CHAPTER FIVE

A Review of the Use of Organic Amendments and the Risk to Human Health

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Advances in Agronomy, Volume 120
ISSN 0065-2113, http://dx.doi.org/10.1016/B978-0-12-407686-0.00005-1

Abstract

Historically, organic amendments—organic wastes—have been the main source of plant nutrients, especially N. Their use allows better management of often-finite resources to counter changes in soils that result from essential practices for crop production. Organic amendments provide macro- and micronutrients, including carbon for the restoration of soil physical and chemical properties. Challenges from the use of organic amendments arise from the presence of heavy metals and the inability to control the transformations required to convert the organic forms of N and P into the minerals available to crops, and particularly to minimize the losses of these nutrients in forms that may present a threat to human health. Animal manure and sewage biosolids, the organic amendments in greatest abundance, contain components that can be hazardous to human health, other animals and plants. Pathogens pose an immediate threat. Antibiotics, other pharmaceuticals and naturally produced hormones may pose a threat if they increase the number of zoonotic disease organisms that are resistant to multiple antimicrobial drugs or interfere with reproductive processes. Some approaches aimed at limiting N losses (e.g. covered liquid or slurry storage, rapid incorporation into the soil, timing applications to minimize delay before plant uptake) also tend to favor survival of pathogens. Risks to human health, through the food chain and drinking water, from the pathogens, antibiotics and hormonal substances that may be present in organic amendments can be reduced by treatment before land application, such as in the case of sewage biosolids. Other sources, such as livestock and poultry manures, are largely managed by ensuring that they are applied at the rate, time and

place most appropriate to the crops and soils. A more holistic approach to management is required as intensification of agriculture increases.

ABBREVIATIONS

B boron
C carbon
CO_2 carbon dioxide
Cd cadmium
CEC cation exchange capacity
Cr chromium
CFU colony forming units
Cu copper
CH_4 methane
C:N carbon to nitrogen ratio
ED_{50} effective concentration in the soil to achieve 50% kill of the target organism
Hb hemoglobin
HCO_3^- bicarbonate ion
H_2S hydrogen sulfide
K potassium
MetHb methemoglobin
Mg magnesium
Mo molybdenum
N nitrogen
NH_3 ammonia
NH_4^+ ammonium
NO nitric oxide
NO_3^- nitrate
NO_2^- nitrite
NO_2 nitrogen dioxide
N_2O nitrous oxide
OH^- hydroxyl ion
P phosphorus
Pb lead
PFU plaque forming units
PMB paper-mill biosolids
SO_2 sulfur dioxide
Zn zinc

1. INTRODUCTION

Chinese records dating back more than 2000 years report the use of organic manures in agriculture (Beaton, 2009). The Greeks applied human sewage to fields and recognized, as did the Romans, that animal and plant residues could be applied for the benefit of crop growth, the amounts required varying according to plant and soil type (Beaton, 2009). In late medieval times, it was understood that the use of animal manure was to replace materials removed by the cultivation of crops (Wild, 1988). The "first factory for the manufacturing of synthetic manure," actually super phosphate, began business in 1842 (Allison, 1944; Dyke, 1993). Although guano—accumulations of the droppings of seabirds or the dung of bats—and some sodium nitrate mined in South America were commercially available as sources of N, at that time the main source of plant nutrients applied to crops in Europe and North America was farmyard manure (Beaton, 2009). This continued to be the position until the dramatic increase in availability of N fertilizers that began in the early 1950s. However, organic amendments continued to be the only form in which nutrients were added to soils in resource-poor regions (Thomas et al., 2006) and animal producers worldwide still spread manure and land application is an important means of disposal for sewage biosolids (Kelessidis and Stasinakis, 2012).

Although the primary reason for applying organic materials to agricultural land may be to recycle plant nutrients back to the soil, the return of organic matter also has importance. The decline in soil organic matter associated with the long-term cultivation of land (Jenkinson, 1991) reduces soil productivity because of deterioration in physical and chemical properties (Rovira, 1995). With the importance of carbon sequestration in soil as one means of combating climate change, there is good reason to take a holistic view of the use of organic sources of nutrients and consider their importance in sustainable food production. Here we review the benefits associated with the application to the land of organic materials, particularly animal manure and sewage biosolids, in support of the plant growth together with the possible threats and risks to human health that can result from their use (Table 5.1). We have chosen to focus on the threats to health from the direct use of amendments leading to contamination of air, water and food crops rather than the broader consequences associated with altered ecosystems impacting food availability (Camargo and Alonso, 2006).

2. TYPES OF ORGANIC AMENDMENTS APPLIED TO SOILS

While there is a very large number of organic amendments applied to soil in support of crop production, they essentially fall into six major categories.

Table 5.1 Benefits and concerns related to the use of organic amendments for plant production

Benefit	**Concerns**
Source of essential macronutrients for plant growth	Imbalance of nutrients applied to soil; the possibility of losses of P and N compounds in runoff to surface water and leaching to ground water resources; release of greenhouse gases and contributions to acid rain
Provision of micronutrients	Buildup of heavy metals in soil
Enhancement of soil carbon levels	Release of antibiotics and endocrine-disrupting compounds into the environment
Increased soil microbial activity	Release of zoonotic pathogens into the environment
Improved soil aggregation	
Reduced soil bulk density and compactibility	
Improved soil permeability	Increased risk of the preferential flow of contaminants to water resources
Greater root exploration of soil	
Improved water-holding capacity	
More resilience against erosive forces of wind and water	

2.1. Animal Manure

Farmyard manure is a combination of feces, urine and animal bedding that is stacked and turned to undergo some level of composting. If the piles are not turned and new material from the barn is added regularly, solid or semi-solid manure results. In the 1950s, efforts to reduce labor costs and improve hygiene in dairy barns resulted in bedding-free stalls and the removal of manure to a holding tank together with wash water. The liquid or slurry collected can then be applied to the land. Liquid manure systems have since been developed for other livestock and poultry. Beef and dairy operations create the greatest amount of manure, followed by pork units. Poultry production systems (broilers for meat and layers for eggs) make a much smaller contribution to total manure production than cattle or pig.

The majority of world manure production is land applied. For example, of the manure collected from confined animals 84% is spread on cropland and 16% on grassland (Beusen et al., 2008). It is estimated that on a worldwide basis domestic animals excrete about 112 Tg N per year.

Approximately 42% of this comes from confined animals, 44% from animals grazing in the field with 14% being excreted outside the agricultural system or contributes to other uses, such as heating fuel or animal feed (Beusen et al., 2008).

Manure is subject to change from the time that urine and feces are excreted and mixed, through periods of temporary storage in the barn and for the whole time they are in longer-term storage prior to application to land. As the enzyme "urease" is both widespread and somewhat resilient, mineralization of urea from the urine begins very rapidly after excretion even if it is not mixed with feces. The essential products are NH_4^+, HCO_3^- and OH^-. The latter two ions increase the pH, which changes the balance between NH_4^+ and NH_3 in solution, leading to the release of NH_3 into the atmosphere. Although the same processes occur with poultry excreta, which are a combination of materials from the digestive tract and the kidneys, there is less loss of NH_3 to the atmosphere because the material is much drier, unless water is added.

When liquid or slurry manure gets put into longer-term storage, the settling of materials can result in the formation of a number of layers, the properties of which vary considerably in space and time (Patni and Jui, 1987). Aerobic decomposition of manure organic matter results in CO_2 and organic carbon compounds that are more resistant to breakdown. However, one consequence of stratification is that anaerobic conditions develop within the stored manure. When free oxygen is not present, organic matter is converted to C compounds of small molecular weight, mainly volatile organic acids, and CH_4 gas is released. In addition, the breakdown of proteins can lead to the formation of H_2S. The formation of volatile fatty acids, a readily available carbon source for microorganisms, leads to a reduction in pH of the manure.

If solid manure is turned frequently and not tightly packed, the breakdown of organic matter takes place under aerobic conditions. The rate of the process is faster under aerobic than anaerobic conditions and more C is lost as CO_2. In part, this is because aerobic breakdown releases more energy, which in turn supports about 5.5 times more microbial biomass per unit of organic substrate consumed than anaerobic fermentation.

2.2. Municipal Biosolids and Septage

The sludge from municipal wastewater treatment operations is commonly applied to agricultural land but this is usually subject to regulatory control. After general screening and grit removal, suspended organic material is separated out by sedimentation (Primary Treatment) and the microbes present digest the easily metabolized fraction (Secondary Treatment). Some treatment plants remove N and P in solution (Tertiary Treatment). The final treatment stage is to stabilize the material, using heating and drying, alkaline stabilization to increase the pH, aerobic or anaerobic digestion together with heating, or composting. The stabilization process is intended to reduce or eliminate pathogens and make the material less attractive to scavengers after land application, which is either as a liquid, or as a cake following dewatering. These materials are commonly referred to as municipal or sewage biosolids. The projected production of sewage biosolids in 2010 for the USA and the original 15 countries of the European Union is about 16 Tg dry matter per year (Epstein, 2003). Although not all European countries use sewage sludge in agriculture, over the period 1996–1998 France applied 60% of its production to agricultural land and comparable values for Spain and the UK are 46%, Germany 40% and Italy 16%. Typically, between 50 and 70% of sewage biosolids produced are land applied (Epstein, 2003). For a comparison of the current treatments and approaches to the disposal of sewage biosolids in Europe, the reader is referred to Kelessidis and Stasinakis (2012).

In many rural areas, where piped water and sewage systems are not available, on-site wastewater treatment systems, such as septic systems, have temporary holding tanks that need to be pumped out, periodically. Although many jurisdictions require the resulting septage to be handled through wastewater treatment plants, some still allow untreated material to be applied directly to land.

2.3. Green Manure and Crop Residues

In addition to the manure from animal production systems, by-products of other agricultural activities including residues from the harvest have been spread on land with or without prior treatment. Examples are straw from grain production and stubbles that are plowed in after harvest. Total dry matter production of cereal crops worldwide is about 5000 Mt, of which about 2600 Mt is straw (Wirsenius et al., 2010), which may be incorporated in the soil, used as animal feedstuff or as bedding material for housed animals. Increased urbanization has provided other waste streams that contain plant nutrients, such as garden waste.

Green manure results from the incorporation into soil of any field or forage crop, while green or soon after flowering, for the purpose of improving soil chemical and physical fertility. An important feature is the transport of bases from lower layers of the soil into the topsoil. A summer green manure crop occupies the land for a portion of the growing season (usually after the main crop is harvested). Warm-season cover crops can be used to fill a niche in crop rotations, to protect fragile soils, prepare land for a perennial crop or provide some additional animal feed. Legumes such as cowpeas (*Vigna unguiculata* (L.) Walp.), soybeans (*Glycine max* (L.) Merr.), annual sweet clover (*Melilotus indicus* (L.) All.), *Sesbania* spp. Scop., *guar* (*Cyamopsis tetragonoloba* (L.) Taub.), *Crotalaria* spp. L., or velvet bean (*Mucuna pruriens* (L.) DC.) may be grown as summer green manure crops to add N along with organic matter. Non-legumes such as millet (*Panicum miliaceum* L.), forage sorghum (*Sorghum bicolor* (L.) Moench), annual ryegrass (*Lolium multiflorum* Lam.), brassicas (Brassica spp. L.) or buckwheat (*Fagopyrum esculentum* Moench) are grown to provide biomass, smother weeds, and improve soil tilth. Some benefit can accrue from the growing crop, particularly from the root system, but in green manuring, the sought-after impacts result from incorporation of the shoots.

In tropical systems, farmers use a wide array of crops as green manure to replenish nutrients. Palm et al. (2001) have established a database of research results (published and unpublished) on the species used as green manure, mostly from the east and southern Africa. Over half of the entries were members of the *Fabaceae* with the second most abundant group being species within the *Asteraceae*.

2.4. Food Residues and Waste

Unsold or unsalable fresh produce from supermarkets are other sources of material from urban centers that have been applied to agricultural land, frequently after composting. A survey in the U.S. showed that in 1995 more than 43 Mt edible foodstuffs were lost at the retailer, foodservice and consumer end of the supply chain. Some 6.6 Mt of grain products were lost prior to consumption as were 7.2 Mt of vegetables—corresponding to 32% and 25% respectively of the total supplied (Kantor et al., 1997).

2.5. Waste from Manufacturing Processes

Residual organic material from pressing oil seeds, hoof and horn meal, fish offal, dried blood, animal hair, feathers and shoddy (waste material from preparing wool) have all been applied in this way (Russell, 1946). Large amounts of biosolids are produced annually as the residues from papermaking (paper-mill biosolids—PMB) but only a small fraction of this material is land-applied (Thacker, 2007). Other residues, such as those resulting after sugar extraction from sugar beet (*Beta vulgaris* L.), oil from olives (*Olea europaea* L.), as well as distillery waste have also been spread on cropped land or investigated for their ability to support crop production (Douglas et al., 2003; Hachicha et al., 2012; Kumar et al., 2009).

2.6. Compost

Composting is one of the commonest treatments to stabilize the N in wastes and improve their handling characteristics, particularly by reducing the volume of organic materials that are to be land-applied. For example, during the composting of solid manure to form farmyard manure as much as 50% of the C may be lost. Typically, material with a large C:N ratio is mixed with waste that is rich in N. The product of composting is a material that has a much smaller C:N ratio than the original mixture. Some of the C, N and P in the raw materials are converted into the microbial biomass that contributes to the composting process.

The activity of the microbial population in the raw material encourages a rise in temperature at the center of a pile. Mixing helps to ensure that all the material is heated but it may prevent the temperature from remaining above 55 °C for sufficient time, while leaving the compost unturned allows the temperature at the surface to remain close to ambient. However, it is important to ensure that all the material is subject to temperatures above 55 °C so that harmful microbes and unwanted weed seeds can be killed. This is a difficult requirement to meet in the absence of forced aeration (St. Jean, 1997). The heat generation from the composting material increases evaporation, which can also adversely affect microbial activity if the water is not replenished.

Any loss of N during composting has to be considered against the potential benefits. Kirchmann and Lundvall (1993) concluded that aerobic composting should not be used to treat amendments containing high levels of NH_4^+–N as it resulted in too great a loss by NH_3 volatilization. For example Ramaswamy et al. (2010) reported a 60% loss of N but only a 2% loss of C from loose-piled poultry manure and hay (initial C:N ratio of 25:1). Beck-Friis et al. (2000) reported significant loss of N as N_2O as well as CH_4 during the composting of household organic waste. Composting of N-rich raw material in the open can increase N-loss through leaching. Kirchmann (1985) reported losses of N from composting manure ranging between 2% and 10% of the NO_3^-–N in the windrow, although NO_3^-–N concentrations generally did not exceed 0.05% of dry matter. During composting about 50% of the organic matter, 40% of K and 20–30% of N in manure can be lost if manure is windrowed without covering (Lampkin, 1990;Vogtmann and Besson, 1978).The net result of composting is a reduction in readily mineralizeable organic N compounds.

Composting can result in considerable reduction in the concentrations of some antibiotics (Dolliver et al., 2008) (see Section 4.3). However, in contrast to N and antibiotics, there is a tendency for Cu, Zn, K, and P to increase in concentration during composting (Ott and Vogtmann, 1982; Tejada et al., 2001).

3. THE NUTRIENT VALUE OF ORGANIC AMENDMENTS

By physiological maturity, crop plants need to have accumulated N and K to an amount ranging between 1 and 4% of their shoot dry weight. Phosphorus contents range from 0.2 to 0.4% of shoot dry weight. Leguminous grain or forage crops tend to contain more N and P than do cereals (Table 5.2). Organic amendments are commonly applied to the soil to meet these mineral requirements. However, whether they are derived directly from plant material or indirectly, as in the manure from livestock, or from

Table 5.2 Approximate amounts of macronutrients present in some agricultural crops at harvest, in various livestock and poultry manures, industrial and municipal biosolids, expressed in g kg^{-1} dry matter

	Nutrient content g kg^{-1}			**Nutrient ratio**
Source	**N**	**P**	**K**	**N:P:K**
Maize (*Zea mays* L.)	13.5	1.8	14.1	7.5:1:7.8
Wheat (*Triticum aestivum* L.)	12.4	1.3	13.8	9.5:1:10.6
Oat (*Avena sativa* L.)	9.0	1.8	16.9	5.0:1:9.4
Soybean (*Glycine max* L. Merr.)	31.7	3.8	15.4	8.4:1:4.1
Alfalfa (*Medicago sativa* L.)	39.9	3.3	29.3	12.1:1:8.9
Liquid manure, dairy cattle	3.5	0.8	2.4	4.3:1:3.0
Solid manure, beef cattle	7.4	2.4	5.7	3.1:1:2.4
Liquid manure, pig	4.0	1.3	1.8	3.1:1:1.4
Solid manure, poultry (layer)	19.3	8.9	8.0	2.2:1:0.9
Primary paper-mill sludge	1.7	0.4	0.6	4.3:1:1.5
Secondary paper-mill sludge	31.3	7.3	4.5	4.3:1:0.6
Municipal wastewater effluent (liquid)	34.0	7.3	1.1	4.7:1:0.2
Municipal wastewater effluent (aerobic)	1.2	0.6	0	2.0:1:0
Municipal wastewater effluent (anaerobic)	2.8	1.4	0	2.0:1:0
Municipal wastewater effluent (cake)	37.6	13.1	1.1	2.8:1:0.08
Composted effluent cake	13.0	24.0	0.5	0.5:1:0.02

mixed sources (e.g. sewage biosolids), organic amendments contain N, P and K in very different ratios to those needed by crops (Table 5.2). Both N and P are usually present in both organic and inorganic forms in organic amendments, whereas K is always as an inorganic cation. However, the proportion of each element in different amendments varies widely (Table 5.3).

