additionally, problems arise when operations apply waste on steep slopes, near waterways and natural drainage-ways or when certain weather or soil conditions exist, such as rainfall, snow-covered land, frozen land, saturated soil and/or poor soil porosity (Hodne 2005). The US EPA has determined that 16% of the more than 300,000 impaired river and stream miles in the US are due to animal feeding operations (Centner 2004). Of the US rivers and estuaries that fail ambient water quality standards, 40% fail because of pathogens (Smith & Perdek 2004). In the western US states, it is estimated that 80% of the impaired river and streams are associated with livestock (Agouridis *et al.* 2005). In addition to ecological impacts, animal waste can pose a risk to human health.

Microbial pathogens

Pathogenic microorganisms in food-animal waste include a large number of species. *Salmonella* spp., *Campylobacter* spp., *Listeria monocytogenes*, *Cryptosporidium parvum*, *Giardia lamblia*, *E. coli* O157:H7 are the most common causative agents of disease outbreaks that most likely originate from animal-feeding operations (Smith & Perdek 2004). The concentration of zoonotic pathogens in a given watershed has been found to increase with proximity to and number of animal operations (Cox *et al.* 2005). Many of these organisms can survive from several days to several months in manure and wastewater, and are often transported long distances in the environment (Stehman *et al.* 1996; Hodne 2005; Graham *et al.* 2009a). Zoonotic protozoa, such as *Cryptosporidium* and *Giardia*, are of particular concern in food-animal waste, due to their high prevalence and environmental stability (Duffy & Moriarty 2003). Prevalence rates of *C. parvum* range from 1.1 to 62.4% in apparently healthy cattle and up to 79% in symptomatic calves (Villacorta *et al.* 1991; Scott *et al.* 1995; Fayer *et al.*

2000; Hoar *et al.* 2001). Asymptomatic carriage is an important aspect as oocysts are excreted in faeces without any sign of infection. According to the CDC, *Cryptosporidium* oocysts have been found in 67–97% of surface water sampled; the percentage attributable to food animals is unknown (Kramer *et al.* 1996). The ability of these microparasites to withstand chlorination and other disinfectants increases the threat of disease (Suwa & Suzuki 2003).

In contrast to bacteria and protozoa, there is relatively less knowledge regarding the survival and infectivity of viruses contained in food-animal waste. From the few existing studies measuring the inactivation of animal viruses, however, it is clear that viral hazards may persist significant amounts of time (some non-enveloped viruses can persist 300 days) without sufficient treatment (Pesaro *et al.* 1995). Animal wastes may also contain influenza and other viruses, which may persist for a number of days (Mawdsley *et al.* 1995; Gerba & Smith 2005).

Further examination of the presence of fungal organisms in animal waste is needed. In its *Agricultural Waste Management Field Handbook*, the USDA identified *Coccidioides immitis*, *Histoplasma capsulatum* and various microsporium and trichophyton organisms as present in waste (Kridler 1999), though the existing literature base characterizing the frequency and levels of these contaminants in waste is limited.

A number of studies have found that the use of antibiotics in food-animal production has contributed to the emergence of antibiotic resistance among pathogenic and commensal bacteria of humans via food and environmental pathways (Levy *et al.* 1976a,b; Feinman 1979; Cohen & Tauxe 1986; Aarestrup 1999; Shea 2003; Angulo *et al.* 2004; Bae *et al.* 2005; Price *et al.* 2005; Poppe *et al.* 2006; Graham *et al.* 2009b). Moreover, studies have found antibiotic-resistant bacteria in soil, surface and ground water, air and wild animal populations near AFOs (Henzler & Opitz 1992; Chapin *et al.* 2005; Cole *et al.* 2005; Sapkota *et al.* 2007a; Stine *et al.* 2007). At present, the fraction of antimicrobial-resistant infections in humans attributable to antibiotic use in food animals has yet to be determined.

Pharmaceuticals

Antimicrobials

Antimicrobials were introduced into food-animal production some 50 years ago (Libby & Schaible 1955), and most use in the US is for improving growth, feed efficiency and disease prevention. Estimates of the amounts of antimicrobials used in US food-animal production currently range from 3.1 million pounds to approximately 25 million pounds annually (i.e. range from 13 to 70% of all antimicrobial use in the US) (Mellon *et al.* 2001; Animal Health Institute 2002). This high variability in estimates stems from the failure of current regulations to require public reporting of actual use. Moreover, ambiguity remains regarding the basis for administering antibiotics at sub-therapeutic levels.

As have been found in biosolids (National Research Council 2002; Jones-Lepp & Stevens 2007), antimicrobial residues have been found in surface and groundwater located near swine and poultry operations (Campagnolo *et al.* 2002), and it is estimated that 25–75% of antimicrobials used in food-animal production are excreted unaltered in the waste (Kummerer 2004). In a study by the US Geological Survey (USGS) of 139 streams across the US, antimicrobials commonly used in food-animal production, such as lincomycin and tylosin, were regularly found (Kolpin *et al.* 2002; McGee *et al.* 2003), although the source of these compounds was undetermined. It has been estimated that 13.5 million pounds of antibiotics are excreted in animal waste per year in the US; this, however, is based on estimates of antibiotic use at AFOs, which is unknown (Florini *et al.* 2005).