The main mineral form of nitrogen, the ammonium ion (NH_4^+), is formed under both aerobic and anaerobic breakdown of N-containing plant and animal constituents (mineralization), while nitrate (NO_3^-) only forms under aerobic conditions by the oxidation of NH_4^+ via NO_2^- (nitrification). The proportion of the total N present in the ammoniacal (NH_3 and NH_4^+–N) form is a key characteristic of organic amendments being considered to be as available as fertilizer N (Beauchamp, 1983; Beauchamp and Paul, 1989), or nearly so (Beauchamp, 1986; Castellanos and Pratt, 1981; Chantigny et al., 2004a; Flowers and Arnold, 1983; Paul and Beauchamp, 1994), and because it can be lost as gaseous ammonia. A helpful overview of NH_3 losses as a result of animal manure application is given in Søgaard et al. (2002). In contrast to inorganic N, the organic N fraction may be only marginally available to plants in the year of application (Beauchamp, 1986). There are also large differences between sources of sewage biosolids in the proportion of organic N that is mineralizable in laboratory tests. Values

Table 5.3 Distribution of N and P forms in some animal manures expressed in g kg^{-1} dry matter

Manure type	Total N g kg^{-1}	Mineral N: NH_4^+–N g kg^{-1}	Mineral N: NO_3^-–N g kg^{-1}	Organic-N g kg^{-1}	Dry matter %
Cattle manure—liquid	1.8	1.3	0.002	0.5	4.2
Cattle manure—composted	5.9	0.07	0.25	5.6	29.9
	Total P g kg^{-1}	**Mineral P* g kg^{-1}**		**Organic P g kg^{-1}**	
Cattle manure	4.1	2.9		1.2	14.0
Pig manure	13.7	12.3		1.4	25.0
Poultry (broiler) litter	15.8	5.6		10.2	84.0

*Mineral P consisted mainly of phosphate but polyphosphate was found in trace amounts in the poultry manure, some in the swine manure but cattle manure contained the greatest proportion.
(Based on data from Goss et al. (1995), Turner and Leytem (2004)).

vary from 4 to more than 60% (Magdoff and Amadon, 1980; Parker and Sommers, 1983; Smith et al., 1998a). Small values are associated with aerobically digested materials and those stabilized by composting. Smith et al. (1998b) were able to categorize sewage biosolids into those that had a high potential to accumulate NO_3^-–N, those with a low to intermediate potential, those that on applying to soil immobilized nitrogen before releasing NO_3^-–N and those where the organic nitrogen was resistant to breakdown.

Organic forms of P are fairly complex molecules (Turner and Leytem, 2004). Many fields that have received regular applications of manure contain large amounts of P. The concentration of P in manure can vary widely according to the dietary intake (Barnett, 1994), in part because of the very different amounts of mineral P that can be provided in feed (Van Faassen and Van Dijk, 1987). The maximum amount of P that is present in the soil solution is related to the natural level of labile P and how much additional P has been applied in organic and inorganic forms (Raven and Hossner, 1993; Sharpley and Smith, 1989).

The action of soil microorganisms is required if the nutrient elements in organic molecules are to become available to crop plants. An important parameter that provides an index of the rate of mineralization of organic materials is the C to N ratio (Stevenson, 1982), with values of 30:1 or less allowing N to be released into the soil. For larger ratios, some N already present in the soil is immobilized before there is a net release (Scarsbrook, 1965). However, studies of the release of N from sewage biosolids have found temporary immobilization of soil N for C:N ratios of 17:1 (Tester et al., 1977) and >20:1 (Parker and Sommers, 1983). The release of NO_3^-–N from organic amendments appears to be a function of thermal time, with a base value of 0 °C (Smith et al., 1998a,b). Soil acidity can also be a factor in governing the rate of N transformations, with faster nitrification occurring when the soil, to which sewage biosolids were added, had a pH of 7.5 than at a pH of 5.3 or 6.0 (Terry et al., 1981).

Another aspect that affects the perceived nutrient value of organic amendments is the heterogeneity in the distribution of nutrients that develops during storage prior to land application. For example, in solid manure, traces of NO_3^-–N can be produced in the surface layer, where gaseous loss of NH_4^+–N also takes place preferentially, so that there is considerable variability in mineral-N concentration with depth in the pile. This variability is not easily overcome, as it is difficult to mix solid manure homogeneously before spreading. Consequently, considerable spatial variability can result in the distribution of nutrients, particularly N, applied to the land. In contrast, even though variability in the content of nutrients develops between strata in stored liquid manure, mixing before spreading is relatively easy so that the material applied to the land can be reasonably uniform in its fertilizer value.

The very different proportions of N and P in organic amendments relative to what is needed for plant growth generally means that if the crop requirements for N are met, the levels of P as well as K and Mg can be excessive (Cela et al., 2010; Edmeades, 2003). If P is used as the basis for the calculation of the load required to meet crop needs, insufficient N will be provided to the crop unless supplemented with mineral fertilizer.

In addition to macronutrients, organic amendments also can supply trace elements, such as Zn, Cu, Cd (Table 5.4) as well as Mo and B. As many soils, especially those in the humid tropics, are deficient in these micronutrients, the use of organic amendments can be important in crop production (Gupta et al., 2008).

Green manure crops vary considerably in terms of their nutrient content and how much becomes available to subsequent crops. The contents

Table 5.4 Ranges of metal contents of manure

Animal category	Manure type	Copper mg kg^{-1} (dry-matter basis)	Zinc mg kg^{-1} (dry-matter basis)	Cadmium mg kg^{-1} (dry-matter basis)
Cattle				
Dairy	Liquid	1.0–352 (55)	<5–727 (266)	<0.08–3.2 (0.31)
Beef	Liquid	12–267 (49)	88–938 (277)	<0.08–0.80 (0.32)
Dairy/Beef	Solid	2.5–80 (35)	40–448 (181)	0.04–3.1 (0.31)
Pigs				
	Liquid	<1–1386 (714)	<5–5832 (1427)	0.06–1.3 (0.65)
	Solid	134–780 (447)	206–1220 (678)	0.19–0.53 (0.34)
Sheep				
	Solid	13	104	ND
Hens				
Layers	Solid (deep litter)	17–486 (72)	237–789 (515)	0.09–2.04 (0.53)
Broilers	Solid	8.4–173 (72)	52–473 (355)	0.11–1.16 (0.27)

Means are given in parentheses.
(Based on Benke et al. (2008), Brown (2008), Menzi and Kessler (1998), Nicholson et al. (1999), Unwin (1998))

of different minerals (mainly N and P) vary with crop species, the parts of the plants incorporated, the growth stage at incorporation, soil fertility and physical conditions. Palm et al. (2001) compared the N contents of crop residues, leaf litter, stems, and roots for different crops and found median *N* values less than 2.0%, with few samples having larger concentrations. In contrast, the N concentration in fresh leaves can range from 1 to 5.5% (median value > 3.0%), with the largest values found in the *Fabaceae* and *Asteraceae* (median concentration of 3.5%). The P concentrations in the majority of crop residue, leaf litter, stems and roots are less than 0.15%, with few examples exceeding 0.2%. Fresh leaves may contain more than 0.15% P but maximum values do not exceed 0.50% (Palm et al., 2001).

The timing and rate of release of the nutrients applied in organic form are dependent on the nature of the materials themselves as well as the climate, soil and land management practices. There may also be a residual effect from previous applications of amendments, which have not been fully mineralized in preceding years (Schröder, 2005). This makes it difficult to be precise about the amount required and when it needs to be applied to make the availability synchronize with the needs of the plants. Consequently, there is a considerably greater risk that nutrients will be lost to the wider environment from organic amendments than from mineral fertilizers.

4. POTENTIAL CONTAMINANTS ASSOCIATED WITH ORGANIC AMENDMENTS

Organic amendments are natural sources of C for microbes, including those that can be pathogenic to humans. Proteinaceous compounds and nucleic acids are degraded to release inorganic nitrogen and sulfur compounds, while trace metals, such as Cu and Zn, can be released from the degradation of certain enzymes. Municipal wastes may contain metals, together with microbes, organic and inorganic chemicals coming from industrial sources and runoff from roofs, roads and parking lots. Amendments based on feces can also contain metals that are not adsorbed from food and water, antibiotics used to treat disease or as a prophylactic, together with sex and growth hormones.

4.1. Pathogens

The number of pathogens released into the environment is an important component of a quantitative risk assessment for the use of organic soil amendments. Animal feces are significant as a source of enteric and other organisms, such as bacteria, viruses, protozoa and helminthic worms, some of which can be parasitic or pathogenic in humans (Strauch, 1987). Enteric pathogens can infect both farm animals and humans through contaminated feed, water supplies and feces (Herriott et al., 1998; Shere et al., 1998). Animals can also become infected from contaminated bedding, which when handled can also be a source of diseases for humans (Hogan et al., 1989). Many of the organisms that are commonly found in animal manure are also found in sewage biosolids (Gerba and Smith, 2005). In addition to these organisms, there are concerns related to the manure from housed animals as well as sewage biosolids from water treatment plants receiving hospital wastes as these might be a source of prions (see Box 5.1).

Box 5.1 Prions

Prions are proteinaceous infective particles lacking nucleic acids that are abnormal (misfolded) forms of membrane proteins normally found in cell surface membranes (Prusiner, 1998). The effect of a prion is to cause other normal proteins to reconform into the abnormal form (Prusiner, 1998). A disease is recognized when the prion exerts its effect in the brain and causes differences in animal behavior. Scrapie in sheep and chronic wasting disease in deer and elk have long been recognized as prion-mediated diseases. Bovine spongiform encephalopathy (BSE) was identified in cattle in 1986. In humans, Creutzfeldt-Jakob Disease (CJD) is a fatal disease caused by a prion protein that attacks the brain, killing cells and creating gaps in the cortical tissue. Collectively, these diseases are known as Transmissible Spongiform Encephalopathies. In the 1996, a new TSE disease, variant Creutzfeldt-Jakob Disease (vCJD) was identified, linked to eating meat, particularly that containing nerve tissue, from cattle infected with BSE.

Prions are resistant to inactivation by UV and ionizing radiation as well as having considerable tolerance to heat (Prusiner, 1998). The tolerance is much less to dry heat than moist heat (Gale, 2007). In addition to tolerance to physical treatments, prions show considerable resistance to chemical treatments, including the use of enzymes that can denature protein (Gale, 2007; Prusiner, 1998). The use of formaldehyde before autoclaving can enhance prion infectivity (Gale, 2007).

Based on evidence from infection of sheep with scrapie, prion infection can take place vertically from mother to offspring as well as by postnatal lateral transfer between individuals (Dickinson et al., 1974; Ryder et al., 2004). Vertical transfer was originally believed to occur by prenatal infection as well by postnatal contact with voided placentas or fetal fluids. More recent evidence indicates the possibility of transfer in milk (Konold et al., 2008), especially if ewes have

Continued

Box 5.1 Prions—cont'd

lentivirus-induced mastitis (Ligios et al., 2011). The demonstration by Tamgüney et al. (2009) and Terry et al. (2011) of the presence of prions in feces of diseased sheep and asymptomatic deer, respectively, together with presence in saliva (Maddison et al., 2010b) and urine (Murayama et al., 2007) suggests that ingestion from contaminated pasture could result in both vertical and lateral transfers. Oral ingestion seems to be the most important route through which infection occurs (van Keulen et al., 2008).

There are possibilities for animal prions to enter manure storage from housed-infected animals, while human and animal prions could enter municipal sewage systems, and hence for both to be spread on agricultural land (Gale and Stanfield, 2001). Once in contact with soil, prions rapidly adsorb onto quartz sand and clay particles, particularly phyllosilicates (Ma et al., 2007; Maddison et al., 2010a; Walter et al., 2011). When sorbed onto clay, prions do not lose their infectivity and if ingested are as infective as unbound agents (Johnson et al., 2007). Prions adsorbed onto montmorillonite shortened the incubation period for disease in hamsters (Johnson et al., 2007). Once bound to phyllosilicates and quartz, there appears to be little desorption, so that infective agents remain close to the soil surface and infectivity can persist for at least 18 months (Jacobson et al., 2010). Risk of transport to water resources would, therefore, likely be associated with particle transport in surface runoff rather than in leaching to groundwater. However, investigation of transport to deeper soil layers and shallow groundwater through preferential flow paths is required. Nevertheless, Gale et al. (1998) concluded that there was negligible risk to humans from drinking water from a chalk aquifer over which abattoir wastewater containing BSE prions was applied in subsurface irrigation.

Some pathogens only infect one species of farm animal while others are found associated with all livestock but animals may not exhibit evidence of the presence of pathogenic strains in their system. Even if they can infect all major farm animals, the organisms present vary between cattle (*Bos primigenius taurus* Bojanus.), pig (*Sus scrofa* L.) and poultry manure (Bicudo and Goyal, 2003; see also Tables 5.6–5.10). Even so, as techniques for identifying virulent strains become ever more precise, evidence may accrue to indicate that subtle differences exist such that the dominant strain associated with one species of farm animal is not the one that actually causes disease in humans. For example, Olson et al. (2004) concluded that the strain of *Giardia duodenalis* that commonly causes diarrhea in cattle is different from the strain that is pathogenic in humans. The pathogens chosen for illustration in this paper are widespread, although the proportion of affected herds varies considerably. For example, *Yersinia enterocolitica*—a concern mainly associated with pig manure—is not found everywhere or in the majority of herds (Côté et al., 2006; Létourneau et al., 2010).

Inside the alimentary system of the host, pathogenic strains compete with long-term colonizers and more transitory strains present. Infected animals do not shed viable pathogenic organisms continuously in their feces. In consequence, the shedding of pathogens by a small number of infected animals can contaminate all the manure in storage (Strauch, 1988).

Some of the nonpathogenic microbial strains of *Enterococcus* and *Escherichia coli* present in the alimentary system are used as indicators of fecal contamination, and these are usually in large numbers in manure and sewage biosolids, certainly in much greater numbers than are pathogenic organisms. Indicator organisms are selected for their ease of culturing and enumeration but their persistence in the environment may not be the same as that of pathogens (Wilkes et al., 2009). Furthermore, pathogens vary widely in the critical number required to initiate illness in humans. Nevertheless, the presence of a small number of viable indicator organisms in sources of drinking water can indicate a health concern (Raina et al., 1999).

Assessment of pathogen loading to land therefore requires a measure of the prevalence of the organism among herds and flocks as well as that within a herd together with an estimate of the number of viable organisms being shed. The information available is at best incomplete for the major pathogens discussed here.

4.1.1. Bacteria

Bacteria are present in large numbers in manure (Table 5.5); for example, there may be 10^{10} mL^{-1} in liquid manure. Due to the greater mobility of bacteria in the liquid phase compared to the solid phase, liquid manure tends to be more uniformly contaminated than solid manure. Bacteria present in greatest numbers are fecal coliforms and *Enterococcus* (Table 5.5). Many bacteria of interest have the ability to adapt to living under anaerobic conditions. Values given in Table 5.5 provide an indication of the potential concentration in a sample of manure. However, the actual value will be influenced by the proportion of animals in a herd that are shedding pathogens with their feces (and urine in the case of viruses) (Tables 5.6–5.10).

Escherichia coli is a facultative anaerobic fecal coliform bacterium. It is usually present in animal feces and most strains are benign or beneficial. These strains are commonly used as indicators of fecal contamination in water resources. However, some strains of *E. coli* can cause disease.

Table 5.5 Bacterial numbers in some animal manure (CFU g^{-1} fresh manure)

Manure type	Fecal coliforms	*Enterococcus*	*Escherichia coli* O157:H7	*Salmonella*	*Campylobacter* spp.	Source
Liquid swine manure	2.4×10^3 to 5.9×10^6	5×10^4 to 7.2×10^4	1.3×10^3	0 to 1.5×10^3	6.1×10^2	Crane et al. (1983), Rüprich (1994), Unc and Goss (2006b)
Liquid cattle manure	2.4×10^3	4.5×10^2 to 9.5×10^5	1.5×10^2 to 1.3×10^7	2.5×10^3	5.3×10^2	Rüprich (1994), Stanley et al. (1998a)
Dairy slurry	6.3×10^4 to 1.0×10^7	4.5×10^2 to 2.7×10^7	2.6×10^2	2.5×10^3	5.3×10^2	Crane et al. (1983), Östling and Lindgren (1991), Stanley et al. (1998b)
Solid beef manure	2.4×10^5 to 6.8×10^6	1.5×10^7				Unc and Goss (2006b)
Solid dairy manure	2.0×10^5 to 1.0×10^7					Östling and Lindgren (1991), Stanley et al. (1998a)
Fresh cow manure	Up to 1.0×10^9		<3 to 2.4×10^3	<1 to 10^5	6.9×10^1 to 3.2×10^5	Clinton et al. (1979), Fegan et al. (2003), Mawdsley et al. (1995)
Sheep	6.0×10^6	6.6×10^5	2.5×10^2	5.8×10^3 to 2.0×10^4	10^1 to 10^5	Crane et al. (1983)
Horse	9.4×10^4	6.3×10^6				Crane et al. (1983)
Poultry	1.3×10^6 to 1.4×10^8	6.2×10^5 to 1.9×10^8	ND*	4×10^3	8.5×10^8 to 10^9	Crane et al. (1983)

*ND—not determined.

Table 5.6 Mean percentage of herds or flocks in a study area with disease carriers identified, infected animals in a study area, or animals within a herd carrying verotoxin-producing *E. coli* (VTEC)

Animal	Prevalence measure	Prevalence % VTEC	References
Dairy cattle	Herds	42 (17–64)	Aslam et al. (2010)
	Animals	5 (<1–16)	Berry et al. (2006)
	Animals in herds	14 (1–41)	Beutin et al. (1993)
Beef cattle	Herds	48 (2–72)	Bolton et al. (2012)
	Animals	13 (2–28)	Chapman et al. (1997)
	Animals in herds	24 (2–54)	Edrington et al. (2004)
Cattle (Unclassified)	Herds	22 (13–39)	Elder et al. (2000)
	Animals	12 (4–21)	Farzan et al. (2010)
	Animals in herds	10	Feder et al. (2003)
Sheep	Flocks	18 (2–40)	Franco et al. (2009)
	Animals	29 (2–67)	Franz et al. (2007)
	Animals in flocks	7	Gannon et al. (2002)
Pig	Herds	15 (0–32)	Hancock et al. (1994)
	Animals	3 (0–8)	Hancock et al. (1997a)
	Animals in herds	2	Hancock et al. (1997b)
Chicken	Flocks	–	Herriott et al. (1998)
	Animals	0	Heuvelink et al. (1999)
	Animals in flocks	–	Hutchison et al. (2002)
Turkey	Flocks	–	Hutchison et al. (2005a)
	Animals	1	Laegreid et al. (1999)
	Animals in Flocks	–	Lahti et al. (2003)
			Looper et al. (2009a,b)
			Mechie et al. (1997)
			Montenegro et al. (1990)
			Neilson et al. (2002)
			Ogden et al. (2004)
			Ogden et al. (2005)
			Omisakin et al. (2003)
			Oporto et al. (2008)
			Paiba et al. (2002)
			Paiba et al. (2003)
			Shere et al. (1998)
			Sidjabat-Tambunan and Bensink (1997)
			Stanford et al. (2005)
			Wells et al. (1991)
			Zhao et al. (1995)

Range of values in parentheses. Preference given to results describing detects in fecal samples.

Table 5.7 Mean percentage of herds or flocks in a study area with disease carriers identified, infected animals in a study area, or animals within a herd carrying thermophilic *Campylobacter* spp.

Animal	Prevalence measure	Prevalence % *Campylobacter* spp.	References
Dairy cattle	Herds	89 (67–100)	Akhtar et al. (1990)
	Animals	36 (19–47)	Alter et al. (2005)
	Animals in herds	54 (67–37)	Altrock et al. (2007)
Beef cattle	Herds	80 (59–100)	Anderson et al. (2012)
	Animals	57 (41–89)	Atabay and Corry (1997)
	Animals in herds	–	Atabay and Corry (1998)
Cattle (Unclas-sified)	Herds	30 (13–63)	Bae et al. (2005)
	Animals	14 (0–24)	Bardon et al. (2008)
	Animals in herds	60 (39–81)	Berndtson et al. (1996)
Sheep	Flocks	46 (21–88)	Berry et al. (20006)
	Animals	44 (22–92)	Bolton et al. (2012)
	Animals in flocks	9	Denis et al. (2011)
Pig*	Herds	64 (14–100)	El-Shibiny et al. (2005)
	Animals	63 (25–100)	Ellerbroek et al. (2010)
	Animals in herds	75 (58–100)	Ellis-Iversen et al. (2009)
Chicken	Flocks	44 (19–86)	Farzan et al. (2010)
	Animals	69 (39–100)	Gebreyes et al. (2005)
	Animals in flocks	–	Grinberg et al. (2005)
Turkey	Flocks	50	Grove-White et al. (2010)
	Animals	75 (66–83)	Guévremont et al. (2004)
	Animals in Flocks	7	Hakkinen and Hänninen (2009)
			Hartnach et al. (2009)
			Haruna et al. (2012)
			Hutchison et al. (2002)
			Hutchison et al. (2005a)
			Jensen et al. (2006a)
			Luangtongkum et al. (2006)
			Manser and Dalziel (1985)
			Meldrum et al. (2006)
			Munroe et al. (1983)
			Oporto et al. (2007)
			Perko-Mäkelä et al. (2011)
			Ridley et al. (2011)
			Sato et al. (2004)
			Schuppers et al. (2005)
			Scott et al. (2012)
			Stanley et al. (1998a)
			Stanley et al. (1998b)
			Varela et al. (2007)

Range of values shown in parentheses. Preference given to results describing detects in fecal samples.
*Mostly *C. coli*.

Table 5.8 Mean percentages of herds or flocks in a study area with disease carriers identified, infected animals in a study area, or animals within a herd carrying *Salmonella* spp.

Animal	Prevalence measure	Prevalence % *Salmonella* spp.	References
Dairy cattle	Herds	56 (11–100)	Baptista et al. (2010)
	Animals	27 (0–41)	Bolton et al. (2012)
	Animals in herds	33 (1–81)	Callaway et al. (2005)
Beef cattle	Herds	46 (10–82)	Clinton et al. (1979)
	Animals	2	Cummings et al. (2009)
	Animals in herds	33	Davies et al. (2003)
Cattle (Unclassified)	Herds	6 (5–8)	Edrington et al. (2004)
	Animals	18	Farzan et al. (2006)
	Animals in herds	–	Farzan et al. (2010)
Sheep	Flocks	9 (8–9)	Fosse et al. (2008)
	Animals	–	García-Feliz et al. (2007)
	Animals in flock	–	Grinberg et al. (2005)
Pig	Herds	38 (2–90)	Hölzel and Bauer (2008)
	Animals	21 (1–100)	Hurd et al. (2003)
	Animals in herds	10	Hutchison et al. (2002)
Chicken	Flocks	18 (13–24)	Hutchison et al. (2005a)
	Animals	48 (7–88)	Jensen et al. (2006b)
	Animals in flocks	–	Kishima et al. (2008)
Turkey	Flocks	–	Lo Fo Wong et al. (2003)
	Animals	–	Looper et al. (2009b)
	Animals in Flocks	–	Mafu et al. (1989)
			Mejía et al. (2006)
			Meldrum et al. (2006)
			Merialdi et al. (2008)
			Nielsen et al. (2011)
			Parveen et al. (2007)
			Rajic et al. (2005)
			Steinbach et al. (2002)
			Van Hoorebeke et al. (2009)
			Vanselow et al. (2007)
			Warnick et al. (2003)

Range of values shown in parentheses. Preference given to results describing detects in fecal samples.

One such is *E. coli* O157:H7 commonly associated with the so-called "Hamburger Disease" in which the genes for "Shiga toxin" or "verotoxin" are present. Prevalence of the organism is greatest in cattle and sheep but it has been detected infrequently in pig and poultry (Table 5.6), so there would appear to be less risk of human infection from these latter sources. There is evidence for seasonality in shedding of *E. coli* O157:H7, including

Table 5.9 Mean percentages of herds or flocks in a study area with disease carriers identified, infected animals in a study area, or animals within a herd carrying *Listeria* or *Yersinia*

		Prevalence %		**References**
Animal	**Prevalence measure**	***Listeria***	***Yersinia***	***Listeria***
Dairy cattle	Herds	46	–	Hutchison et al. (2002)
	Animals	–	–	Hutchison et al. (2005a)
	Animals in herds	–	–	Esteban et al. (2009)
Beef cattle	Herds	31	–	Farzan et al. (2010)
	Animals	3 (<1–5)	–	Mohammed et al. (2010)
	Animals in herds	–	–	Pradham et al. (2009)
Cattle (Unclassified)	Herds	29 (29–30)	–	Skovgaard and Nørrung (1989)
	Animals	–	18	
	Animals in herds	21	–	***Yersinia***
Sheep	Herds	25 (14–30)	–	Mafu et al. (1989)
	Animals	–	–	Farzan et al. (2010)
	Animals in herds	2	–	Fukushima et al. (1983)
Pig	Herds	18 (4–27)	48 (5–100)	Liang et al. (2012)
	Animals	3 (2–3)	12 (5–20)	Pilon et al. (2000)
	Animals in herds	0	40	Apslund et al. (1990)
Chicken	Flocks	19 (19–19)	–	Gürtler et al. (2005)
	Animals	–	–	
	Animals in flocks	–	–	
Turkey	Flocks	–	–	
	Animals	–	–	
	Animals in flocks	–	–	

Range of values shown in parentheses.

Table 5.10 Percentage of herds or flocks in a study area with disease carriers identified, infected animals in a study area, or animals within a herd carrying zoonotic protozoan pathogens

		Prevalence %		References	
Animal	**Prevalence measure**	***Cryptosporidium parvum***	***Giardia* spp.**	***Cryptosporidium parvum***	***Giardia***
Dairy Cattle	Herds	14	60	Castro-Hermida et al. (2002)	Coklin et al. (2007)
	Animals	6 (0–22)	27 (9–42)	Causapé et al. (2002)	Heutink et al. (2001)
	Animals in herds	35 (8–59)	7	Coklin et al. (2007)	Hutchison et al. (2002)
Beef Cattle	Herds	–	–	Duranti et al. (2009)	
	Animals	11 (5–17)	100	Enemark et al. (2002)	Hutchison et al. (2005a)
	Animals in herds	43 (21–64)	–	Grinberg et al. (2005)	
Cattle (Unclassified)	Herds	24 (5–63)	20 (4–53)	Huetink et al. (2001)	Maddox-Hyttel et al. (2006)
				Hutchison et al. (2002)	
	Animals	45 (20–70)	29	Hutchison et al. (2005a)	Olson et al. (1997)
	Animals in herds	–	–	Lorenzo Lorenzo et al. (1993)	Quílez et al. (1996a)
Sheep	Flocks	48 (29–84)	21 (21–22)	Maddox-Hyttel et al. (2006)	Ralston et al. (2003)
	Animals	34 (23–46)	38	McEvoy and Giddings (2009)	Wade et al. (2000)
	Animals in herds	–	–	Olson et al. (1997)	Winkworth et al. (2008)
Pig	Herds	52 (14–100)	30 (2–84)	Quílez et al. (1996a,b)	
	Animals	12 (11–13)	10 (9–11)	Ralston et al. (2003)	Xiao et al. (1994)
	Animals in herds	–	–	Scott et al. (1995)	
Chicken	Flocks	ND*	ND*	Wade et al. (2000)	
	Animals			Winkworth et al. (2008)	
	Animals in herds			Xiao et al. (1994)	
Turkey	Flocks	100	–		
	Animals	6	–		
	Animals in herds	8	–		

*Not determined.

those that carry genes for antimicrobial resistance, with more being released in feces in warmer seasons than in cooler times of the year (Aslam et al., 2010). Both the shedding of *E. coli* O157:H7 and the concentration in feces are reduced in animals carrying endemic bacteriophages infecting this bacterium (Niu et al., 2009).

Campylobacter species *Campylobacter jejuni* and *Campylobacter coli* are a very common cause of gastrointestinal illness in humans but may also result in urinary tract infections. *Campylobacter jejuni* is most frequently associated with consuming contaminated poultry meat or raw milk. *Campylobacter* spp., when present, have been found in greater numbers in manure from chicken (*Gallus gallus domesticus* L.) (8.5×10^8 to 10^9 CFU g^{-1}) than in cattle manure (6.9×10^1 to 3.2×10^5 CFU g^1) (Table 5.5). Prevalence of Campylobacter spp. within and between herds and flocks is also much greater than found for *E. coli* O157:H7 (Table 5.7). The proportion of animals carrying Campylobacter is also large, with little difference between species, including turkey (*Meleagris gallopavo* L.). There is evidence that flocks of chicken can be infected by *Campylobacter* strains originating in cattle, although at least some of those strains may have preference for cattle (Ridley et al., 2011).

Salmonella is a common foodborne bacterial pathogen that gives rise to gastric influenza-like symptoms in humans (Pell, 1997). Drinking unpasteurized milk and eating undercooked eggs have been factors in the transmission of this disease organism (Bicudo and Goyal, 2003). The proportion of infected animals appears to be greater in pig and chicken than in cattle or sheep (*Ovis aries* L.) (Table 5.8). The organisms tend to be more prevalent in manure over summer months (Farzan et al., 2010).

Listeria monocytogenes gives rise to gastrointestinal disease but can cause severe neurological disorders in humans (Pell, 1997). It is a disease organism associated with a high mortality rate for those infected (Esteban et al., 2009). Most farm animals and some humans can be symptomless carriers of the bacterium. There appears to be little seasonality in the shedding of the microbe by farm animals or its presence in animal produce (Esteban et al., 2009; Mohammed et al., 2010). The limited number of reports suggests that herd prevalence in cattle may be somewhat greater than in other species (Table 5.9). The proportion of infected animals appears to be relatively small (Table 5.9).

Clostridium is a genus of anaerobic spore-forming bacteria, whose spores are resistant to environmental stresses, including disinfecting agents. Two species, *Clostridium perfringens* and *Clostridium difficile* are excreted in the feces of many animals and are an important cause of foodborne illness.

They commonly cause diarrhea in neonatal pigs, calves and sheep. In one study of *Clostridium* across the Midwestern USA, all of the 11 integrated pig operations tested positive for both *C. perfringens* and *C. difficile*. All the 16 regional pig farms had positive tests for *C. perfringens* but only 10 showed positive for *C. difficile* (Baker et al., 2010).

Yersinia enterocolitica is an enteric bacterium, which in human subjects causes acute gastro-enteritis characterized by watery diarrhea and abdominal pain. It is a significant foodborne pathogen being able to grow at refrigeration temperature. It has commonly been associated with pig (Tauxe et al., 1987) but has also been isolated from both healthy and diseased cattle and diseased sheep (Brewer and Corbel, 1983) (Table 5.9).

4.1.2. Protozoan Parasites

In addition to the bacterial pathogens, two protozoan parasites may be found in manure, especially that coming from cattle. These are *Cryptosporidium* spp. and *Giardia* spp. (Table 5.10).

Cryptosporidium parvum requires the ingestion of between 1 and 100 oocysts (the infectious agent) to cause disease in humans. It is a threat to the human population through potable water supplies, as it does not seem to be controlled by chlorination at levels that are safe for use in domestic water supplies (Pell, 1997). However, the oocysts range in size from 4–6 μm in diameter so they can be removed by appropriate sand filtration. In the only reported national survey (Great Britain), the geometric average concentration of oocysts in infected cattle manure was 19 g^{-1} but the maximum value recorded in feces from an individual animal was 3500 g^{-1} (Hutchison et al., 2004a). The comparable values for pig manure were 58 g^{-1} and 3600 g^{-1}. However, larger concentrations have been found in more localized studies (Scott et al., 1994; Xiao et al., 1994).

Giardia lamblia only requires the ingestion of about 10 of the ovoid (8–14 × 5–10 μm) cysts to cause disease in humans. Hutchison et al. (2004a) reported a geometric average concentration of *Giardia* cysts in infected cattle manure of 10 g^{-1}, with a maximum value in feces from an individual animal of 5000 g^{1}. Values for calves generally exceed those for cows (Ralston et al., 2003). Values for pig manure were about 10 times greater than for cattle.

4.1.3. Helminthic Worms

Two helminths can be found in animal manure. *Ascaris suum* is associated with pig manure. *Tenia solium*, the human tapeworm, is very uncommon in North America and Western Europe, but pigs and sheep can be intermediate hosts.

4.1.4. Viruses

Other common contaminants of manure include viruses (Addis et al., 1999). Coronaviruses, which give rise to diarrhea in calves and pigs, and reoviruses, which are excreted by cattle, are found in manure (Strauch, 1987). Bovine rotaviruses may be isolated from cattle manure, but this is not thought to be common (Pell, 1997). Nevertheless, rotaviruses cause diarrhea in neonates of humans and a number of other animals (Estes and Cohen, 1989).

Importantly, many of the viruses of animals that are likely to be found in manure apparently do not give rise to diseases in humans (Pell, 1997). For example, bovine parvoviruses do not appear to be related to those that affect humans. Enteroviruses and adenoviruses in animals have not been considered to represent a significant threat to humans (Stelma and McCabe, 1992). However, swine hepatitis E is closely allied to the form of the virus found in humans, who can be infected through inadequately prepared liver (Yazaki et al., 2003). People can also contract hepatitis E through drinking contaminated water (Meng et al., 1997).

Influenza virus is very widespread, and poultry and pig can be considered as potential reservoirs of human strains. Viruses are able to survive outside the host for a prolonged period. For example, the infectious avian influenza virus can survive in water for 207 days at 17 °C (Brown and Alexander, 1998). However, information about the occurrence and longevity of viruses in animal manure is very limited.

4.1.5. Changes in Potential Pathogen Numbers during Manure Storage

With time in storage the total microbial loading in the manure acts on the organic matter present. Straw provides additional carbon sources that assist survival. Initially, the number of pathogens may increase as manure is put into storage, and then decline (e.g. Himathongkham et al., 1999). Alternatively, there may be an immediate reduction in numbers, which steadily continues with time (Hutchison et al., 2005b). The rate of decline of bacterial populations is much faster in solid manure stores than in liquid manure storage (Table 5.11). In contrast, *Enterovirus* appears to decline much more rapidly in liquid manure. Rotoviruses are stable in feces for as long as 7–9 months. Some bacteria present in manure can adversely affect the longevity of viruses. These bacteria have developed various strategies to inactivate viruses, including the formation of proteases (Pell, 1997).

Differences between winter and summer storage in the rate of loss of pathogens are relatively small and there appears to be no consistent

Table 5.11 Rate of decline of pathogens in stored manure

Livestock waste type	Time for one log_{10} reduction in CFU of bacteria, PFU of viruses and oocysts of *C. parvum* (days)					
	VTEC	*Salmonella*	*Campylobacter*	*Listeria*	*Cryptosporidium parvum*	*Enterovirus*
Liquid manure						
Summer						
Dairy	16.3	12.5	8.8	10.3	270	
Beef	31.6	18.0	16.2	12.3	202	9*
Pig	22.6	24.6	12.9	18.1	239	13
Winter						
Dairy	21.6	13.3	10.2	12.7	250	
Beef	26.4	17.0	14.2	12.2	158	
Pig	19.0	15.0	7.4	15.1	244	
Solid manure						
Dairy	1.5	1.8	2.3	2.9		
Beef	2.3	2.2	1.5	3.7		
Sheep	1.6	1.6	2.2	2.2		35*
Pig	1.4	1.8	1.9	3.2		300
Poultry	1.4	1.8	2.5	2.1		

*Human strain.
(Based on Bolton et al. (1999), Hutchison et al. (2005b,c), Lund and Nissen (1983), Nicholson et al. (2005))

pattern (Table 5.11). However, the slowest decline is associated with the infective agents of protozoa (Table 5.11). This means that these organisms persist longer in stored manure than do other pathogens. Temperature (Fig. 5.1), water content and pH of stored manure can all affect the rate of decline of pathogens and their persistence in storage (Arrus et al., 2006; Olson, 2000; Wang et al., 1996). Urea hydrolysis affects the level of carbonate ions and NH_3 in solution, which have antimicrobial properties, thereby influencing microbial survival (Park and Diez-Gonzalez, 2003; Wells and Varel, 2008).

In solid manure stores, pathogens close to the periphery of a pile may be subject to a cooler temperature regime than those near the center. Consequently, the latter may not survive, even if those at the periphery do and become a source for contamination, when spread on the land (Sutton, 1983). *Giardia* unlike many other pathogens is sensitive to freezing. Temperatures above 30 °C reduce survival times for most pathogens, with the possible exception of *Giardia* (Olson, 2000; Wang et al., 1996). None of the organisms appear to survive for long periods of time in dry manure (Olson, 2000).