Roxarsone, an organo-arsenical, is currently used as a feed additive in the diets of chickens and swine to improve growth and feed efficiency and to prevent coccidial infections (Han *et al.* 2003). The compound is excreted in animal waste and degrades to arsenite and arsenate (leachable forms of arsenic). Studies of poultry litter have found that 70 to 90% of the total arsenic in litter is water soluble (Garbarino *et al.* 2003; Jackson *et al.* 2003). Levels of arsenic in poultry litter are roughly ten times higher than those found in biosolids (Han *et al.* 2003). The USGS has calculated, based on arsenic concentrations measured in poultry waste, that between 250,000 and 350,000 kilograms of arsenic (from poultry feed additives) is applied annually

to land in the US (Rutherford *et al.* 2003). The finding that arsenate is a degradation product from Roxarsone is important because arsenate is one of the two inorganic forms of arsenic commonly found in drinking water and associated with cancer (National Research Council 2001). In addition, unmetabolized Roxarsone has been demonstrated to have angiogenic potential in cultured human aortic and lung microvascular endothelial cells, providing suggestive evidence of the possibility of vascular effects resulting from Roxarsone exposure (Basu *et al.* 2008).

Hormones

All birds and mammals (including humans) naturally excrete hormones in waste and it is known that the use of hormones by human populations can increase the concentrations of estrogens in human wastes and wastewater treatment effluent (Kolodziej *et al.* 2004; Zheng *et al.* 2008). Similarly, there are impacts associated with the treatment of livestock with hormones or synthetic compounds to synchronize reproductive cycles and/or improve growth and feed efficiency. The most commonly used anabolic agents in the US, to increase animal growth, include androgens (testosterone and trenbolone acetate), estrogens (17 β -estradiol and zeranol) and progestogens (progesterone and melengestrol acetate) (Soto *et al.* 2004). Most of these agents end up in the excreta of animals and have been found in surface and groundwater impacted by animal waste, persisting for long periods of time (Arai *et al.* 2003; Han *et al.* 2003; Lorenzen *et al.* 2004; Soto *et al.* 2004). Metabolites of trenbolone acetate, for example, have been found to persist in animal waste for more than 270 days (Schiffer *et al.* 2001). Sellin and colleagues found that minnows exposed to faecal slurry from steer implanted with trenbolone acetate experienced feminizing and demasculinizing effects as compared to those exposed to the slurry from unimplanted steer (Sellin *et al.* 2009). Natural and synthetic estrogens, including steroidal estrogens (17 α -estradiol, 17 β -estradiol and estrone) have been found to cause reproductive and developmental effects on a number of aquatic organisms, and it has been projected that the amount of these compounds emitted by dairy cows and swine is ten times greater than the mass flow of estrogen from US wastewater treatment plants (Raman *et al.* 2004). An estimated 90% of

beef cattle raised in animal feeding operations receive hormone supplements to improve feed efficiency and growth (Soto *et al.* 2004). In a study of water bodies impacted by cattle feedlot effluent, researchers found that samples contained high levels of hormonally active chemicals (Soto *et al.* 2004). Estrogen excretion from cattle, pigs and chickens in the US has been estimated to be 45, 0.8 and 2.7 metric tons (Mg) estrogens yr⁻¹, respectively (Hanselman *et al.* 2003).

Environmental pathways of exposure to contaminants in animal waste

Environmental pathways of exposure to contaminants in animal waste are increasingly documented as surveillance efforts improve (Cole *et al.* 2005). Historically, research has focused on occupational and food-borne exposure pathways (associated with meat consumption), and transmission by these pathways is well documented (Price 2006). Because many of the contaminants in animal waste are able to persist in different environments, there is concern that waste and land where waste is applied may serve as a reservoir for different contaminants found in the waste. The spread of these contaminants could occur through: 1) crops fertilized with waste or irrigated with water contaminated with waste (probable pathway for two recent outbreaks of *E. coli* 0157:H7) (Jay *et al.* 2007); 2) aerosolized particles of waste emitted from confinement or waste storage facilities, fields fertilized with waste or trucks transporting animals for processing (Chapin *et al.* 2005; Gibbs *et al.* 2006; Rule *et al.* 2008); 3) runoff of waste into groundwater and surface water (Burkholder *et al.* 2007; Sapkota 2007a; Silbergeld & Nachman 2008); and 4) contamination of other fomites. Additionally, there is the potential for the spread of microorganisms by insects (Graham *et al.* 2009b), rodents (Vindigni *et al.* 2007; Burriel *et al.* 2008) and wild avians (Cole *et al.* 2005) that may be particularly attracted to AFOs where sources of food exist (e.g. spilled feed and animal manure).

HISTORY OF BIOSOLIDS AND ANIMAL WASTE MANAGEMENT IN THE US

By the early part of the nineteenth century, the management of human excreta was firmly established under the rubric

of public health. Ordinances by public health authorities regulated construction of waste storage systems, timing and processes of cleaning, and determination of the location where contents could be applied. In the latter part of the nineteenth century, with the advent of the 'germ theory', as well as numerous outbreaks associated with poor sanitation around that time, there has been a continuous and concerted effort in the US by federal, state and local governments to increase treatment of public wastewater and sewage sludge in order to protect public health and the environment (Melosi 2000). In fact, nearly 80 billion dollars in grants and low-interest loans have been given by the federal government to provide for wastewater treatment and biosolids disposal (National Research Council 2002). Management of biosolids continues to receive a great deal of public attention, highlighted by the more than 2,000 technical papers written on the subject within the last three decades (O'Connor *et al.* 2005).