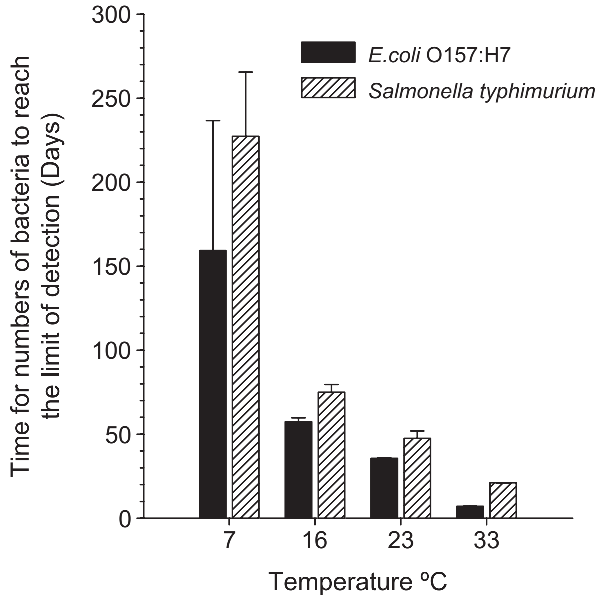

Figure 5.1 Effect of temperature on the survival of *E. coli* and *Samonella typhimurium* in beef cattle manure. Limit of detection was 1 $\log_{10}$ CFU g^{-1} dry weight. *(Data from Semenov et al. (2007)).*

Nazir et al. (2011) investigated the survival of a number of avian influenza viruses in duck (*Anas platyrhynchos*) feces at incubation temperatures of 10 and 30 °C. The time to 90% inactivation (T_{90}) ranged from 2 to 3 weeks at the cooler temperature but at 30 °C it was only 1–2 days. Two other viruses, Newcastle disease virus and enteric cytopathogenic bovine orphan virus, persisted longer, having T_{90}values at 10 °C of 9.3 and 23.7 weeks respectively (Nazir et al., 2011).

4.2. Macronutrients

Threats to human health from forms of N in soil amendments come from NH_3, NO_2^- and only indirectly from NO_3^-. Threats to human health from forms of N in soil amendments come from the products of the mineralization and nitrification of organic compounds, NH_4^+–N, NO_2^-–N and NO_3^-–N. Only the latter two pose a direct threat to human health when formed after the application of organic amendments to the land.

Once it enters a potable water source, NO_3^- has been a concern because once in the mouth it can be reduced to NO_2^- at the back of the tongue (Duncan et al., 1995). Following uptake into the bloodstream,

NO_2^- is responsible for nitrite-induced conversion of oxy-hemoglobin (Hb) to methemoglobin (MetHb) (Forman et al., 1985). In MetHb, the iron in the heme group is oxidized but as the reaction is not readily reversible in neonates, tissues become starved of oxygen resulting in a cyanosis. However, the evidence suggests the need for a combination of high levels of NO_3^- and bacteria in the stomach for MetHb to develop (Addiscott and Benjamin, 2004; Rudolph et al., 1992; WHO, 1984). However, the formation of MetHb can occur in the absence of high levels of NO_3^- in the diet as a response to gastroenteritis (Hegesh and Shiloah, 1982). Under the normally very acid (pH 1 to pH 3) conditions in the stomach (and at various points in the mouth), NO_2^- (pKa 3.4) can be converted to NO and NO_2 (Benjamin et al., 1994). However, in infants younger than 6 months and in a very few adults the levels of acidity are lower, possibly limiting the conversion. NO is a very effective antimicrobial agent, suggesting that its formation is consistent with a beneficial role for nitrate in our food and water (Addiscott and Benjamin, 2004). Nevertheless, it is thought that in young infants NO released in response to gastroenteritis may enter the bloodstream and convert Hb to MetHb and the development of cyanosis. There is now evidence that deoxy-hemoglobin can reduce NO_2^- to NO, consistent with a protective role for nitrite in the circulatory system (Cosby et al., 2003). The other concern is that in the stomach and intestines NO may form in response to chronic inflammation and following reaction with O_2 it can be converted to the nitrosating compounds N_2O_3 and N_2O_4. These oxides of N can react with secondary and tertiary amines and with amides to form N-nitroso compounds (NOC), believed to be responsible for cancers of the gastrointestinal tract. NOC have also been linked to other health issues, diabetes and adverse reproductive outcomes, such as premature births and congenital malformations (Ward et al., 2005). L'hirondel and L'hirondel (2002) have provided a critical account of the evidence linking NO_2^- and NO_3^- intake to human health. For a discussion of the significance of these ions in the maintenance of a healthy body, see Bryan and van Grinsven (2012).

When organic amendments are stored, there is a health risk from volatilization of NH_3 following the hydrolysis of urea (cattle, sheep, pigs), uric acid (poultry) or its derivative ureides (Hristov et al., 2011). The NH_3 can build up sufficiently to cause irritation in poorly ventilated barns and adversely affect the lungs of workers (Kirkhorn and Garry, 2000). Ammonia has a strong, sharp characteristic odor, which can be a cause of great discomfort, especially for people suffering from asthma or allergies

(Schiffman and Williams, 2005; Schiffman et al., 2000). It is the irritation threshold levels, 4–8 ppm (v/v), rather than the odor threshold of 0.8 ppm (v/v), which can be a cause of the related health symptoms from NH_3 (Schiffman and Williams, 2005). Human detection of NH_3 odor occurs at sufficiently small concentrations that harmful exposures are generally avoidable by moving away from the source. Residents downwind from a store of organic amendments releasing NH_3 can also experience health discomforts such as increased eye irritation, nausea, weakness and even psychological disorders (Schiffman et al., 1995; Thu et al., 1997). However, the severity of the impact is determined by factors such as wind speed, wind direction and distance from the NH_3 source. For example, McGinn et al. (2003) observed that NH_3 concentrations declined asymptotically with distance, such that for two locations 200 m apart, the concentration was 65–82% less at the downwind site. The concentrations were generally greater during the evenings when wind speeds were less than during the day and dispersion was more limited.

Elevated NH_3 emissions from organic amendments can also potentially negatively impact the environment, and therefore be a human and animal health hazard, by contributing to eutrophication of surface waters, nitrate contamination of ground waters, soil acidity and fine particulate matter (PM) formation (Cisneros et al., 2010; Spiehs et al., 2010). While NH_3 itself is not a greenhouse gas, it may also contribute to global warming through nitrous oxides formation (Arogo et al., 2006). Probably, the greatest risk to public health results from reactions between NH_3 and sulfuric, nitric or hydrochloric acids in the atmosphere to form aerosols of fine particulate material with an aerodynamic diameter ≤ 2.5 µm ($PM_{2.5}$). Sulfuric and nitric acids are products of the oxidation in the atmosphere of SO_2 and NO_2, respectively. These two gases are formed from the burning of fossil fuels: SO_2 being mostly associated with power generating plants and NO_2 with road transport vehicles. The chloride ion in hydrochloric acid originates from particles of sea salt from the oceans (Finlayson-Pitts et al., 1989). $PM_{2.5}$ is of concern to human health as it is associated with irritation of the bronchial tubes, coughing, difficulty in breathing, decreased lung function, aggravated asthma, development of chronic bronchitis, irregularity of the heartbeat, nonfatal heart attacks and premature death in people with heart or lung disease (Arogo et al., 2006; WHO, 2006). Recent evidence points to a role for fine particulates in ischemic strokes (Wellenius et al., 2012) and cognitive decline in older women (Weuve et al., 2012). Chronic exposure to fine particle aerosols may also be a factor in developing lung cancer.

The significance of the finer $PM_{2.5}$ particulates is that they can be inhaled and reach deep into the lungs, where at the peripheral regions of the bronchioles and alveoli they can interfere with gas exchange (WHO, 2011). However, the evidence strongly suggests that differences in the chemical composition, including acidity, of particles rather than their size is critical to the development of health effects (Donaldson and Seaton, 2012; Seaton et al., 1995; Steenhof et al., 2011), with a decrease in metabolic activity in affected tissue being directly related to particulate oxidative potential (Steenhof et al., 2011).

Under anaerobic conditions, the breakdown of carbon compounds in organic amendments can release CH_4. This odorless gas can result in asphyxiation of anyone entering the close confines of tanks used for spreading liquid amendments. The breakdown of sulfur-containing proteins can release H_2S gas, which is poisonous. Proper attention to health and safety guidelines can avert these risks.

The P from organic amendments does not directly threaten human health, but it is the loss to surface water that can be problematic. When P enters freshwater bodies, it makes them more able to support the growth of aquatic plants, particularly algae—the process of eutrophication. The death of a bloom of these algae and its subsequent breakdown by microorganisms depletes the oxygen concentration in the water, limiting its availability to fish and other creatures, and in extreme conditions fish can be asphyxiated (Chambers et al., 2001). One group of algae that respond to the phosphorus by growing rapidly is the *Cyanobacteria*. Some of these organisms produce toxins that can cause illness in humans. Some toxins appear to attack the liver, others the nervous system, while others are more general in their impact (Falconer, 2001). It has been established that the exotoxins produced by cyanobacteria show specificity in the target organ, although cylindrospermopsin, a hepatotoxin, can also damage kidneys (Falconer et al., 1999) and cause gastroenteritis (Humpage et al., 2005). Exposure to cyanobacterial endotoxins, specifically lipopolysaccharides from the cell wall, has often been associated with the more general disease symptoms, but Stewart et al. (2006) have questioned that linkage. Although it is rare that drinking water containing the toxins results in acute or severe illness (Falconer, 2001), it has been claimed that long-term exposure may be associated with the development of cancer (Carmichael et al., 1985; Falconer, 1991). The evidence for specific toxins promoting tumor growth or being carcinogenic has come mainly from in vitro studies and with animal models (Falconer and Humpage, 2001).

4.3. Trace Elements

Common trace-element contaminants found in organic amendments are Cu, Zn and Cd but depending on the source, especially of sewage biosolids, there can be many other trace elements present. Trace elements Cu and Zn are frequently included in the diet of grower and finisher pigs as a simple means of reducing disease in animals confined to the barn, as well as improving weight gain. Lameness in dairy cattle may also be treated with Cu (Bolan et al., 2003). The provision of other metals can be required: for example, in North America animal diets may require the addition of selenium to avoid deficiency problems in the Great Lakes region, the pacific northwest of the U.S., and along much of the Atlantic coastal area (Ullrey, 1992). Much of a trace element supplied in the feed ends up in the manure from the animals. Increased use of recycled paper in pulp mills has resulted in elevated levels of some trace elements, such as Cu and Zn, in paper mill biosolids.

In addition to Cu and Zn, there can also be significant amounts of Cd (Table 5.4), Pb and Cr in manure (Menzi and Kessler, 1998; Nicholson et al., 1999; Unwin, 1998). However, Taiganides (1987) suggested that in the USA, cattle excrete only trace amounts of these metals. There is considerable variation in metal content between sources and types of manure on a dry matter basis. For example, the Cu content of pig manure is almost 10 times than that in poultry manure (Table 5.4). With the breakdown of organic compounds and associated loss of mass, the tendency is for the concentration of metals within a source of manure to increase with time. For Cd, Pb and Se, the median annual input to fields in sewage sludge is at least an order of magnitude greater than that applied in animal manure. In contrast, the median value for the input of Cu with sludge is about 7 times greater than that with manure. The input of Zn is similar from both sources (Sheppard et al., 2009).

Once organic amendments have been incorporated into the soil, metals tend to accumulate, as total removal is small. The bioavailability of metals to crops in freshly applied amendments, particularly manure, does not necessarily remain the same over time but tends to diminish (Bolan et al., 2004; Brandt et al., 2008; Stomberg et al., 1984), by a factor of a little less than 2 in the case of Cu (Smolders et al., 2012), possibly as result of an increase in the size of organic ligands to which they are attached (Del Castilho et al., 1993a), or through forming bonds with other metal oxides (Kukier et al., 2010). For example, when Cu is added to soil in organic amendments, it can be almost 6 times less available to plants than if the same amount is added as a Cu^{2+} salt (Smolders et al., 2012). The reduced bioavailability of Cd is not directly related to its exchangeability, which appears to be >60% in a number of soils receiving organic amendments (Kukier et al., 2010). Interactions with Fe may contribute to its reduced bioavailability (Kukier et al., 2010).

Recently, Chaney (2012) reviewed the safety of food associated with the bioavailability of the trace elements and heavy metals applied to soils in fertilizers or organic amendments. When crops absorb trace metals, these elements enter the feed and food cycle of animals and humans (De Vries et al., 2007). If they enter water resources, they can be taken into potable water supplies. Threats to human health come from concentrations of trace metals in food and water that either are inadequate, leading to deficiency diseases, or toxic (Benke et al., 2008; Gupta and Gupta, 1998; Oliver, 1997). In addition, there can be interactions between the availability of some metals in the human diet and the absorption of others. For example, absorption and retention of Cd can be enhanced in individuals whose diets contain insufficient amounts of Zn (Reeves and Chaney, 2008). Absorption of some categories of antibiotics, such as tetracyclines (Penttilä et al., 1975) and fluoroquinolones (Lomaestro and Bailie, 1995), can also be impaired by the simultaneous ingestion of Zn. There is some evidence that typical levels of Cd in the environment can reduce the likelihood of women to become pregnant (Buck Louis et al., 2012).

4.4. Antibiotics and Other Pharmaceuticals

Antibiotics are fed to or injected into animals to deal with specific disease, as a prophylactic or to induce changes in the microbial population in the rumen of cattle. The compounds and classes commonly administered to farm animals are similar or the same as those used in the health services of the human population (Gorbach, 2001, Table 5.12). A wide range of personal care products include compounds, such as triclosan and triclocarban, that have bactericidal properties and pass into sewage systems. Whether the antibiotics are administered by subcutaneous injection or in feed, only a part of that administered is metabolized in farm animals or in humans. Between 40 and 90% of an administered pharmaceutical dose is excreted soon after the treatment (Aust et al., 2008; Hansen et al., 2009; Winckler and Grafe, 2001), mainly in feces but some is also found in urine. For example, Lamshöft et al. (2007) administered sulfadiazine to pigs. After 10 days, only 4% of the administered sulphonamide drug remained in the animals, while the drug present in the manure accounted for 44% of the dose administered, with two main metabolic derivatives accounting for another 46%. It was

Table 5.12 Classification of some antibiotic drugs and their use in human and farm veterinary medicine

Antibiotic class	Commonly used representatives	Usage
Aminoglycosides	Neomycin	Used to prevent wound infections and to reduce cholesterol in humans. Feed additive (chicken)
	Gentamycin	Broad spectrum bacteriocide in farm animals
β-Lactams	Ampicillin	Broad spectrum bacteriocide in humans and occasionally for pig
	Penicillin	Widespread use against Gram-positive bacteria in human and veterinary medicine
Fluoroquinolones	Enrofloxacin	Broad spectrum bacteriocide used in veterinary treatment
	Difloxacin	Treatment of lower respiratory tract bacterial infections in farm animals
Glycopeptides	Vancomycin	"Drug of last resort" against Gram-positive bacteria in humans
Imidazoles	Metronidazole	Used as a treatment against anaerobic bacteria (e.g. *Clostridium difficile*) and protozoa in humans. Used for horse but not food animals.
Macrolides	Erythromycin	Similar usage to penicillin, but used against pneumonia
	Tylosin	Bacteriocide for beef cattle, pig and poultry. Feed additive (pig)
Polyethers	Monensin	Antimalarial properties. Bacteriocide. Feed additive (poultry, beef and dairy cattle)
	Salinomycin	Bacteriocide. Used in breast cancer therapy in humans and as a feed additive in chickens
Polypeptides	Ivermectin	General use against parasites (nematode, helminth) in both human and veterinary medicine
	Virginiamycin	Feed additive (pig and poultry)
Quinoxaline derivatives	Olaquindox	Feed additive (pig)
Sulphonamides	Sulfadiazine	Used in treatment of urinary infections in humans and as an antihelmithic in animals
	Sulfadimidine	Used in treatment of urinary infections in humans. Widespread use as an antibiotic in farm animals and as a feed additive (pig)
Tetracyclines	Tetracycline	Broad spectrum bacteriocide used in human and veterinary treatment
	Oxytetracycline	Broad spectrum bacteriocide used in human and veterinary treatment, including pneumonia

considered that these metabolites could be easily transformed back to the original antibiotic. As a result, large quantities of antibiotics can enter the environment each year via excretion (Christian et al., 2003) and are applied to soils in manure and in sewage biosolids for crop production. Information on the concentrations of different antibiotics in manure is scarce, but values vary greatly between source species and the type of operation (Table 5.13).

Degradation of antibiotics during storage of amendments is important in terms of total mass and the concentration at which they are applied to soil. Some antibiotics degrade rapidly in storage. For example, monensin, one of the most commonly used feed additives was degraded by about 85% within 11 weeks (Donoho, 1984). However, Dolliver et al. (2008) found that monensin and tylosin had undergone a reduction of between 54 and 76% after composting was complete, consistent with a half-life in the process of 17 and 19 days respectively. The concentration of tylosin in beef calf manure declined below the limit of quantification within 30 days of storage (De Liguoro et al., 2003). In contrast, concentrations of sulfdiazine and difloxacin in pig manure showed no decline after 150 days storage (Lamshöft et al., 2007). The breakdown of tylosin appears to be enhanced with increased dry matter content of the manure (Loke et al., 2000). Composting at a temperature of 25 °C reduced chlortetracycline and its isomer, iso-chlortetracycline by about 45% but between 36 and 45 °C more than 90% of oxytetracycline and chlortetracycline were eradicated. The process was even more effective if the temperature could reach 70 °C (Arikan et al., 2009a,b). Cessna et al. (2011) found similar results for the breakdown of chlortetracycline and iso-chlortetracycline (that had been formed during the formation of windrows for composting) and for tylosin, but results for sulfamethazine were more varied (93% removal in year 1 of an experiment but only a 59% reduction in the repeat year). Dolliver et al. (2008) obtained no decrease in the content of sulfamethazine by composting spiked turkey litter, although chlortetracycline was almost completely degraded over the same time period. King et al. (1983) showed that monensin reduced the organic N content of manure, indicating that antibiotics can affect microbial processes in manure (Haller et al., 2002; Schlüsener et al., 2003).

Some antibiotics can be persistent in soils (Christian et al., 2003; Hamscher et al., 2002; Samuelsen, 1989). Consequently, the concern is that zoonotic disease organisms (microorganisms that can cause disease in both humans and animals) will develop resistance to the families of antibiotic compounds used. This can then increase the risk of pathogens in the human environment developing greater resistance to treatment. Similarly,

Table 5.13 Concentration of some antibiotics in animal faeces or manure

Antibiotic class	Antibiotic	Source	Concentration mg kg^{-1} dry wt	References
Fluoroquinolones	Difloxacin	Pig	34–460	Karci and Balcioğlu (2009)
	Enrofloxacin	Chicken	0.05	Lamshöft et al. (2010)
				Sukul et al. (2009)
Macrolides	Tylosin	Beef calves	<0.1–115.5*	De Liguoro et al. (2003)
Polyethers	Monensin	Chicken	4000	Hansen et al. (2009)
Polypeptides	Ivermectin	Beef cattle	3.9–9.0	Sommer and Steffansen (1993)
Sulphonamides	Sulfachloropyridazine	Chicken	10–35	Aust et al. (2008)
	Sulfadiazine	Chicken	1–2	Burkhardt et al. (2005)
		Pig	2517.9	Christian et al. (2003)
	Sulfadimidine	Beef cattle	<0.1–10	Haller et al. (2002)
		Beef calves	1.0–6.2	Karci and Balcioğlu (2009)
		Pig (grower)	1–440	Lamshöft et al. (2010)
	Sulfamethoxazole	Chicken	<5	
	Sulfathiazole	Chicken	<6	
		Pig (grower)	0.1–5.3	
		Pig (farrowing)	12.4–375.8	
Tetracyclines	Chloretracycline	Beef cattle	0.1–401	Arikan et al. (2009a,b)
		Beef calves	113	Aust et al. (2008)
		Pig	880	Bao et al. (2009)
		Chicken	0.25–95	Karci and Balcioğlu (2009)
	Oxytetracycline	Beef cattle	0.05–18	Arikan et al. (2009a)
		Beef calves	0.8–871.7*	De Liguoro et al. (2003)
		Chicken	0.06–0.45	Karci and Balcioğlu (2009)

*Fresh feces.

if the same or similar products are applied to agricultural soil in sewage biosolids, this can increase the locations with microbes that are resistant to multiple drugs (Hölzel et al., 2010). Ramchandani et al. (2005) showed that some antimicrobial-resistant pathogenic strains of *E. coli* causing urinary tract infections in women were closely associated with isolates having the same molecular resistance markers obtained from cattle. Significantly, Chapin et al. (2005) reported the presence of airborne multidrug-resistant *Enterococcus* and *Staphylococcus* at a swine finisher barn in the USA, such that inhalation could provide a direct pathway of exposure to people entering such a facility.

4.5. Hormonally Active Compounds

The final group of contaminants found in organic amendments, which is used to supply plant nutrients, is composed of those that have hormone-like activities. A number of synthetic compounds, such as alkylphenols and degradation products of alkylphenolethoxylates (Brix et al., 2010), appear to mimic or interfere with the action of naturally occurring hormones and have been variously referred to as environmental estrogens or xenoestrogens. They are also known as endocrine-disrupting substances (Servos et al., 2001). In addition, compounds have been manufactured to simulate the action of natural sex hormones, which function to stimulate the growth of sexual structures and the development of secondary sexual characteristics. Both naturally occurring hormones and manufactured compounds can interfere with the action of the endocrine control mechanisms of other species. Long-term exposure can result in impaired growth, development and reproduction in fish, wildlife, and possibly even humans. Animal manure contains significant amounts of hormones (Kinney et al., 2008), which are produced mainly during reproductive phases. Estrogens are present in the urine or feces of pregnant and lactating females and testosterones in those from males. Similarly, testosterone and estrogens together with other endocrine disrupting substances can be present in sewage sludge (Andersen et al., 2003; McClellan and Halden, 2010; Muller et al., 2010; Stumpe and Marschner, 2007). Liu et al. (2011) described 26 compounds with hormonal activity that are widely used in oral contraceptives and are likely to be present in sewage biosolids. However, our knowledge of their fate in the environment is limited.

4.5.1. Natural Hormones and Endocrine Disrupting Substances

During reproduction, the manure of farm animals contains significant amounts of natural estrogens. An assessment of the relative estrogenic potency suggests that estradiol ranked first among a list of naturally occurring estrogens. The soybean constituent, genistein, was given a potency of 1, but the potency of estradiol was 10,000 to 20,000 times greater (Ivie et al., 1986).

Estrogens are excreted in both urine and feces. Estradiol exists in the form of two steroisomers: 17α- and 17β-estradiols. Cattle excrete both 17α-estradiol and 17β-estradiol, together with estrone and their conjugated forms linked to sulfate and glucuronide. In contrast, pig and poultry excrete mainly 17β-estradiol together with estrone, estriol and their conjugated forms (Hanselman et al., 2003). Ivie et al. (1986) reported that most of the 17β-estradiol injected into steer calves was metabolized before being excreted in feces (57% of initial material) or urine (42% of initial material). However, the metabolites also had estrogenic activity. 17β-estradiol is excreted by mature laying hens (MacRae et al., 1959), with the mass excreted (Mathur and Common, 1969) and the concentration in manure being greater than in that from nonlaying birds (Table 5.14). More estrone was excreted by laying hens than from nonlaying birds and peaked at, or around, the day that the first egg was laid (Mathur et al., 1966). Only small concentrations of estrogens have been detected in manure from beef cattle (Table 5.14).

Some compounds in the diet can contribute to the total loading of endocrine disruptive compounds in manure. For example, pigs and dairy cattle excrete equol, a metabolite of the phytoestrogens daidzein and formononetin, when fed with soybean products. Although equol was found in relatively large concentrations in the manure, its relative potency is considered to be much less than that of 17β-estradiol (Burnison et al., 2000).

After reviewing the available evidence, Lange et al. (2002) concluded that degradation of natural hormones was very slow during the storage of both liquid and solid manures. Most of the compounds excreted would therefore be applied to the land unless removed by a separate treatment process.

4.6. Compounds in Waste from Processing Agricultural Products

Some plant species, such as olive, cassava, oil palm, produce phenols and polyphenols in their tissues. Such compounds can be released from the tissue during the processing of plant material. Some of these phenols have biological effects, including antibiosis (González et al., 1990), antimicrobial (Obied et al., 2007) and phytotoxic properties (Hanifi and El Hadrami, 2008). However, these phenols do not usually persist in soil long enough to enter the food chain and represent any direct threat to human health.

Table 5.14 Concentrations of some natural hormones in animal manure

Animal category	Compound	Concentration (μg kg^{-1})	ng L^{-1}	Reference
Chickens				
Layer	Total estrogen	810–1600		Calvert et al. (1978)
Layer	Estradiol-17α			Shore et al. (1988)
Layer	Estradiol-17β	533		
Broiler	Total estrogen	330		Calvert et al. (1978)
Broiler	Estradiol-17α		133–380	Hutchins et al. (2007)
Broiler	Estradiol-17β	130	32–66	Nichols et al. (1997)
Broiler	Estrone		1500–2800	
Broiler	Estriol		187–468	
Pigs				
	Estradiol-17α	520	680–3000	Furuichi et al. (2006)
	Estradiol-17β*	1400	1500–4000	Hutchins et al. (2007)
	Estrone	4752	5400–10,100	Raman et al. (2004)
	Estriol		3000–6300	Burnison et al. (2000)
	Equol	40,000†		
Cattle				
Dairy	Estradiol-17α	210–1400	700–3750	Gadd et al. (2010)
				Hutchins et al. (2007)
	Estradiol-17β	100–150	1500	Raman et al. (2004)
	Estrone	210–540	4500	Zheng et al. (2008)
	Equol	3000*		Burnison et al. (2000)
Beef	Estradiol-17α		5.5	Hutchins et al. (2007)
	Estradiol-17β		<20	
	Estrone*		20	
Horse†				
	Estradiol-17β	40,000		Busheé et al. (1998)

*Soluble fraction only, approximate dry-matter content used to obtain concentration.
†Bedding present.

5. AGRICULTURAL BENEFITS FROM ORGANIC AMENDMENTS

The application of organic amendments to agricultural land provides nutrients, both macro- and micronutrients, to crops as well as contributing to the improvement in soil physical and chemical conditions for plant production. When mineral fertilizers are scarce, the use of organic amendments may be the only means of providing nutrients in sufficient quantity to achieve more than subsistence production.

5.1. Impact on Soil Properties and Conditions

An important reason for applying organic amendments is to offset the natural decline in organic matter and enhance the physical and chemical properties of arable soils. Surface application of organic amendments can improve the C and N content in the top 5 cm of the soil but may have no significant effect below this (Peacock et al., 2001). The magnitude of the effect depends on the dry matter content of the amendment. Consequently, effects can be small with liquid manure, such as that from pig (Angers et al., 2010). The increase in C is associated with greater cation exchange capacity (Bernal et al., 1992; Bulluck et al., 2002) and better resilience of soil aggregates to raindrop impact (Bresson et al., 2001; Singh and Agrawal, 2008), which in turn can improve infiltration and decrease runoff. In addition, adding organic matter to mineral soil increases water-holding capacity and porosity, and decreases bulk density (Wesseling et al., 2009). Permeability of soil can also be enhanced because of greater macroporosity at depth where organic amendments have been applied (Munyankusi et al., 1994). Johnston et al. (2009) reviewed the dynamics of soil organic matter and its effects on soil properties based on an analysis of the long-term experiments at Rothamsted Experimental Station (U.K.). The organic C content of a soil was a function of the input of organic material from crops or other sources, its rate of decomposition, the rate at which existing soil organic matter mineralized, soil texture, and climate. Under a given cropping system, the soil organic matter content tended toward an equilibrium level, which was greater for a clay soil than a sandy soil. Crop yields have tended to reflect the organic matter content of the soil. In part, this was the result of a greater annual release of N together with improved availability of P, when the amount of soil organic matter was enhanced. However, the loss of organic carbon was also greater under these conditions (Johnston et al., 2009).

Lack of organic C can reduce the surface charges in soil that are required for cation exchange and retaining basic elements, particularly Ca and Mg. This can result in increased soil acidity. Simply adding organic material is not entirely sufficient to counteract all the adverse changes in soil chemical and physical properties, since the nature and form of organic material is important in the formation of soil aggregates (Musgrave and Nichols, 1943). By providing resources to enhance the activity of soil microbes, the functions of the soil can rebuild with time. In a long-term experiment in northern Italy, a yearly application of 40 Mg ha^{-1} of farmyard manure from cattle to a clay soil for 44 years increased the concentration of humic carbon to ~26 g C kg^{-1} from the value of ~18 g C kg^{-1} measured in treatments that received no fertilizer or only mineral fertilizer (Simonetti et al., 2011). This application increased the amount of well-humified material in soil aggregate fractions, indicative of enhanced breakdown of readily catabolized organic matter.

The application of PMB, either untreated or composted, increased the mean weight diameter of water-stable aggregates by up to 71%, with the greatest change being in the numbers <2 mm (Bipfubusa et al., 2008). Bresson et al. (2001) reported that aggregate resilience to raindrop impact improved after the incorporation of composted sewage biosolids. In consequence, there was a longer time interval between the start of precipitation and the generation of runoff as the surface slumped. Less sediment was carried in runoff from the treated soil (Bresson et al., 2001).

The effect of organic material application on soil bulk density is variable. Khaleel et al. (1981) used results from 12 published studies and estimated the percentage decrease in soil bulk density that would result from the application of organic materials. Greatest benefits were reported for coarse-textured soils, and of the seven amendments considered the most effective was poultry manure, although in one experiment the application of 450 t ha^{-1} sewage biosolids in each of two years reduced bulk density by almost 30%. Bulluck et al. (2002) reported a significant reduction in soil bulk density associated with the application of a variety of organic materials. However, the impact on water-holding capacity is not always as obvious because changes in water content at both the upper and lower limits can result in no net change in the water held between the two (Khaleel et al., 1981). Nevertheless, Bernal et al. (1992) reported a 12% increase in water-holding capacity but it was confined to a soil with clay content close to 20%. The application of PMB and composted PMB to loamy sand enhanced plant available water (Foley and Cooperband, 2002; Newman et al., 2005), with increases being between 5 and 45% in the plant available water. Differences in permeability, as measured by hydraulic conductivity at saturation, can also be difficult to establish, although some large increases have been recorded (Khaleel et al., 1981). One reason for the lack of clarity might be the size of samples used. For example, Munyankusi et al. (1994) found that in soil with a history of mineral fertilizer, the greatest macroporosity was found in the top 5 cm of soil but somewhat deeper in soil that had regularly received liquid manure from dairy cattle. The continuity of macropores also extended to greater depths in the manured plot, so that preferential flow was somewhat greater (Munyankusi et al., 1994).

In India, Joshi et al. (1994) found that incorporating green manure into the soil increased soil water storage, which was still evident after a subsequent wheat (*Triticum aestivum* L.) crop. Settling index (cm cm^{-1}), an estimate of structural instability, was least in Sesbania-treated plots (0.26), a little larger (0.29) where weeds were incorporated, 0.30 in Leucaena-treated plots and greatest (0.34) in plots treated with NPK-fertilizer only. Soil dispersion increased from 6.0 to 10.0 g 100 g^{-1} through the same sequence of treatments. Saturated hydraulic conductivity in the Sesbania-treated plots was greater than in plots receiving only NPK-fertilizer. Sesbania was superior to other green manures for improving soil physical properties after its incorporation (Joshi et al., 1994; Sultani et al., 2007).

Changes in basic chemical properties of soil can result from applying organic amendments. Improved CEC can result from the application of organic soil amendments (Bernal et al., 1992; Bulluck et al., 2002). The effect was only observed on a soil with less than 10% clay, a soil with nearly 20% clay showed no significant difference in organic C content or CEC. In the soil with less clay, the CEC was directly related to the organic C content (Bernal et al., 1992). Soil pH showed a small but significant rise after application, followed by a decline at the end of two years. In the longer term, continued application of manure from cattle or pig resulted in a decrease in soil pH, with bigger changes resulting from the application of pig manure (Bernal et al., 1992; Chang et al., 1991).

A significant enhancement of the soil microbial biomass has frequently been reported with organic amendments (e.g. Narula et al., 2002; Peacock et al., 2001; Ros et al., 2006). Not only can there be an increase in the total population but the community structure can change with a significant shift from Gram-positive organisms in soil under crops given mineral fertilizer to one with many Gram-negative bacteria and more protozoa (Peacock et al., 2001). Satchell and Martin (1984) during an investigation of alkaline phosphatase activity in earthworm casts noted that the application of cattle manure greatly increased the background level of the enzyme complex. Similarly, green manuring induced a rapid increase in microbial activity after incorporation due to supply of readily degradable carbohydrates (Breland, 1995). Seedbed preparation and root penetration were also much easier for subsequent crops.

The structural improvements brought about by organic amendments help in reclaiming degraded soils. The addition of poultry manure (10 Mg ha^{-1})

to a heavily degraded tropical sandy soil significantly decreased soil bulk density, increased soil organic matter content, total porosity, water infiltration and saturated hydraulic conductivity (Obi and Ebo, 1995), therefore allowing better management and retention of mineral fertilizer. Similarly, application of crushed cotton gin compost and poultry manure increased plant cover in the treated plots to 88 and 79%, respectively, compared with 5% for the control treatment (Tejada et al., 2006). The same treatments enhanced soil microbial biomass and soil enzyme activities. Municipal solid waste considerably improved structural stability, infiltration rate and water-holding capacity, allowing a better reclamation of a salinized soil as a result of increased susceptibility to leaching (Lax et al., 1994).

Changes that take place in microbial communities due to organic amendments can have other beneficial effects. Organic amendments have been found to suppress several soilborne pathogens and soil animals. The list of organic materials tested for suppression of diseases is long and includes animal manure, green manure, crop residues, composts, agricultural processing waste water, crab shell (chitin), paper-mill biosolids and municipal wastes. Garbeva et al. (2004) identified a number of bacteria (*Pseudomonas*, *Burkholderia*, *Bacillus*, *Serratia*, Actinomycetes) and fungi (*Trichoderma*, *Penicillium*, *Gliocladium*, *Sporidesmium*, non-pathogenic *Fusarium* spp.) that exhibit antagonistic effects on soilborne plant pathogens. Suppressive mechanisms include competition, amensalism, microbial antagonism, parasitism, and systemic induced resistance. Bonanomi et al. (2010) concluded that organic amendments can suppress disease in plants by decreasing pathogen numbers, reducing pathogen saprophytic capability through N starvation (associated with high C:N ratio amendments), accumulation of ammonia or nitrous acids (in acidic soils), alteration of soil pH, increasing antagonistic and antibiotic effect on pathogens, and mycoparasitism. Green manures are among the most effective of amendments in suppressing soil pathogens and animal pests, especially brassica crops that contain glucosinolate, a toxic secondary metabolite. Lewis and Papavizas (1971) demonstrated that products from the decomposition of cabbage (*Brassica oleracea* var. *capitata*) suppressed pea root rot and adversely affected morphology, development of oospores, and mycelial growth of *Aphanomyces euteiches*. The sulfur-containing volatiles reported to be present in cabbage, or obtained from its decomposition, include mercaptans, sulfides of various types, and isothiocyanates.

Isothiocyanates include methyl isothiocyanate, which is also a breakdown product of the pesticide metham sodium (Mojtahedi et al., 1991) and are effective biocides (Johnson, 1985; Lear, 1956). For example, Hu et al. (2011)

showed that brassicaceous oilseed meal was effective in inhibiting the germination of sclerotia and the subsequent growth of *Phymatotrichopsis omnivora*, the fungus responsible for cotton root rot. Similarly, soil application of rapeseed and crambe (*Crambe abyssinica* Hochst.) meals reduced the number of galls on tomato roots growing in a soil infested with eggs of the root-knot nematode (*Meloidogyne arenaria*) (Walker, 1996). Both meals, however, were phytotoxic to tomato and reduced the height of *Celosia argentea* L., *Dianthus chinensis* L., and rose periwinkle (*Catharanthus roseus* (L.) G. Don) plants, and resulted in poorer germination of periwinkle seed. When the planting of tomato was delayed for 3 weeks after addition of 1% meals, the phytotoxicity and root injury were diminished, but not alleviated. The ED_{50} for rapeseed meal against peanut root-knot nematode was determined as 0.64% v/v when added to a piedmont soil and planting occurred immediately but increased to 0.86% if planting was delayed by just 1 week.