In contrast to biosolids, little progress and few enforceable requirements for testing and treating food-animal waste have been developed. It was not until the mid-1980s that large livestock operations could be subject to regulations in the revised Federal Water Pollution Control Act (i.e. Clean Water Act). The provisions set forth, however, contained conditions that allowed most operations to avoid compliance, and only a small fraction of operations had obtained pollution permits by 1995 (Metcalf 2000). Additionally, they focused solely on nutrients. Other revisions, such as amendments to the Clean Water Act in 1987, established the Section 319 National Monitoring Program to evaluate the effectiveness of non-point source pollution control technologies and augment monitoring in selected watersheds. At the same time, voluntary management programmes began under the US Department of Agriculture's Natural Resources Conservation Service (NRCS). The NRCS now assists livestock and poultry operations to encourage improved nutrient management practices (Golleson *et al.* 2001). Under the NRCS, the Environmental Quality Incentives Programs (EQIP), for example, was developed to help food-animal producers comply with regulatory requirements and address natural resource concerns. Currently, 60% of programme funds go to assist farmers in addressing natural resource

issues related to animal production, and at least half of the funding for the programme goes to small and medium AFOs (Gollehon *et al.* 2001). Additionally, the NRCS has implemented the Conservation Technical Assistance programme to help land users plan and implement conservation systems for improving soil and water quality. Much of the effort involving the management of animal waste, however, does not address pathogenic microorganisms or pharmaceuticals, and remains voluntary and under-funded (Ruhl 2001).

US POLICY OPTIONS TO IMPROVE MANAGEMENT

Federal and state regulations have not kept pace with the growth and increased concentration of food-animal production, and few policies have been established to adequately mitigate the growing evidence of public health impacts associated with food-animal waste.

For small (fewer than 300 animal units) and medium (300–999 animal units) sized AFOs, the regulatory framework relies almost completely on voluntary practices focused solely on nutrient management. Programmes funded by the USDA aim to assist small farmers in evaluating existing facilities and management systems, and highlight opportunities to take voluntary actions. It is predicted, however, that voluntary efforts to reduce impacts associated with nutrients will not be successful, based on the lack of incentives (Ribaudo *et al.* 2003a; Hodne 2005).

The main US federal regulatory policy to address the management of food-animal wastes is the US Environmental Protection Agency (EPA) concentrated animal feeding operations rules (referred to as the CAFO rules from this point forward), established in 2003 under the Clean Water Act. According to the new rules, animal-feeding operations with more than 1,000 animal units must obtain a water pollution permit and implement a nutrient management plan (United States Environmental Protection Agency 2003). The minimum number of animals required to be defined as a CAFO is approximately 1,000 head of beef cattle, 700 dairy cattle, 2,500 hogs or 125,000 broiler chickens (United States Environmental Protection Agency 2008b).

The US concentrated animal feeding operations (CAFO) rules

In 2003, the US EPA established the CAFO final rule (CAFO rules) under the Clean Water Act that applies mainly to operations having 1,000 animal units or more. Approximately 15,500 animal-feeding operations (AFOs) out of the 238,000 are now considered CAFOs and must obtain water pollution permits and develop and implement a nutrient management plan. CAFOs have been regulated for more than 25 years, however, prior regulations were deemed unsuccessful at reducing their contribution to non-point source pollution (United States Environmental Protection Agency 2003). The new CAFO rules explicitly require all CAFOs to obtain water pollution permits. These permits are administered by the EPA or by state agencies authorized by the EPA to regulate the programme. Authorized states are required to have legislation that is at least as stringent as the federal programme. Prior to the rule, only 4,100 operations out of 12,700 that met the EPA regulatory definition of a CAFO had obtained pollution permits due to many operations having circumvented the permitting process (Copeland 2008b). Although a significant improvement over past legislation, the CAFO rules would apply to only 40% of food-animal waste (i.e. 200 out of 500 million tons wet wt.) and does not address pathogenic microorganisms in food-animal waste (United States Environmental Protection Agency 2003).

The National Pollutant Discharge Elimination System (NPDES) permit, issued by the EPA or federally approved states, is the key technology-based mechanism for regulating CAFOs. CAFOs are required to procure NPDES permits that place limits on the type and quantity of pollutants that can be released into US bodies of water. The NPDES permit requires technology-based restrictions on water pollution and includes: 1) adequate storage of animal waste and wastewater, including operation and maintenance capability; 2) proper management of animal mortalities (disposal in waste is not allowed); 3) diversion of clean water from the operation; 4) prevention of contact between animals and US bodies of water; 5) proper chemical disposal (disposal in waste is not allowed); 6) identification of site-specific conservation practices to control runoff of pollutants into surface water; 7) annual nutrient analyses

of waste, wastewater and soil to be disposed of and every five years analyse soil from fields where waste is applied; 8) land application of waste in accordance with site-specific nutrient management plans; and 9) maintenance of records that will document implementation and management of the topics described above (Koelsch 2005).

Effectiveness of CAFO rules

According to the United States Government Accountability Office (GAO), neither the states nor EPA are sufficiently equipped to implement the CAFO rules. Moreover, EPA's limited oversight of state programmes is expected to result in ineffective implementation of the CAFO rules (Government Accountability Office 2003). Many states have opted to use their own permitting system for AFOs that include more stringent management requirements (Metcalf 2000); however, the permit programmes in a number of states do not meet the standards of the NPDES programme. Additionally, many states do not issue any type of permit to CAFOs, leaving the operations effectively unregulated (Copeland 2008b). In some cases, inspections of CAFOs have come only after citizen complaints or accidental releases. Furthermore, the GAO recommended that the EPA develop and implement an enforcement plan, laying out a strategy of how the agency will operationalize the increased oversight responsibilities. It is also expected that limited budgets in many states will impact the implementation of the revised regulations, and, hence, improvements to water quality are doubtful (Centner 2004).