Larkin and Griffin (2007) used experiments in the greenhouse and the field to evaluate the effectiveness of canola (*Brassica napus* L.), rapeseed (*Brassica napus* L.), radish (*Raphanus sativus* L.), turnip (*Brassica rapa* var. *rapa* L.), yellow mustard (*Sinapis alba* L.) and Indian mustard (*Brassica juncea* (L.) Czern.), for control of various soilborne potato pathogens and diseases. In vitro assays showed that volatiles released from chopped leaf material of *Brassica* crops and barley (*Hordeum vulgare* L.) inhibited growth of a variety of soilborne pathogens of potato (*Solanum tuberosum* L.), including *Rhizoctonia solani*, *Phytophthora erythroseptica*, *Pythium ultimum*, *Sclerotinia sclerotiorum*, and *Fusarium sambucinam*, with Indian mustard resulting in nearly complete inhibition (80–100%). All *Brassica*s and barley reduced inoculum levels of *R. solani* (20–56% reduction) in greenhouse tests, and radish, rapeseed and Indian mustard reduced subsequent potato seedling disease by 40–83%. In an on-farm field experiment at a site with a substantial powdery scab problem (caused by *Spongospora subterranea* f. sp. subterranea), Indian mustard, rapeseed, canola, and ryegrass (*Lolium perenne* L.) grown as green manure crops in the rotation reduced the disease in the subsequent potato crop by 15–40%, and canola and rapeseed reduced black scurf (caused by *R. solani*) by 70–80% relative to a standard oat rotation. At another field site where common scab (caused by *Streptomyces scabies*) was the primary disease problem, using Indian mustard as a green manure reduced the incidence of the disease by 25%, while rapeseed, yellow mustard and ryegrass also reduced black scurf relative to a standard ryegrass rotation. Disease reductions were not always associated with higher glucosinolate-producing crops but were also observed with non-Brassica crops, barley and ryegrass, indicating that other mechanisms and interactions are important, particularly for control of *R. solani*. Overall, Indian mustard was most effective for reducing powdery scab and common scab diseases, whereas rapeseed and canola were most effective in reducing *Rhizoctonia* diseases (Larkin and Griffin, 2007).

5.2. Impact on Plants and Crop Yields

Numerous studies have investigated the effects of organic amendments on crop yields. A one-time application of olive pomace at 10 or 20 Mg ha^{-1} increased wheat yields by up to 50% through the increase of kernel weight and number of kernels (Brunetti et al., 2005). Long-term trends in crop production have demonstrated that as the yield potential has increased, production has often been greater on soils with more organic matter compared to those on soils with less (Johnston et al., 2009). Organic amendments might be beneficial for crops even when they are not nutrient rich. In the long-term experiments at Rothamsted (U.K.) yields of spring crops (barley and potato) increased from nutrient-poor peat applications with or without additional mineral nitrogen, whereas winter cereals showed no benefit when mineral N was also applied (Johnston et al., 2009). This was attributed to the beneficial effect of the peat-improved soil structure on allowing a fast development of the root system for spring cereals. Similarly, a recent study in Italy showed that application of 30 or 60 Mg ha^{-1} of biochar considerably increased durum wheat grain yield (by 23–39%) and straw production, probably due to higher soil temperatures early in the season (Vaccari et al., 2011).

In addition to improvement in soil physical properties and microbial activity, organic amendments provide direct beneficial effects on crop yields through the provision of macro- and micronutrients. In many developing countries, organic amendments represent the only available option for soil fertility replenishment. According to Lal (2005), crop yields can be increased by 20–70 kg ha^{-1} for wheat, 10–50 kg ha^{-1} for rice (*Oryza sativa* L.), and 30–300 kg ha^{-1} for maize (*Zea mays* L.) with every 1 Mg ha^{-1} increase in soil organic carbon pool in the root zone.

Despite the beneficial changes in soil from the use of organic amendments, application of the same amount of nutrients in the form of inorganic fertilizers can result in similar crop yields. The application of farmyard manure for 15 years produced almost double the yields of wheat and maize compared to unfertilized control. However, the application of similar amounts of inorganic fertilizer resulted in the same level of production (Liang et al., 2012). Edmeades (2003) reviewed long-term studies comparing yields at comparable N inputs and found no difference between organic amendments and fertilizers with regard to crop yield or pasture production. He concluded that large amount of organic matter inputs over long periods of time are required to achieve larger yields than comparable fertilizer inputs. Similarly, a recent study in Pakistani Kashmir showed that it is possible to produce the same amount of wheat yield while replacing 25% of urea N with poultry manure or cattle farmyard manure that contributed an equivalent amount of N (Abbasi and Tahir, 2012). The effects of nitrogen, improvements in soil P availability, and other factors are discussed in detail in Johnston et al. (2009).

The effects of green manures, including red clover (*Trifolium pratense* L.), buckwheat (*F. esculentum* L.), millet (*Echinicloa crus galli* L.), mustard (*Brassica hirta* Moench), and colza (*Brassica campestris* cv. oleifera L.) on soil organic matter, wheat yield and N nutrition were studied in eastern Canada by N'Dayegamiye and Tran (2001). The incorporation of green manure into the soil increased soil C content, N content and microbial respiration to the greatest extent after mustard and least after red clover. Green manure contributed 25–31% of the total N uptake by wheat in the season following incorporation. Maximum wheat yields were larger following colza, mustard and millet than after buckwheat or clover. Application of green manure together with a fertilizer-N application of 30 kg ha^{-1} gave heavier yields than a control treatment without green manure but with 90 kg fertilizer N ha^{-1}. Warman (1991) investigated the effect of alsike clover (*Trifolium hybridum* L.), sweet clover, red clover (single or double cut) and buckwheat as green manure crops on oats (*Avena sativa* L.) production, also in eastern Canada. Incorporation of red clover gave the heaviest yields, followed by alsike clover, while buckwheat gave the smallest yields (Warman, 1991).

There can also be some negative impacts on crop plants of nonlegume green manures, which may immobilize soil N during decomposition. In Norway, there was a temporary net N immobilization of up to 0.9 g N m^{-2} in soil incubated with ryegrass compared with unamended soil. In clover-amended soil, mineral N exceeded that in unamended soil by up to 5 g N m^{-2} (Breland, 1996).

6. THE EXPOSURE OF HUMANS TO THREATS FROM CONTAMINANTS IN ORGANIC AMENDMENTS THROUGH AIR, WATER AND FOOD

With the exception of the failure of a liquid manure store, the greatest risk to the health of humans from the use of organic amendments is associated with their application to the land. Potential pathways of exposure to contaminants from the amendments include their transport in air or water moving from the site of application. Movement in air may be the result of bioaerosol formation or lead to the creation of airborne PM. Movement in runoff or leaching from soil to surface or groundwater can result in contaminated water resources that are ingested during recreation activity, such as swimming, or directly imbibed from domestic drinking water supplies. Both these latter pathways have resulted in disease outbreaks associated with *E. coli* O157:H7 (e.g. Ackman et al., 1997; Goss and Richards, 2008). The water may also be used in preparation and manufacture of food that is then eaten. The most direct pathways are direct ingestion of fresh fruit and vegetables or the soil in which these have been grown with the aid of organic amendments.

6.1. Impact of Application Techniques and Timing of Operations on Exposure

There are three main approaches to land application of organic amendments other than green manures: broadcasting (solid, semisolid, and liquid amendments), irrigation, and injection (liquids). Banding of amendments can be considered as a special case of broadcasting or injection that may involve surface or subsurface application. The mode of application has the greatest effect on the amount of ammonia that is lost by volatilization at this stage, with more being lost after broadcasting than injection (Beauchamp, 1986; Søgaard et al., 2002). Shallow incorporation in arable fields (top 10 cm) of liquid manure may have little benefit over broadcasting (Leytem et al., 2009). The more nitrogen lost through this route, the less that is potentially available to plants or at risk of later loss to water resources. Uniform distribution is essential if producers are to rely solely on the nutrients contained in an amendment to meet the needs of their crops. The common periods for application are in the fall, winter and spring. Spring applications may be made before planting, as a side- or top-dressing. Spreading manure on the land in fall or winter results in smaller recovery of applied nitrogen by the crops, while the risk of surface runoff, leaching and denitrification is greater (Fleming and Fraser, 2000; Goss et al., 1995; Thompson et al., 1987).

Solid amendments are usually broadcast using flails to distribute the material from the side or back of a spreader. The greatest problem is the lack of uniform spreading, partly due to the nature of the material, which is often heterogeneous in the size and weight of particles, and partly because of the mechanical action of the flails. Broadcasting of liquids has traditionally used a splash-plate for distribution, and this is still used for liquid manure. In addition, there are low-level or low-pressure nozzles on booms, which offer the advantage that liquids can be distributed in bands between rows. Uniformity of application is less of an issue, but can result from changes in topography causing localized runoff and accumulation. Irrigation is usually by sprayer that is linked by pipes to the storage system. Odor issues tend to make this technique less acceptable to many. Furthermore, during irrigation the pH of the slurry tends to increase with the loss of volatile compounds, which may contribute to even greater volatilization of NH_3 as the manure droplets reach the soil (Safley et al., 1992). Broadcasting manure and leaving it at the soil surface is likely to reduce the survival of pathogens. Sunlight reduces the longevity of bacteria, directly through the effect of ultraviolet light and as a result of drying (Gerba and Bitton, 1984; Mubiru et al., 2000; Sinton et al., 1999). However, to capture as much of the NH_4^+–N as possible, broadcast soil amendments, especially those applied as a liquid, need to be incorporated as soon as possible after application (Beauchamp et al., 1982; Chantigny et al., 2004b). This also reduces the risk that contaminants will be carried off-site in surface runoff water. However, it appears that incorporation of liquid amendments by disking forms bioaerosols with a greater emission rate than does the application process (Paez-Rubio et al., 2006, 2007). Injection is associated with less bioaerosol formation than surface spreading (Millner, 2009). Spreading of dry waste, such as poultry litter, can also create considerable amounts of particulate material and bioaerosols (Millner, 2009).

Formation of bioaerosols and particulates is an important aspect of the airborne risk to humans. However, a national survey in the USA found that local residents faced little risk of infection from microorganisms that became airborne during the act of spreading sewage biosolids (Brooks et al., 2005). Nevertheless, there was an annual risk of virus infection of 34% for workers involved in the land application (Tanner et al., 2008). Boutin et al. (1988) investigated the respiratory risk to people living "nearby" from microorganisms released during land spreading of manure from pig and cattle operations. These authors identified the greatest risk as the use of irrigation guns that had a strong upward trajectory but the concentration of bacteria in the bioaerosols was less than that observed from spreading the sewage biosolids. Boutin et al. (1988) and Hutchison et al. (2008) concluded that the intake of pathogens via the respiratory pathway as a result of manure spreading posed a low level of risk to human health. However, Hutchison et al. (2008) recognized that the potential for the contamination of fresh vegetable crops and surface water resources needed to be considered as indirect pathways for contaminants to reach people. In a review of the fate and transport of aerosols containing biological particles from animal production operations Dungan (2010) identified current limitations in estimating risks to the health of the community beyond the farm gate. None of these studies considered the risks associated with the incorporation of soil amendments, so the work of Paez-Rubio et al. (2006), which indicated that this produces more intense bioaerosols, suggests that a more holistic approach is required.

6.2. Potential for Exposure after Land Application

The main pathways of exposure to contaminants after the application process are via water or the ingestion of contaminated plant material. The soil can be a major barrier to contaminants entering these key pathways and reduction in their concentration in transporting water is an important mechanism.

6.2.1. Pathogens

Several factors influence the persistence of pathogens in soil (Table 5.15). Some relate to the likelihood that they will survive. Even pathogens can survive for prolonged periods outside their normal hosts. Organisms introduced with the organic amendment are subject to competition or even predation from living organisms in the soil (Botzler et al., 1974; Bradford and Segal, 2009; Cools et al., 2001; García et al., 2010; Jiang et al., 2002; Unc and Goss, 2006a). However, there is also evidence that some soil organisms can act as hosts to pathogens and extend their persistence in the environment (Douesnard-Malo and Daigle, 2011).

6.2.1.1. Bacteria

Earthworms can greatly reduce the numbers of bacteria in the soil, while free-living protozoa, nematodes, and the soil bacterium *Bdellovibrio* also prey on bacteria in the soil (Peterson and Ward, 1989).

How microbes move within the soil can have an important role in deciding whether they find niches, where they are protected from predation but have a source of carbon and other nutrients. A number of microbes are self-motile, but are not necessarily able to move a long distance in this way and movement is generally associated with the percolating water following precipitation. Microorganisms adsorb onto soil particles, especially clays with a large surface area (MacLean, 1983), which allow them to survive

Table 5.15 Factors in the retention and survival of enteric bacteria and viruses in soil

Factor	Enteric bacteria	Viruses
Moisture content	Greater survival time in moist soils and during times of high rainfall	Some viruses persist longer in moist soils than dry soils
Moisture-holding capacity	Survival time is less in sandy soils than in soils with greater water-holding capacity	Soils with a larger water-holding capacity will retain moisture longer than those with a smaller capacity
Temperature	Longer survival at low temperatures; longer survival in winter than in summer	Viruses survive longer at lower temperatures
pH	Shorter survival time in acid soils (pH 3–5) than in alkaline soil	Most enteric viruses are stable over the pH range 3–9: survival may be prolonged at near-neutral values
Sunlight	Shorter survival time at soil surface	Desiccation will reduce survival
Organic matter	Increased survival and possible regrowth when sufficient amounts of organic matter are present	Presence of organic matter may protect viruses from inactivation; others have found that it may reversibly retard virus infectivity
Antagonism from soil microflora	Increased survival time in sterile soil	Some viruses are inactivated more readily in the presence of certain microorganisms; but, adsorption to the surface of bacteria can be protective
Salt species and concentration	Salt concentration is important for the initial retention of bacteria in soil	Some viruses are protected from inactivation by certain cations; the reverse is also true
Association with soil	Adsorption onto soil particles is important for bacterial retention	In many cases, survival is prolonged by adsorption to soil; however, the opposite has been observed
Aggregation of microbes	Reduces their mobility and hence increases the retention of bacteria in soil	Enhances survival
Soil properties	The mobility, and hence the residence time of bacteria, is very dependent on the surface properties of soil particles	Effects on survival are probably related to the degree of virus adsorption
Microbe physiology	Different pathogenic bacteria vary widely in their response to environmental stress.	Different virus types vary in their susceptibility to inactivation by physical, chemical and biological factors

(Based on Gerba et al. (1975), Yates and Yates (1988)).

longer than those in the liquid phase, as organic substrate and nutrients are more readily available to them (Sobsey, 1983). The presence of roots can be one source of carbon for microbes (Goss and Kay, 2005; van Veen et al., 1997) and it can enhance their persistence (Avery et al., 2004; Turco and Brouder, 2004). There is also evidence that some plant roots release compounds that reduce persistence of enteric pathogens in soil (Rothrock et al., 2012). Given that soil microbes are generally carbon starved and that top soil undergoes periods of wetting and drying so that the water supply is uncertain, the survival of an organism depends on soil properties such as texture, pH, water and organic carbon content, adsorption properties, availability of nutrients, the temperature regime together with the composition of the indigenous population of soil organisms, and the chemical and physical nature of the organic amendment. They also depend on their own survival mechanisms (MacLean, 1983). Among bacteria, many Gram-positive organisms form resistant spores, whereas in Gram-negative organisms physiological adaptation to environmental stress may involve the reduction in cell size and metabolic rate (Roszak and Colwell, 1987).

Although persistence in soil is very variable, bacteria introduced with manure can survive in soil for periods up to 120 days (Table 5.16) (Nicholson et al., 2005; Nyberg et al., 2010; Rogers et al., 2011). Variation in persistence can result from differences between the various sources (e.g. animal species), the form of amendment (liquid or solid) as well as soil type

Table 5.16 Persistence of pathogenic bacteria from animal manure in agricultural soils after land application and incorporation

Type of livestock manure	Persistence (days)			
	VTEC	*Salmonella*	*Campylobacter* spp.	*Listeria* spp.
Liquid manure				
Dairy	64	120	34	120
Beef	16	120	64	120
Pig	16	56	36	120
Solid manure				
Dairy	34	120	64	120
Beef	64	34	120	120
Sheep	16	120	34	120
Pig	16	120	34	120
Broiler chickens	16	56	64	56

(Based on Hutchison et al. (2004b)).

(Nicholson et al., 2005; Unc and Goss, 2006a). Although *E. coli* appears to persist longer if applied in solid rather than liquid amendments, Semenov et al. (2009) obtained the opposite result in one study with *Salmonella* serovar Typhimurium. Enhanced soil organic matter content favors the persistence of enteric bacteria (Franz et al., 2008; Cools et al., 2001) as does a high level of nutrients (Gagliardi and Karns, 2000; Ravva and Korn, 2007; Unc et al., 2006).

Soil temperature is an important factor in bacterial survival, with persistence being longer in cold soils than in warm soils (Cools et al., 2001; García et al., 2010). However, the availability of water in soil overrides the impact of other factors (Gerba and Bitton, 1984; Mubiru et al., 2000). Cools et al. (2001) found that the best survival of *E. coli* and *Enterococcus* spp. occurred in soils close to field capacity, while Garfield and Walker (2008) found that *E. coli* survived for less than 3 days at water potential of −22.4 MPa in animal feces. In a study of the persistence of 18 strains of *E. coli* O157, Franz et al. (2011) found that microbial survival time in a manure-amended sandy soil was directly related to the ability to oxidize propionic acid, α-ketobutyric acid and α-hydroxybutyric acid, with isolates from farm animals taking longer to decline to the limit of detection than those from humans.

Solomon et al. (2002) and Looper et al. (2009a) observed movement of *E. coli* O157:H7 into plant tissue under field conditions and this was supported by laboratory studies (Warriner et al., 2003). Although these reports were challenged by Johannessen et al. (2005) and Zhang et al. (2009), further laboratory investigations by Fink et al. (2012) found that on undamaged lettuce leaves, *E. coli* O157:H7 altered the regulation of genes involved in the production of fibers used for attachment, the suppression of biofilm formation (upregulated) and metabolic rate (downregulated). These reports raise concerns about the use of organic amendments to provide nutrients for fresh-market vegetables, even if association with plant cells occurs infrequently (Erickson et al., 2010), especially as one multistate outbreak of infection is believed to have originated from manure contamination of a lettuce crop (Hilborn et al., 1999).

6.2.1.2. Viruses

Factors influencing the effectiveness with which soils retain viruses are essentially similar to those identified for enteric bacteria (Table 5.15). Viruses near the soil surface are rapidly inactivated by the stresses imposed by sunlight, soil drying, predation, and other soil-based factors such as pH. Kowal (1985) reviewed the literature on virus survival. Moisture content appears to be a major factor once the virus has penetrated the soil surface as they are inactivated by desiccation (Yeager and O'Brien, 1979). About 100 days is the longest survival time for enteric viruses (Gerba et al., 1975).

6.2.1.3. Protozoan Parasites

The survival of *Cryptosporidium* in soil was related to pH, with greater numbers of oocysts being found in acid soils (Barwick et al., 2003). *Giardia* was more likely to be found on wet soils than in dry conditions (Barwick et al., 2003). The presence of manure appears to change the cell physiology of *Cryptosporidium* such that it is less susceptible to environmental pressures (Robertson et al., 1992).