In addition, many of the provisions initially proposed for the CAFO rules were weakened based on industry feedback. For example, operations with more than 500 animal units were proposed to be designated as CAFOs; under the '500 animal units' proposal, 25,590 AFOs would need a NPDES permit. However, the EPA rejected this based on the costs imposed to regulators and producers (United States Environmental Protection Agency 2003). The EPA also opted to not include requirements for co-permitting of large companies (i.e. integrators) who own the animals and contract with the farmers. Integrators have extensive control over AFOs, and the proposed changes would have required companies to be responsible for the waste that their animals generate. The agriculture industry,

however, argued that it would make corporations liable for waste management decisions that are out of their control (Copeland 2008b). Other significant changes to the revised regulations are the lack of requirements to monitor groundwater and surface water and limitations on heavy metals (e.g. arsenic in poultry waste), pathogens and antibiotics, which have been found in the environment in significant amounts (Ribaudo *et al.* 2003a; Rutherford *et al.* 2003; Pierini *et al.* 2004; Raman *et al.* 2004). Moreover, small and medium-sized operations lack mandatory guidelines, resulting in 40% of food-animal waste generated in the US going unregulated (Centner 2004).

Judicial challenges to the CAFO rules

In 2005, a number of provisions of the CAFO rules were challenged by both agricultural and environmental groups in *Waterkeeper Alliance et al. v. Environmental Protection Agency* in the Second Circuit Court of Appeals. Both groups argued that certain provisions of the CAFO rules were in violation of the federal Clean Water Act (CWA) (United States Court of Appeals for the Second Circuit 2005; Centner 2006; Centner & Feitshans 2006). In this paper, only those provisions that were vacated or remanded back to the US EPA are reviewed (see Centner *et al.* for a more complete discussion) (Centner 2006; Centner & Feitshans 2006); the provisions not vacated or remanded through the 2005 action remain effective.

Farm petitioners (the American Farm Bureau, the National Chicken Council and the National Pork Producers Council) argued that owners and operators of CAFOs were not mandated under the CWA to obtain NPDES permits if there was no history of discharge of pollutants to navigable waters (United States Court of Appeals for the Second Circuit 2005; Centner 2006). The CWA authorizes EPA to regulate operations, using NPDES permits, only if they discharge pollutants. Thus, if there is no violation (i.e. proven discharge), there is no obligation of CAFOs to obtain an NPDES permit. The CWA gives the EPA power to regulate only 'actual discharges—not potential discharges'.

Environmental petitioners (Waterkeeper Alliance, Inc., Sierra Club, Natural Resources Defense Council, Inc. and the American Littoral Society) asserted that EPA's oversight

of NPDES permits was inadequate. They claimed that the CAFO rules do not provide for a sufficient review of each operation's nutrient management plan and failed to require the terms of the nutrient management plan in the permit. Moreover, they stated that public participation was precluded due to the exclusion of the terms of the nutrient management plans in the NPDES permits. Without the nutrient management plans, the public is denied access to the essential part of the NPDES permit. The court agreed that there was no 'meaningful review' of nutrient management plans, and that this part of the rule was contrary to the Clean Water Act; permitting authorities must now review nutrient management plans ([United States Court of Appeals for the Second Circuit 2005](#)). Additionally, the court ruled that nutrient management plans must be included as part of the NPDES permit and that the absence of the plans does not comply with regulatory participation that the CWA guarantees the public.

Other arguments raised were the lack of standards for reducing pathogens in food-animal waste. The EPA did not dispute their responsibility to promulgate a standard for reducing pathogens, but argued that the technology controls they evaluated, such as anaerobic digesters, would impose high costs and not necessarily lead to a reduction in pathogens. The court found that this did not agree with the CWA and remanded this aspect to the EPA to set standards for pathogen reduction. The court noted, however, that EPA may determine that current practices may be sufficient to reduce pathogens; thus improved treatment standards may not be required ([United States Court of Appeals for the Second Circuit 2005](#); [Centner & Feitshans 2006](#)).

The final challenge, submitted by the environmental petitioners, argued that the CAFO rules fail to set water-quality-based effluent limitations and also limited states from doing the same. The CWA requires that the EPA set effluent limitations based on water-quality objectives for a specific portion of the navigable waters that may be affected by a point source (i.e. CAFO) or group of point sources. In the CAFO rule, it states that where a CAFO has implemented site-specific nutrient management plans, 'no further effluent limitations will be authorized to ensure compliance with water quality standards'. The court directed the EPA to explain more clearly its decision

not to promulgate water-quality-based effluent limitations for discharges, other than agricultural stormwater discharges.

Revisions associated with the *Waterkeeper* case as embodied by the 2008 CAFO rule no longer mandate CAFO owners and operators to apply for a NPDES permit, except for in the case of a known or proposed discharge. Since most CAFOs remain without NPDES permits and no financial (or other) incentives to obtain a permit currently exist, improved waste management is unlikely ([Golleshon et al. 2001](#)). States will now be placed in a difficult position of having to first prove a facility has discharged and then take enforcement action ([Copeland 2008a](#)).

The *Waterkeeper* case does require the EPA to set a best conventional technology standard for pathogen reduction ([Centner 2006](#)). In their proposed changes to the CAFO regulations in response to the court decision, EPA has determined that it will not add specific conventional technology standard expectations for pathogen reduction. In the CAFO rules, the EPA states, 'the magnitude of the human health risk from pathogenic organisms that directly originate from CAFOs and are transported through US waters has not been established' ([United States Environmental Protection Agency 2003](#)). Other risks associated with food-animal waste, such as antimicrobial resistance, and pharmaceutical compounds present in the waste, are not considered in the CAFO rules; the EPA states that the environmental impacts associated with antimicrobials and non-metabolized drugs in animal waste are not clear ([United States Environmental Protection Agency 2003](#)).

Waste management strategies

[Table 3](#) presents recommendations found in the literature for improving the management of food-animal waste. Some of these recommendations have been imposed at the state or sub-state level (i.e. county or natural resource district) ([Metcalf 2000](#)). The benefits and limitations of each are briefly addressed here.

Waste treatment

As previously mentioned, most waste from food animals is collected and stored for an undetermined period of time either as wastewater or solids (in the case of poultry).

Table 3 | Recommendations extracted from peer-reviewed articles, books and reports for improving animal waste management

Type	Recommendation (reference)
Production	<ul style="list-style-type: none">• Limit use of feed additives that adversely affect the quality of waste produced (antibiotics, hormones and arsenicals) (Refsdal 2000; World Health Organization 2002; Nachman <i>et al.</i> 2005; Nachman <i>et al.</i> 2008; Silbergeld & Nachman 2008)• Remove feed ingredients that may include animal tissue, faecal matter or contaminated water from other animals (Gilchrist <i>et al.</i> 2007; Thorne 2007)• Handle waste in dry form to decrease volume of waste and prevent spills (Szögi & Vanotti 2003; TetraTech 2004)• Limit co-location of swine and poultry facilities to reduce risks arising from new strains of avian influenza (Gilchrist <i>et al.</i> 2007; Thorne 2007)• Provide clean, less stressful living conditions, such as hoop housing for swine, to reduce disease susceptibility (Lay <i>et al.</i> 2000)• Treat waste and wastewater prior to discharge (Pell 1997; Hill 2003; Gilchrist <i>et al.</i> 2007)• Increase storage time of waste to destroy pathogens (TetraTech 2004)• Develop institutional and regulatory policies to limit the global transformation of the livestock sector into countries with weak public health, occupational and environmental standards (Delgado <i>et al.</i> 1999)
Research & Monitoring	<ul style="list-style-type: none">• Monitor private wells, streams and aquifers located in regions densely populated by food animals (Kellogg 2000; Burkholder <i>et al.</i> 2007)• Require reporting of all pharmaceuticals used at AFOs (Mellon <i>et al.</i> 2001)• Use environmental assessment tools such as cumulative risk index analysis (CRIA) to systematically assess AFO impacts (Osowski <i>et al.</i> 2001)• Continue to develop reliable, sensitive and affordable methods for the detection of food-animal pathogens in environmental samples (Sobsey <i>et al.</i> 2004)• Determine better indicator microorganisms that can serve as reliable surrogates for pathogen levels in animal waste (Sobsey <i>et al.</i> 2004)• Gather performance data on technologies for pathogen removal (Gerba & Smith 2005)• Gather more data on the fate and transport of parent compounds, metabolites, as well as microbes in soil, water and air (Gerba & Smith 2005; Burkholder <i>et al.</i> 2007)• Improve estimates of impacts from flooding (Wing <i>et al.</i> 2002)• Improve data on AFOs (e.g. map locations, size, environmental conditions, type of waste management systems in place) (Wing <i>et al.</i> 2002; Hodne 2005)• Prioritize watersheds in terms of vulnerability to food-animal waste impacts (Kellogg 2000)• Increase studies of ecosystem health in proximity to CAFOs (Burkholder <i>et al.</i> 2007)• Require public notice and public hearings when a facility is proposed for construction (Metcalf 2000)• Identify and prioritize contaminants that are most significant to environmental and public health (Burkholder <i>et al.</i> 2007)
Environmental Modifications	<ul style="list-style-type: none">• Stabilize the number of animals produced regionally or in a watershed (Westhoek <i>et al.</i> 2004)• Place liability of animal waste management on intermediary owners of animals (i.e. integrators) (Moore <i>et al.</i> 1995)• Develop and implement an oversight and enforcement plan of AFOs (Government Accountability Office 2003)• Use solid tanks or reservoirs in place of earthen waste lagoons (Gilchrist <i>et al.</i> 2007), and regularly test storage tank performance (Meyer & Mullinax 1999)• Create market incentives to improve waste management (Hodne 2005)• Implement alternative pasture management strategies, such as riparian buffers, cover crops, manure injection and no-till/low-till farming to reduce pollution (Chesapeake Bay Foundation 2004)• Apply manure before or during tillage (Moore <i>et al.</i> 1995)• Apply manure in warm months to augment pathogen die-off (Guan & Holley 2003)• Subsidize the cost of transporting animal waste off-site (TetraTech 2004)• Establish a moratorium on new CAFOs (Hodne 2005)• Develop vegetation buffers (i.e. trees & shrubs) around AFO facilities to mitigate air emissions (Tabler 2004)• Create alternative, non-polluting uses for manure, such as restoring abandoned mine lands (Chesapeake Bay Foundation 2004)• Require escrow accounts or insurance policies to cover cost of spills and ensure restoration of vacated waste storage areas (Gilchrist <i>et al.</i> 2007)

Research suggests that dry handling systems can be an effective method of treatment, as these systems can eliminate human pathogens, stabilize nitrogen and limit runoff (Petit & Vanderwerf 2003).

Processes for treating food-animal waste include: air-drying, composting, storage, lime stabilization, anaerobic digestion, aerobic digestion and facultative lagoons (Gerba & Smith 2005). Applications of these waste-treatment technologies have been studied for only a few indicator microorganisms, and data are currently insufficient for determining a safe level of treatment. Pathogenic microorganisms are found to survive in liquid waste storage ponds unless additional stabilization ponds are added in a series or other modifications are included (Hill 2003). Constructed wetlands have been identified as a potential biological treatment technique, but it is unclear whether wetland vegetation can tolerate the high nutrient loads in flushed waste (Hill 2003). Other techniques, commonly used at wastewater treatment plants, have been used for treating animal waste; interest in anaerobic digestion, for example, has grown due to its potential to generate methane for energy recovery (Hill 2003). Anaerobic digesters, however, are difficult to operate, are expensive and do not sufficiently reduce microbial loads, except under highly controlled conditions. Aeration and the addition of chemicals for disinfecting the waste have also shown promise, but data on cost-effectiveness are scarce (Hill 2003).