6.2.2. Macro Nutrients and Trace Elements

Organic N compounds are slowly mineralized to NH_4^+–N, which can be lost by volatilization or oxidized by soil microbes to NO_2^-–N and NO_3^-–N. Plants can utilize NO_3^-–N and NH_4^+–N, so the amount available to be lost to surface and ground water resources depends greatly on the instantaneous demand of the crop and the rate at which water can carry the nutrients out of the rooting zone. Soil chemistry greatly limits the amount of NH_4^+–N that is present in solution compared with NO_3^-–N.

Most of the P applied in organic amendments to the soil becomes associated with clay minerals, sparingly soluble calcium phosphates or is in organic form. Generally, the concentration in solution is limited. Most of the trace elements applied also become bound in clay minerals and soil organic matter.

6.2.3. Antibiotics and Other Pharmaceuticals

Antibiotics and other pharmaceuticals readily undergo sorption onto organic matter, and so can remain with dissolved and particulate material on entering the soil. They are more likely to link with organic matter than the mineral fraction, but adsorption is stronger on swelling clays than on sand (Litskas et al., 2011; Thiele-Bruhn, 2003). Diffusion into pores within aggregates, hydrogen bonding, electrostatic linkages with minerals through cation bridges, and cation exchange provide the key means of sorption and fixation. The sorption tends to be pH-dependent (e.g. Rabølle and Spliid, 2000), with sulphonamides being more adsorbed in acid soils (Thiele-Bruhn, 2003). Jjemba (2002a) summarized reports on the persistence of various pharmaceuticals after application to soil and wide variation in their recovery by various extractants. There was some indication that the compounds were more labile at higher temperatures.

Many antibiotics are not readily degraded by microbial action in the soil (Chee-Sandford et al., 2009) and so are persistent. However, the antimicrobial compounds triclocarban and triclosan remained in the organic matrix of incorporated dewatered sewage biosolids, whereas in liquid biosolids they were more readily accessible to microbial action (Al-Rajab et al., 2009). Some 45% of the triclosan applied was apparently converted into forms or products that were more tightly bound than the parent compound, whereas for triclocarban this fraction only accounted for 15% of that applied. Nevertheless, 40% of the triclosan stayed in the extractable pool compared with 30% of the triclocarban (Al-Rajab et al., 2009).

Some antibiotics in manure have been shown to affect plant growth and yields. Jjemba (2002b) reported reduced growth or seedling death of soybean (*G. max* (L.) Merr.) in the presence of some antibiotics, indicating their adsorption by plants. The concentration of a number of veterinary antibiotics taken up into leafy tissue or roots was used to predict the adult human exposure from these compounds and was found to represent only about 10% of the acceptable daily intake (Boxall et al., 2006). However, by analogy, this pathway of exposure could be significant for antibiotics with a small acceptable daily intake or when a number of pathways operate concurrently (Boxall et al., 2006).

The important area of concern over the use of organic amendments from animals routinely treated with antibiotics and sewage biosolids is the possible increase in microbial resistance to multiple antibiotic drugs. There is little evidence that bacteria in soils subject to regular manure applications have developed more antibiotic resistance because of the feeding of subtherapeutic antibiotic doses to enhance growth of livestock and poultry (Chee-Sandford et al., 2009). However, any impact on the survival of enteric pathogenic microbes due to the use of antibiotics in feed has not been evaluated.

6.2.4. Natural Hormones and Endocrine Disrupting Substances

Once incorporated into the soil, natural hormones in manure do not appear to persist, but are readily broken down by soil microbes (Lucas and Jones, 2006; Xuan et al., 2008). The presence of organic amendments in the soil enhances the microbial degradation in short-term studies (Lucas and Jones, 2006; Stumpe and Marschner, 2010a). However, Lucas and Jones (2009) concluded that the presence of urea could impair the microbial degradation of natural hormones, making them more persistent in the environment. Furthermore, Stumpe and Marschner (2010a) found that degradation of hormones was not consistently modified in the soil receiving amendments with large organic matter contents over long periods. These same authors suggested that hormones adsorbed onto dissolved organic C compounds in soil amendments can exchange between these and soil-based dissolved organic C (Stumpe and Marschner, 2010b). The association with organic C makes the endocrine disrupting compounds less readily subject to microbial degradation. Kinney et al. (2008) demonstrated that endocrine disrupting compounds accumulate in earthworms, indicating their persistence in the soil.

Much of the research to date has focused on the persistence and potential for transport of 17β-estradiol rather than 17α-estradiol, although the latter is excreted by beef and dairy cattle in much greater amounts than the 17β-stereoisomer (Mashtare et al., 2011). However, the sorption of 17β-estradiol in soil is approximately twice that of 17α-estradiol, which suggests that the potential for transport could be underestimated in the absence of direct studies using the latter stereoisomer (Mashtare et al., 2011).

7. IMPACTS ON THE QUALITY OF WATER RESOURCES

Organic amendments can improve soil structure and have a direct effect on how much water runs off the soil surface to water courses and how much will infiltrate and permeate to ground water or be intercepted by field drains if these are installed. The water that infiltrates can carry soluble components but it can also leach materials already within the pore system or adsorbed to a greater or lesser extent on soil particles or aggregates. Runoff can carry with it any of the components of the organic amendments in solution, in suspension or as particles. As sunlight is an important factor in killing microbes, any factor that can reduce the transmission of light because of increased suspended solids tends to increase their survival time. This could be important for the survival of some pathogens in surface water resources (Aramini et al., 2000).

There is good evidence of the formation of a surface seal (Smith et al., 2001a,b) or biofilm crust (Hoese et al., 2009) as a result of applying liquid manure. For example, the manure from pig so limited infiltration that an average of 87% of water applied in simulated rainfall was lost in the runoff (Hoese et al., 2009). Even in the absence of an obvious surface barrier, Unc and Goss (2006b) reported changes in surface hydraulic properties following the application of liquid manure from pig that resulted in the increased surface runoff. These same authors showed that when solid manure from beef cattle was applied to the soil surface, infiltration was diverted from preferential flow paths resulting in soils wetting to a shallower depth than occurred with liquid manure (Unc and Goss, 2006b).

Applying organic amendments to wet soil can result in runoff or deep percolation occurring with the next precipitation, resulting in losses of plant nutrients and contaminants to surface and groundwater resources. Adding amendments in the spring as soils are drying can reduce the likelihood of losses (Edwards and Daniel, 1993; Misselbrook et al., 1996; Smith et al., 2001a,b).

7.1. Movement of Contaminants in Surface Runoff

Runoff initiation, volume and contaminant loading are highly dependent on the intensity and duration of individual rainfall events as well as the number of events (Edwards and Daniel, 1993; McLeod and Hegg, 1984). The period of time between application of organic amendments and the next precipitation event is important in determining the level of contaminant loading (Allen and Mallarino, 2008; Edwards and Daniel, 1993; Hubbard and Sheridan, 1983; Hubbard et al., 1991; Lowrance, 1992; Sharpley, 1997; Sistani et al., 2009).

7.1.1. Macronutrients

N, P and organic compounds from soil amendments represent concerns for the surface water quality if they are removed in surface runoff. The N in any organic material left on the soil surface or associated with fine particles, which are readily entrained in runoff and beginning the process of water erosion, can be lost in this way. In consequence, the contaminant can move from fields to surface watercourses. The factors that determine N-loss by water erosion are the amount of sediment moved, the N content of the soil moved and the N content of the solid fraction of the organic amendment. The other form of N that is subject to loss to surface water is that dissolved in runoff water, NH_4^+–N and NO_3^-–N together with soluble organic matter.

Runoff generally accounts for only a small portion of applied N compared with the portion that can be leached out (Burgoa et al., 1993). However, it can be a far more important route for the loss of NH_4^+–N (Smith et al., 2001a). The actual proportions of nutrients lost vary according to cropping practices, and the type and timing of applications of organic amendments. The concentration of NH_4^+–N and the total mass of N in runoff are greater from untreated than composted amendments (Miller et al., 2006). When organic amendments are incorporated rather than left on the surface, the losses via surface runoff of NH_3^-–N (Sistani et al., 2009) and NO_3^-–N (Adeli et al., 2011) are reduced. Nutrient losses are determined by the nutrient status of the soil rather than the properties of the amendment (Miller et al., 2006). Kirchmann (1994) reviewed 15 reports of NO_3^-–N in runoff and water percolating below soils amended with animal manures. The concentration in percolating water was greater under arable than grassland soils. The concentration in runoff was less than in percolating water, with the concentration in the latter tending to be related to the rate of manure application.

Applying organic amendments usually increases the P content of the soil and hence the concentration on any eroded sediment. For nonparticulate P, the loss in runoff is related to the soluble P fraction in the material applied (Kleinman et al., 2004). On the other hand, as organic amendments tend to improve soil structure, there tends to be less erosion of soil particles. Incorporating manure reduces the amount of P that is at risk of runoff, but the action of tillage implements can increase the erodability of the soil. The net effect is still to reduce runoff loss compared with no incorporation (Allen and Mallarino, 2008; Kaiser et al., 2009; Sistani et al., 2009).

7.1.2. Bacteria and Viruses

Thelin and Gifford (1983) showed that if fresh fecal material was subject to precipitation within 5 days of application, then the concentration of fecal coliform bacteria in runoff was of the order of 10^4 mL^{-1}, but this number declined to 4×10^2 mL^{-1} after 30 days. Sistani et al. (2009) showed that increasing the time between the application of poultry litter and the first precipitation event reduced the concentration of *E. coli* in runoff, but effects were not as great as placing the litter in a subsurface band. The movement in surface runoff of viruses from organic amendments has not received significant attention.

7.1.3. Antibiotics

There is limited information on the presence of antibiotics in runoff after the application of organic amendments, but Kay et al. (2005b) investigated the effect of precipitation 24 h after applying three veterinary antibiotic drugs in liquid manure from pig. Water in runoff along tramlines in the field contained sulphachloropyridazine at a concentration of 703 μg L^{-1} and oxytetracycline at 72 μg L^{-1}, which when combined with the total flow represented 0.4 and <0.1% of the material applied. Blackwell et al. (2007) reported somewhat similar findings, but with much smaller concentrations of the two antibiotics being detected. Kay et al. (2005a) could not detect in runoff any of the tylosin applied. In contrast, Hoese et al. (2009) reported that the total tylosin removed in surface runoff from clay, silt and silty clay loam soils was equivalent to between 8.4 and 12% of that applied in liquid manure from pig. However, the amount of chlortetracycline removed in surface runoff ranged from 0.9 to 3.5% of that applied in the manure (Hoese et al., 2009). Burkhardt et al. (2005) also reported the formation of a surface seal following an application of 30 m^{-3} ha^{-1} liquid manure from pig to a grassland loam soil. Although the film enhanced the surface runoff 6-fold compared with control plots receiving no manure, loss of any sulphonamide antibiotic (sulfadiazine, sulfathiazole and sulphadimidine) in runoff was <1% of that applied if irrigation was applied after 24 h of application or a maximum of 2.1% if the irrigation was delayed for 72 h (Burkhardt et al., 2005).

Sabourin et al. (2009) determined the loss in surface runoff of the antimicrobial pharmaceuticals triclosan and triclocarban from dewatered municipal biosolids incorporated into the top 15 cm of a gray brown Luvisol. Although the concentrations in the soil amendment were similar, approximately 40 times more triclosan was removed in surface runoff than triclocarban. However, this represented <1% of material applied to land.

Total loss of lincomycin in surface runoff from cropland to which liquid manure from pig was applied was greater from fall than spring applications (Kuchta et al., 2009). Following fall applications, losses were similar from rain-induced runoff and from that induced by snowmelt water. Nonetheless, total losses were <1% of the content of the manure spread (Kuchta et al., 2009).

Although the antibiotics were not applied in organic amendments, Davis et al. (2006) determined the loss in runoff from a freshly cultivated sandy clay loam for two tetracyclines (tetracycline and chlortetracycline), two sulphonamides (sulfathiazole and sulfamethazine), two macrolides (tylosin and erythromycin) and the polyester, monensin. The losses in water and in sediment were differentiated. For the tetracyclines, the average loss in runoff was 0.003% of that applied of which almost half was carried on sediment. Loss in sediment of the sulphonamides and monensin were approximately 10% of total loss although the total loss of the polyester antibiotic (0.08% of that applied) was four times that of sulfathiazole or sulfamethazine (each approximately 0.02% of applied). In contrast, about 75% of the total loss of the macrolides was associated with the sediment and the total loss of tylosin being about half that of erythromycin (0.05% of applied). Dolliver and Gupta (2008) applied chlortetracycline, tylosin and monensin in solid manure from beef cattle or chlortetracycline and tylosin in liquid manure from pig in late fall in each of three years and measured the subsequent losses in runoff water. The largest losses as a proportion of that applied were associated with runoff from snowmelt. Losses of tylosin were greater following the solid manure applications than from the liquid manure, whereas chlortetracycline was only found in runoff from the land treated with liquid manure and losses were much less than for tylosin. For the same year and manure type, the loss of monensin in this study was much greater than that of tylosin (Dolliver and Gupta, 2008).

7.1.4. Natural Hormones and Endocrine-Disrupting Substances

Most studies of surface runoff have only reported the transport of the natural hormones from manure. The most detailed investigation (Nichols et al., 1997) showed that the concentration of 17β-estradiol in runoff was directly proportional to the application of broiler chicken litter per unit area. The longer the litter remained on the soil surface, the smaller was the concentration of 17β-estradiol in runoff. Similar results were obtained by Finlay-Moore et al. (2000) and they also found that testosterone behaved in a similar fashion. However, concentrations of testosterone in runoff were smaller than those of 17β-estradiol. They also found that both hormones could accumulate in the soil after several applications of poultry litter. In contrast, Jenkins et al. (2006, 2008) indicated that at appropriate agronomic rates of poultry litter application, the concentration of testosterone in runoff increased above background, whereas that of 17β-estradiol in soil or runoff did not.

In addition to the free forms of natural hormones, runoff from fields to which organic amendments have been applied can contain metabolites that are easily converted to active compounds, even though in the conjugated form they are benign. Dutta et al. (2010) found that the concentration and total mass of conjugate estrogens in the runoff from land to which poultry manure was applied were greater than those of the free forms. They recommended that both forms should be considered when assessing the threat to health.

7.2. Movement of Contaminants by Infiltration and Leaching

The complexity of the pore system can result in a wide range of transport velocities within the same soil layer, depending somewhat on texture but mostly on structural development. This may limit the dispersion of contaminants between finer pores with very slow rates of flow and larger pores, where flow is much faster (Jardine et al., 1990). At peak flows, most water will move preferentially through the largest continuous pores that are water filled. Tile drain systems provide some continuous flow paths between the soil surface and the drain, thereby allowing rapid transport between the field and a receiving surface watercourse (Goss et al., 1983).

7.2.1. Macro- and Micronutrients

In many cases, dissolved N is transported into the soil with the initial infiltration that precedes runoff, so this portion is usually small but highly variable (Blevins et al., 1996; Meisinger and Randall, 1991). It depends on a number of factors, such as the degree of soil cover, source of N applied, application rate, timing and duration of the application. The major N species lost by leaching is NO_3^-–N. In the case of subsurface flow to tile drains, there can be large losses of N (Vinten, 1999). On arable land in the Netherlands, annual losses of N through tile drains averaged as much as 22 kg N ha^{-1} during a 10-year period (Kolenbrander, 1969).

Nielsen and Jensen (1990) reported that NO_3^-–N losses from the rooting zone in soils receiving liquid manure were greater than those from a similar soil to which the same amount of N had been applied as inorganic fertilizer. Bergström and Kirchman (1999) compared the loss of N by leaching from 100 kg N ha^{-1} applied to a sandy soil as solid poultry excreta with that after the same rate of NH_4NO_3 was applied. The N in the poultry manure and both the NH_4^+–N and NO_3^-–N in the fertilizer were labeled with ^{15}N. Spring barley was grown and leaching losses were investigated over three seasons, with the next two crops receiving 100 kg N ha^{-1} as unlabeled NH_4NO_3. Leaching losses of labeled N from the manure application were considerably greater than those from the original fertilizer application in all years. Bergström and Kirchman (2006) also reported greater leaching loss of N from a sandy soil following the application of liquid manure from pig than after adding mineral fertilizers. In both experiments, the increased leaching loss associated with manure was related to a smaller uptake of N by the crops than when mineral fertilizer was used (Bergström and Kirchman, 1999, 2006).

Few studies exist that compare the potential for nitrate leaching of different organic amendments, although more information is available for different sources (animal species) or types (solid, liquid, farmyard manure). Younie et al. (1996) found that nitrate leaching was greater where liquid cattle manure was the source of nitrogen than where solid beef manure was used. Ritter et al. (1990) studied soil nitrate profiles under 16 sites, some of which received fertilizer N, either alone or in combination with either broiler manure or liquid swine manure. Although direct comparison of manure types was not made on the same site, N application rate was found to be the major determinant of N in the soil profile. It appears that manure from poultry, cattle or pig operations has the potential to contaminate ground water, especially if it is applied at excessive rates.

When economically optimum rates of N were applied to row crops, such as maize, the concentration of NO_3^-–N in leachate from the root zone exceeded 10 mg L^{-1}, the quality guideline value for drinking water (Jemison and Fox, 1994; Toth and Fox, 1998). If dairy manure was applied to maize, the loading of NO_3^-–N was similar or slightly smaller loading than agronomic equivalent rates of fertilizer N (Jokela, 1992). Jemison and Fox (1994) found very little difference in NO_3^-–N concentrations or mass of NO_3^-–N leached between non-manured maize and maize manured at the economically optimum rate.

Timing of application is also an important factor determining the total NO_3^-–N lost by leaching. An early fall application of solid manure from dairy cattle resulted in about 6–10% of the N applied being leached over the following winter, and a 21% greater loss than nonmanured soil, when summed over 3 winters (Thomsen, 2005). In contrast, the total N lost following a spring application was only 13% greater than that from nonmanured land. Leaching losses under continuous maize also showed that applications of liquid manure from dairy cattle resulted in greater leaching when made in early fall compared with early spring, with a late fall application being somewhat intermediate (van Es et al., 2006). Splitting the early spring application had no effect on the loss by leaching. Following the fall application to loamy sand, the flow-weighted mean concentrations in drainage water were twice those in clay loam (van Es et al., 2006). Hernandez-Ramirez et al. (2011) reported a 40% difference in the leaching of NO_3^-–N following a fall application of liquid manure from pig (33.3 kg ha^{-1}) compared with one made in spring (19.8 kg ha^{-1}). The different results for the amount of NO_3^-–N leached following manure applications highlight the importance of N transformations, such as mineralization and denitrification, as well as differences in uptake of N by crops that in combination influence the availability of nitrate in the soil. Differences in precipitation and the uptake of water by crops modify the potential for deep percolation and N transport.