Managing waste in a dry form, such as on-farm composting, if carried out under more controlled conditions, may be an effective method for treating waste (Hutchison *et al.* 2005). This method has the potential to raise the temperature of the waste and greatly reduce the number of enteric microorganisms. This option may decrease the likelihood of seepage associated with lagoons and helps control odours associated with anaerobic conditions of flush-and-discharge systems. Moreover, composting can stabilize nutrients in the waste. These systems, however, require proper management to ensure optimal conditions and prevent ammonia emissions.

Research studies are beginning to evaluate how waste treatment may affect the degradation of pharmaceuticals in the waste (Sarmah *et al.* 2006). Given the variability of waste-treatment strategies and associated conditions, with

respect to soil type and pH, redox conditions, temperature, bacterial populations, moisture and other factors, little can be gleaned from what has been observed under controlled laboratory experiments (Sarmah *et al.* 2006). For example, Haling-Sorensen *et al.* have found degradation half-lives of two antimicrobials, chlortetracycline and tylosin, to be on the order of four to ten weeks in field experiments in contrast to laboratory studies that observed much shorter half-lives (Haling-Sorensen *et al.* 2005; Sarmah *et al.* 2006). Moreover, little information regarding the formation of metabolites from veterinary pharmaceuticals is available (Sarmah *et al.* 2006).

Location of operations

Contamination of surface waters has been found to be associated with the regional concentration of food-animal production (Weldon & Hornbuckle 2006). In the state of Iowa, whose rivers have among the highest nutrient concentrations in the central US, more than 3,800 CAFOs exist and are increasingly becoming more spatially concentrated (Weldon & Hornbuckle 2006). The Environmental Quality Board of the state of Minnesota has begun to conduct environmental impact studies on food-animal operations and is gathering information on location, size, species and number of animals as well as other characteristics that include population, population density, land use, water resources, groundwater sensitivity and land base (Schmidt 1999). Knowing the number of food animals at both the regional and operation level and knowing the location of waste application fields could greatly assist in mitigating the impacts. Environmental assessment tools, such as cumulative risk index analysis and use of GIS technology, are being developed and validated to provide a more systematic and robust approach to assessing the regional or watershed-based impacts of CAFOs (Osowski *et al.* 2001). These tools have the potential to increase the transparency of decisions and facilitate communication of information to stakeholders (Osowski *et al.* 2001). Further, these technologies help ensure that sufficient land is available for waste application from any particular animal-feeding operation. Unfortunately, these tools are not generally used for small and medium AFOs and remain underutilized, especially for follow-up monitoring (Osowski *et al.* 2001).

In an economic analysis of livestock waste regulation, Innes notes that current federal animal waste-handling standards result in incentives for integrators to concentrate an excess number of animals in facilities that are too large (Innes 2000). With these larger facilities that house higher numbers of animals, additional environmental costs, such as heightened potential for manure storage spills, nutrient over-application and introduction of waste-borne pathogens, are increased. Since integrators are not responsible for these costs, there is no incentive for them to incorporate these considerations into the livestock facility siting, planning and operation. Further, many of these costs are difficult or impractical to routinely measure. To address this, Innes suggests that improvements could be made by regulating producer choices that are known to impact the environment through waste management, such as facility size, entry and location, rather than directing regulatory attention only to waste-management procedures.

Imposing severe restrictions, such as moratoria, on operations of certain sizes or operations exceeding a certain level of production has been used to limit regional impacts. These are intended to limit the expansion of the industry and may occur locally or statewide. Implementation of a moratorium is typically a temporary effort to slow growth and allow more time to gather data and measure impacts. An eighteen-month moratorium on new CAFOs in North Carolina was first passed in 1997 (Mallin & Cahoon 2003) and has been repeatedly extended, most recently in 2007, to last until 2010 (General Assembly of North Carolina 2007). Despite the moratorium, production has continued to increase by expansion of existing farms, and the state is now estimated to produce 10,000,000 head of swine annually, the vast majority in CAFOs (Mallin & Cahoon 2003). Minnesota set local moratoria to limit the geographic concentration of facilities and Arkansas has imposed limits on the number of permits to be issued for animal feeding operations (Metcalf 2000). Other recent attempts to enact moratoria, often at the county level, have been met with mixed success (Farnese 2003; Hundt 2007; City Council of Galena 2008). In 2003, the American Public Health Association issued a policy statement calling for precautionary moratorium on all new CAFOs until adequate data to support evaluation of their potential public health risks has been collected (American Public Health Association 2003).

Other medical and professional societies have also issued statements supporting CAFO moratoria (American Public Health Association 2003). Literature documenting the effects of these moratoria is limited, making it difficult to assess their impacts; however, since moratoria are intended only to be temporary cessation of growth, it is unlikely that these measures can be effective in addressing public health risks.

Feed inputs

At the pre-production end, the removal of non-therapeutic uses of antibiotics, arsenicals, hormones and other feed additives is considered important for reducing public health risk (Refsdal 2000; Chee-Sanford *et al.* 2001; Mellon *et al.* 2001; World Health Organization 2002; Nachman *et al.* 2005; Sapkota 2007a; Nachman *et al.* 2008; Silbergeld & Nachman 2008). Limiting these feed additives will likely decrease the hazards associated with land application of the waste. With respect to growth-promoting antibiotics, the European Union, in 1999, banned their use (Casewell *et al.* 2003). Researchers have noted that since the ban on GPAs in the EU, significant drops in levels of resistance have been observed in meat products, faecal samples of food animals and healthy humans (van den Bogaard & Stobberingh 2000). In the US, legislation to ban GPAs in food-animal production has been proposed but not adopted.

Alternative uses

Alternative uses of food-animal waste have been developed that include energy production, feedstock (i.e. adding animal waste into the feed of other animals) and creating commercial fertilizer. In integrated fish farming, mostly in less economically developed countries, poultry litter is often added to fish ponds either for direct consumption of the litter by fish, or for the nutrients in litter that support the growth of photosynthetic organisms, which are subsequently eaten (Petersen *et al.* 2002). Many agricultural researchers are focused on solutions that will result in value-added products derived from animal waste. Many of these uses, however, face barriers due to microbial and chemical contaminants. The use of organo-arsenicals in poultry and swine feed, for example, that end up in the waste, present a barrier to use as a commercial fertilizer or

for energy production, as arsenic is not degraded (Nachman *et al.* 2005; Nachman *et al.* 2008). Currently, the value of alternative uses of waste is estimated to be low, and researchers suggest that application to nearby cropland or forests remains the highest value use (Lichtenberg *et al.* 2002).

Total maximum daily load agreements and other options

States are required to develop a total maximum daily load (TMDL) as a second line of defence for protecting bodies of water that do not meet water quality standards (United States Environmental Protection Agency 2003). A TMDL is a calculation of the maximum amount of a particular pollutant that a water body can receive and still meet water quality standards. In 1997, the state of Delaware, where intensive poultry production takes place, entered into a TMDL agreement with the EPA as a result of a lawsuit by environmental groups against the EPA. In the agreement, Delaware was mandated to reduce nitrogen and phosphorous runoff into surface waters by 60–85% (Sims 1999). In some states, TMDL regulations may increasingly be used to establish quantitative standards for microbiological or chemical contaminants from AFOs. It should be noted, however, that for non-point sources of pollution such as farms, the TMDL programme does not provide an independent authority to enforce load reductions and relies on a state's efforts (Ruhl 2001).

More recently, the EPA has promoted the trade of animal waste between watersheds as a way to increase flexibility and improve water quality (Hegg 2006). Some states and watershed groups have set up trading between wastewater-treatment plants in order that those with advanced treatment processes can offset discharges by other less efficient plants. This has not worked for food-animal producers because of the difficulty in quantifying the amount of non-point source pollution. The US Department of Agriculture is currently working on trading policies that may improve its future potential (Hegg 2006).

A few states have moved to require bonding or insurance policies to cover the costs of manure spills (Metcalf 2000). These requirements also generally cover the restoration costs of vacated waste lagoons or waste systems. These are important changes that could help

relieve municipal governments or communities of the financial burden of clean up and reclamation.

Non-point source management

Management practices such as creating or restoring riparian buffer zones or developing on-farm vegetative treatment systems have mostly been studied for removal of nutrients and are generally based on non-CAFO applications (Koelsch *et al.* 2006). In a review of 32 studies evaluating vegetative treatment systems for managing runoff from 25-yr, 24-h storm events, only seven small-scale studies evaluated removal of bacteria. Performance of the systems varied greatly with bacteria removal ranging from 15 to 100% (Koelsch *et al.* 2006).

The US Department of Agriculture and the Environmental Protection Agency have set a goal for all operations raising animals for food to develop and implement management plans based on nutrients. The plans include: proper waste handling and storage, use of proper land application rates, improved site management (such as creating riparian buffer zones); and record keeping. Where insufficient land is available for waste application, other alternatives such as transporting waste off-farm have been suggested. Current evidence indicates that nutrient-based standards are incapable of being met. One study found that 20% of very small AFOs did not have sufficient land to apply waste versus 90% of CAFOs (Gollehon *et al.* 2001). In Chesapeake Bay watersheds, the USDA estimates that in order to transport all of the excess waste from the area, 60% of all available cropland would need to utilize waste, versus the current 10–20% (Ribaldo *et al.* 2003a), and given human population growth in this region, that level of waste utilization is unlikely. Further, transporting waste is a costly endeavour (Ribaldo *et al.* 2003a; Collins & Basden 2006). A study in Alabama using phosphorous as a limiting nutrient in poultry litter found that the break-even distance for which poultry litter could be exported to other counties would not provide sufficient economic benefits to set up a system of exports without subsidies (Paudel *et al.* 2004).

Additional research

In 2002, funding for research on food-animal waste management received an estimated 1.7% of the funds

available for animal agriculture research (Hegg 2006). More support for research will be vital to improve the current state of food-animal waste management.

Research includes: improved methods to assess hormone activity and the effects of other pharmaceuticals (Lorenzen *et al.* 2004); developing a 'manure related infectious disease incidents database'; identifying treatment technologies to reduce pathogens in animal waste; and improving methods for microbial source tracking (Smith & Perdek 2004). Additionally, a better understanding of how operation size or the regional concentration of animals affects environmental quality is desperately needed as we move towards more large operations and greater regional concentration of production. Efforts to monitor and evaluate the effect of the CAFO rule on food-animal waste management and environmental quality should be augmented.

Further, research into how housing conditions affect pathogen transfer and disease rates among animals is needed. While it has been recognized that crowding, inadequate housing and unsanitary conditions facilitate the spread of infectious disease in human populations, this knowledge has not been sufficiently transferred to industrial food-animal production that concentrates animals in small, unsanitary spaces (Krieger & Higgins 2002). For example, research has demonstrated that hogs raised in non-bedded confinement systems exhibit more aberrant behaviour, have higher plasma cortisol levels and suffer a greater incidence of injuries in contrast to hogs in bedded hoop housing (Lay *et al.* 2000).

The microbiological quality of water is generally assessed using indicator organisms because they are relatively easy to measure and have been considered appropriate for routine surveillance. Indicators include faecal coliforms, total coliforms, *E. coli* and enterococci. More recently, however, the use of indicators for assessing public health risks has been judged inadequate. In fact, a growing body of literature shows poor correlations between the level of pathogenic microorganisms in the environment and the level of indicator organisms (Chauret *et al.* 1995; Johnson *et al.* 2003; Dorner *et al.* 2004). Further, very little research has been done on virus runoff from fields where waste is applied and virus leaching from food-animal waste pits. Additional research and improved microbiological

detection methods are needed; both will assist in determining the sources and extent of faecal pollution. The ability to link animal-waste contamination to human exposures is limited and will depend greatly upon improved methods for tracking non-point sources of faecal pollution (Bigras-Poulin *et al.* 2004). Microbial source tracking shows promise for improved precision in determination of waste origin, though this field is in its early stages, and no universally accepted methodology has emerged (Stoeckel & Harwood 2007). Research aimed at validating these methodologies is ongoing (Foley *et al.* 2009; Harwood *et al.* 2009).

In order to determine which long-term waste-management strategies would be most effective, additional studies in a variety of settings are needed. These studies will require substantial financial and analytical resources and will necessitate increased collaboration between schools of public health and agriculture, as well as other disciplines. However, sufficient evidence already exists to reduce potential adverse environmental and public health consequences from animal waste. These incremental steps, in tandem with larger-scale, longer-term re-evaluation of waste management strategies, have the potential to ensure that animal waste can be reused without posing downstream threats.

CONCLUSIONS

Food-animal wastes can be a resource; however, like many wastes, their volume, content and distribution have changed dramatically. In addition to nutrient management, there is a deep need to address other aspects of food-animal waste. In this discussion, the various management strategies are placed into one of the following categories: 1) improving the quality of the waste, 2) reducing the volume of waste and 3) changing the geographic distribution of waste. Some strategies may affect more than one category. For example, treating the waste can both reduce pathogens (i.e. improve the quality of the waste) and reduce the volume of the waste.

There is a critical need to develop and implement standards for treating food-animal waste and reduce zoonotic hazards before waste is applied to land. Currently, no standards exist for the roughly 335 million tons of

food-animal waste applied to land in the US; in contrast, standards are in place for biosolids, despite the fact that they account for roughly 1% of all wastes applied to land (United States Environmental Protection Agency 2003). Treatment of the waste is especially vital due to the increased geographic concentration of food-animal production. In peri-urban areas of Asia, Africa and Latin America, where human populations are densely populated and safeguards for public health and the environment are lacking, food-animal waste treatment is essential.

The quality of food-animal waste is inextricably linked to the feed. The use of feed additives for growth promotion, including antibiotics, hormones and arsenicals that in turn end up in the waste, is short-sighted and limits the potential for recycling the waste (Emborg *et al.* 2001; Miller *et al.* 2003). Moreover, the use of sub-therapeutic antimicrobials increases the likelihood that antimicrobial resistant bacteria are present in the waste (Ramchandani *et al.* 2005). Further, research has shown that improving farm management and hygiene can boost animal productivity exclusive of the use of growth-promoting antibiotics (Miller *et al.* 2003).

Changing the diets of animals to decrease the excretion of nutrients, in the case of phytase, is an unlikely solution to decrease the amount of waste produced. One of the most basic measures for reducing the volume of waste is to not add water. Flush-and-discharge systems that add water to animal waste intensify the challenges of managing animal waste. These systems increase the volume of waste requiring treatment and augment the potential for spills and leaks that may contaminate both surface water and groundwater. Moreover, dry handling of waste conserves water and can be relatively efficient for reducing pathogenic microorganisms. Using a dry waste management system is especially important to food-animal operations near densely populated urban centres in regions that do not have the institutional capacity to effectively control pollution associated with flush-and-discharge waste systems.

Changing the distribution of where waste is applied may be the biggest obstacle for adequate management as many food-animal operations already exist in regions where food-processing facilities are present. Moreover, it is predicted that domestic production will continue to grow as it is unlikely that production will move abroad

(Cummings 2000). A more likely scenario is that food-animal production will grow in states with less regulatory oversight. In some cases, zoning ordinances, developed by local communities attempting to regulate the siting of AFOs, have been pre-empted by state laws with less stringent standards (Sikora 2002). For the foreseeable future, subsidies for transporting waste from regions where food animals are produced will be necessary, which could have detrimental effects if the waste has not been treated to reduce zoonotic pathogens. More needs to be done to ensure that new operations are placed in areas that have sufficient land for applying treated waste at agronomically sustainable levels before construction is allowed.

The burden of waste management must be shared with integrators, otherwise increased regulations may fall solely upon farmers and further exacerbate the loss of farms, further concentrating the industry (Ruhl 2001). Although much emphasis on technology-based solutions has been suggested, without the adoption and reinforcement of appropriate regulations, such technologies will not be implemented by producers and farmers due to the lack of enforcement or other incentives.

The EPA's revised CAFO rules are likely to be ineffective at adequately controlling waste from industrial food animals based on limited oversight and focus on nutrients only (United States Environmental Protection Agency 2003). It will be vital for organizations concerned with environmental and public health impacts to measure changes and highlight performance of the regulations. As a result of the *Waterkeeper* case, information on CAFO nutrient-management plans and details from NPDES permits will be made available. With the availability of this information, additional opportunities for citizen participation and law suits may arise (Centner 2006). Without other means for controlling the siting of food-animal operations, it is likely that lawsuits, an expensive form of regulation, will grow.

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First received 17 May 2009; accepted in revised form 16 March 2010. Available online 8 June 2010