The inclusion of forage crops, such as alfalfa, in a rotation greatly reduced the amount of NO_3^-–N leaving a farm in leachate (Toth and Fox, 1998). Similarly, consistently smaller losses of N in leachate have been observed under grass crops than under maize or other row crops (van Es et al., 2006; Hernandez-Ramirez et al., 2011). These observations may relate to the longer period of water removal under forage crops, which results in less drainage, or a more effective root distribution leading to greater uptake and immobilization of N by a perennial crop. However, injecting anaerobically digested sewage biosolids into freely draining grassland soils resulted in 24% of total N-applied being lost in drainage water compared to only 11% if the biosolids were surface-applied (Misselbrook et al., 1996). If raw sludge was injected, only 6% of total N was lost.

Although leaching of P is generally considered less significant for the environment than the leaching of nitrate, particular combinations of agricultural management practices, soil properties and climatic conditions can lead to significant losses of soluble and particulate P through leaching (Sims et al., 1998). Less P is likely to be lost by leaching than by surface runoff (Sharpley and Withers, 1994; Smith et al., 2001a,b). Leaching of P from organic amendments may occur in both inorganic and organic forms (Campbell and Racz, 1975; Eghball et al., 1996). Complexation of P with mobile organic compounds may favor the deep transport of P in organic forms even through layers with a great P adsorption capacity, such as carbonate soil layers. Experimental results from Eghball et al. (1996) showed that P from mineral fertilizer had not moved below a carbonate layer (0.9 m) of soil even after 40 years of mineral P fertilization while organic P from manure moved to 1.8 m. The ability of this soil to adsorb P did not affect its loss by leaching. The P associated with organic acids of small molecular weight also had increased mobility because of greater dissolution. This form of P also tends to have greater bioavailability (Bolan et al., 1994). Increasing the proportion of labile, weakly bound P results in greater vulnerability of manure-treated soils to lose P by leaching (Johnston and Poulton, 1997; Robinson et al., 1995; Stephenson and Chapman, 1931). This results in deeper penetration of P compounds after the manure application (Campbell and Racz, 1975).

The leaching of P is not related to crop uptake as reported for N. The N-use efficiency (dry matter yield per mass of N acquired) of oat and barley crops was greater when the N source was mineral fertilizer than when it was manure and the same was true for P-use efficiency with these two sources of P (Bergström and Kirchman, 2006). However, the P-use efficiency was much smaller than the N-use efficiency and leaching tended to decrease with increases in manure-P application. Leaching of N increased with manure-N application (Bergström and Kirchman, 2006).

As P is a reactive ion, soil enrichment generally decreases sharply with depth. Application of cattle feedlot waste resulted in an increased proportion of available P in the first 30 cm, but with little increase below 50 cm (Campbell and Racz, 1975). Decreasing enrichment or only slight enrichment with depth does not necessarily indicate the absence of leaching because the residence time of some drainage water in the subsoil may have been too short, perhaps because of preferential flow, to allow absorption of P on to soil particles (Johnston and Poulton, 1997). Furthermore, some subsoils, e.g. those in sandy soils, may have limited capacity to retain P.

Leaching of P in water-soluble and particulate forms from soil is enhanced by the presence of tile drains (Harrison, 1987). Heckrath et al. (1997) found a critical concentration of soluble P in the plowed layer that, if exceeded, resulted in an enhanced contribution of P losses through tile drain in clay loam soils. Hergert et al. (1981) found P losses in the field tile drain effluent to be increased where manure was applied compared to unfertilized control plots. Dils and Heathwaite (1997) found that subsurface transport of P could occur as both water-soluble and particulate P in both undrained and tile-drained land.

At the field scale, even when sorption capacities in the surface horizon are exceeded and the water-soluble P concentration becomes elevated, deeper soil horizons may be able to sorb the leaching P and minimize the potential for water-soluble P movement to surface waters via drainage (Provin et al., 1995). Similarly, immobilization of P on metal-oxide coatings within an aquifer can decrease as the P loading of soil increases (Walter et al., 1996). However, preferential flow can be an important factor in tile drain losses of particulate P (Gaynor and Findlay, 1995; Heckrath et al., 1997). The importance of preferential flow for P transport was confirmed by its leaching from soils with large adsorption potential in subsoil layers (Eghball et al., 1996; Thomas et al., 1997).

Transport of micronutrients and trace metals from organic amendments through soil is mostly a function of pH and dissolved oxygen concentration. Accumulation of organic material close to the surface may possibly decrease the availability of Zn while increasing the solubility of Fe and Mn (Shuman, 1988). Other studies show that although the addition of organic material to soil in manure tends not to alter the total dissolved Zn as it changes from being linked with light molecular weight organic particles to being associated with heavy organic particles. These heavier particles tend to be adsorbed by the soil complexes (Del Castilho et al., 1993b). On the other hand, Cu interacts with dissolved organic carbon of low molecular weight, which is more likely to stay in solution, and thereby increase the percentage of mobile Cu (Del Castilho et al., 1993b; König et al., 1986). Increases in the ionic concentration of the soil solution, as happens shortly after manure application to soils, may decrease the percentage of metallic ions attached to soil mineral and organic particles by increasing the competition for adsorption sites (Stevenson, 1991).

7.2.2. Bacteria

Bacteria easily enter tile-drains when liquid manure is applied to the land following farming guidelines current in North America (Ball Coelho et al., 2007; Foran et al., 1993; Lapen et al., 2008) and can contaminate shallow and deep groundwater aquifers (Goss et al., 1998).

Transport of bacteria through soil is concentrated in regions of preferential flow (Goss et al., 2010). Consequently, there tends to be less vertical movement in sandy soils than soils with more clay that have greater structural porosity (Bech et al., 2010). Such soils also tend to have a larger water content that maintains microbial activity. Bacteria can be removed from the aqueous stream in which they are moving through the soil pore system by the process of straining at pore necks or by reversible sorption onto solid surfaces (Mosaddeghi et al., 2009). They may then remobilize at some later time. This discontinuous transport creates an apparent retardation of the bacteria relative to nonadsorbed tracers, which can be as large as 10 in porous aquifers (Matthess et al., 1988). However, Harvey (1991) showed that in fact the transport of bacteria could be faster, slower or similar to that of tracers. Travel may be significantly faster than conservative tracers due to their motility (Jenneman et al., 1985) and because their transport is restricted to macropores away from the finer pores with slow movement. Exploitation of faster pathways also may be constrained as travel may occur only during peak flow (Bitton and Harvey, 1992).

In addition to the physical conditions of the soil there are characteristic of the microbes that may influence their transport, including their size, motility and surface properties, including surface charge and hydrophobicity (Gannon et al., 1991). For microbes present in soil amendments applied to soil, the properties of those materials can also be of considerable importance (Guber et al., 2007; Mosaddeghi et al., 2009; Semenov et al., 2009; Unc et al., 2004, 2012).

Unc et al. (2012) applied filtration theory in an integrated analysis of the transport of microbes or particulate tracers applied in manure through soil at length scales ranging from 0.07 to 4.5 m. Effective removal efficiency depended on the distance of travel and followed a power law behavior. The power coefficient varied from a value of ~0.4 over short distances to >0.9 for lengths greater than 1 m (i.e. very little filtration for a finite fraction of biocolloids), consistent with there being reduced influence of soil solution and biocolloid properties for longer distances of travel. About a third of studies on microbial transport reviewed by Pang (2009) showed a similar effect of the distance of travel, mostly related to locations where the vadose zone was contaminated with organic matter. In the remaining studies, the removal of microbes was linear with travel distance. In addition to transport processes, the kinetics of bacterial population growth and decay must be considered in relation to the timing and numbers of organisms reaching a water resource.

7.2.3. Viruses

Movement to groundwater has been inferred from studies on wastewater application or has been investigated in model systems (Lance et al., 1976). Penetration was deeper in sandy soil than in loamy or clay soils, with movement to 17.4 m. Penetration of virus particles was greater under conditions of saturated flow than under unsaturated flow (Lance and Gerba, 1984; Lance et al., 1982). However, Lance et al. (1976) found that movement was independent of the actual infiltration rate.

7.2.4. Protozoa

The transport of protozoa has been studied in far less detail than has bacterial transport. Brush et al. (1999) carried out model studies of *C. parvum* oocysts movement through saturated columns of glass spheres, coarse sand, or shale aggregate. They observed that the oocysts, which were separated from dairy calves, did not adhere to any of these materials, and moved throughout the system of pores between particles in the case of sand and glass spheres. These oocysts moved preferentially in the larger pores between shale aggregates. Sand was more effective at removing oocysts than were the other particles, probably by filtration. The authors suggested that their results were indicative of significant transport being possible in both surface runoff and with infiltrating water.

7.2.5. Antibiotics and Other Pharmaceuticals

Tylosin tends to be strongly adsorbed on soil and is only slightly mobile (Rabølle and Spliid, 2000). Nevertheless, even though monensin is less readily adsorbed than tylosin, both leached from fall-applied manure but chlortetracycline did not (Dolliver and Gupta, 2008). Kay et al. (2005b,c) investigated the leaching of oxytetracycline, sulphachloropyridazine and tylosin through columns of soil in the laboratory. Although tylosin was redistributed throughout 0.3 m-long columns, it was not recovered in leachate, whether or not it was applied with organic amendments. Some oxytetracycline was found in the first leachate from short columns but not thereafter. Only 0.0002% of the total application of sulphachloropyridazine was detected in leachate with a maximum concentration 0.5 μg L^{-1}. At the end of 4 months, no antibiotic was found in the clay-loam soil, presumably degraded by microbes or other processes. Blackwell et al. (2007) investigated the leaching of oxytetratetracycline and sulphachloropyridazine in liquid manure (pig) through a sandy loam soil under field conditions. They concluded that oxytetratetracycline posed little risk to surface or groundwater, while sulphachloropyridazine posed a moderate risk to surface and groundwater. Gottschall et al. (2012) determined the presence of antibiotics, antimicrobial and other pharmaceuticals in tile drainage and shallow groundwater after the application of dewatered sewage biosolids. No tetracyclines or fluoroquinolones were detected in either tile drain or groundwater but triclocarban and triclosan were present in both, although neither persisted in groundwater. Gielen et al. (2009) studied the leaching of carbamazepine from sewage biosolids and found no movement through an allophane soil but almost no retardation in a sandy loam. Edwards et al. (2009) land-applied sewage biosolids, containing the antimicrobial pharmaceuticals triclosan and triclocarban, to an Orthic Humic Gleysol. Despite the concentration of triclosan in the amendment (~14,000 ng g^{-1} dw) being almost twice that of triclocarban (~8000 ng g^{-1} dw) the average concentration of the former in tile drainage water (43 ± 5 ng L^{-1}) was almost 59 times greater than that of triclocarban (7.3 ± 0.14 ng L^{-1}).

7.2.6. Endocrine-Disrupting Compounds

A study with pharmaceuticals associated with sewage biosolids confirmed the importance of macropores for their leaching through soil and the potential for discontinuous transport to impact their retention in soil (Larsbo et al., 2009). If a manure containing endocrine-disrupting compounds is left on the soil surface, it can move to surface water via tile drains (Kjær et al., 2007). The identification of Equol in the tile drainage water is a further evidence that these compounds can at least leach from soil and move onto surface water (Burnison et al., 2000). Given that these compounds do not persist in the soil (Lucas and Jones, 2006; Xuan et al., 2008), it seems likely that the movement though soil is mediated by preferential flow. Therefore, it is possible for these compounds to move to groundwater. Certainly 17β-estradiol has been detected in groundwater under mantled karst limestone, together with *E. coli* and other fecal coliform bacteria, suggesting that they originated from the same source (Peterson et al., 2000). Estrogens in liquid manure from pig were shown to move to a depth of 1 m in both a loam and a sandy soil, both of which showed evidence of preferential flow regimes (Lægdsmand et al., 2009). The concentration of estrogens in the leachate was sufficient to disrupt the endocrine system of some aquatic organisms. Lucas and Jones (2009) observed that in the presence of urine, 17β-estradiol and Estrone rapidly moved through a soil column, suggesting that it could move to much greater depths than the 7 cm of soil used in their experiment.

8. FARMING PRACTICES TO MINIMIZE RISKS TO HUMAN HEALTH FROM ORGANIC AMENDMENTS

A number of agronomic practices can help to reduce the risk to human health associated with the use of organic amendments. However, not all practices are applicable in every jurisdiction. Usually, the management practice will be chosen based on economic feasibility, which in turn may be determined by the availability of labor, equipment and climatic conditions. The guiding principles for fertilizer use, developed by the internationally based industry, focus on the application of the right material or product at the right rate, the right time and in the right location in the soil (Roberts, 2006). More recently, Bruulsema and Ketterings (2008) considered how this "4R nutrient stewardship concept" applied to the identification of best nutrient management practices for dairy farms in the northeastern USA. Within the concept of "Right Source," they included determination of the nutrient credits to be given for the manure used as well as for previous leguminous and perennial grass crops used for hay or pasture. Under "Right Rate" and "Right Time," there were no practices that fell outside the current North American guidelines for cash crop farming. However, for "Right Place," although calibration of fertilizer spreaders was included, these authors did not consider the importance of calibrating manure spreaders. Nonetheless, they indicated the benefits from injecting rather than broadcasting manure but stressed the need for rapid incorporation of volatile N sources. In developing the 4R concept, industry has stressed the need for stakeholder involvement, including consumers and the general public, in identifying the best management practices (Bruulsema et al., 2009). Inevitably, when people with diverse interests participate in such activities, decisions have to be made that involve evaluating the relative risks stemming from multiple threats. Some points of potential conflict are indicated in the following approaches that are aimed at mitigating the adverse impact of one or more components.

The greatest loss of NH_3 to the atmosphere occurs during storage or unconfined treatment, such as composting in open fields. Release of NH_3 and H_2S from stored liquid manure can be greatly reduced by the use of covers (e.g. Bicudo et al., 2004; Neeteson, 2000). Aeration and acidification are two techniques employed for managing volatile organic compounds (VOCs), NH_3 and greenhouse gas (GHG) emissions from animal waste composts and manure stored in lagoons (Amon et al., 2001; Demirer and Chen, 2004; Fitz et al., 2003; Parker, 2008; Shen et al., 2011; Smith et al., 2008). In dairy operations, manure can be handled as an effluent stream of liquid or slurry manure by means of a hydraulic flushing and lagoon storage followed by irrigation of wastewater. A large content of solids results in a rapid buildup of sludge in the lagoons, thereby reducing available storage volume. Lagoon acidification can be helpful in countering odors produced during decomposition of manure in storage, especially from NH_3 at pH values greater than 8.0, and in breaking up the sludge by enhancing microbial activity. Careful control is necessary to ensure that the pH of the lagoon does not fall below 6.2 as this favors H_2S emission. So, for a dairy farmer desirous of using the dairy effluent as a nutrient source for his silage crop, the dilemma is to balance the trades-off associated with optimum crop growth, salt build up in soils and air quality regulations.

Injection of liquid organic amendments reduces NH_3 loss during application to arable crops as will surface banding or shallow incorporation under grass. The rapid incorporation of broadcast or irrigated amendments has similar beneficial effects. A large loading of solids to soil, when wastewater from storage lagoons is applied in irrigated, can hinder the crop establishment and growth. Under climates with high levels of evaporation, the large amounts of nutrients applied in the wastewater can lead to the buildup of salts that can further impair crop production (Cabrera et al., 2009).

Another important issue is that N and P are usually present in a mixture of organic (often complex) and mineral forms in organic amendments. The organic forms need to be transformed into mineral forms before they are available to crops. Our ability to make field-specific accurate predictions of how much and when sufficient nutrient will become available is extremely limited. Furthermore, because proportion of N and P present is different to that required by plants, the correct rate of application has to be finely adjusted to avoid over supply of one or other nutrient and the potential consequences for human health. Voluntary codes of good agricultural practice have been developed aimed at limiting N availability to periods when the crop needs nutrients, preventing application of amendments near water courses if ground is steeply sloping, frozen or snow covered and developing crop rotations that restrict nutrient leaching during the drainage season. These practices together with a specific limit to the application of animal manure equivalent to 170 kg N-organic ha^{-1} y^{-1} have been made mandatory in some areas that are particularly vulnerable to NO_3^-–N-loss to water resources (European Union Nitrates Directive 91/676/EEC).

Organic amendments may also contain trace elements that are greatly in excess of crop needs. Many jurisdictions now legislate a limit to the amount that can be applied to prevent threats to human health (e.g. European Union Sewage Sludge Directive 86/278/EEC; EPA Part 503 Biosolids Rule).

Pathogens pose the greatest risk to human health from the use of organic amendments but few practices ensure that pathogens cannot contaminate food or water resources (Suslow et al., 2001). Within the European Union (EU), the guidelines developed for the United Kingdom (ADAS, 2001) prohibit the use of untreated sewage biosolids on vegetable or salad crops. These guidelines require that the application of conventionally treated biosolids to land for vegetable production is limited to crops, such as potatoes, that are cooked before consumption. Even then, an application cannot be later than 12 months before the crop is harvested. Following application of treated biosolids to land for salad crops, a period of 20 months must elapse before plants can be harvested.

The best approach is to eliminate pathogens before an amendment is land applied, but most treatments available for manure may result in some microbes surviving or are not cost-effective at the small-farm scale (Heinonen-Tanski et al., 2006). Soil, especially sandy soil, provides important but imperfect barriers to pathogen transport to water resources. Many of the practices that reduce nutrient loss after the application of amendments, such as rapid incorporation, may aid survival of bacteria and other pathogen groups. There can be an interaction between tillage and manure application with respect to the persistence of pathogens. Gagliardi and Karns (2000) investigated the fate of *E. coli* O157:H7 strain B6914 in soil. In disturbed soil, the organism replicated better in the absence of manure, whereas in intact soil cores (to simulate no-till conditions) replication and recovery of B6914 were much greater when manure was added.

Application of soil amendments should not precede rainfall events (Ramos and Martínez-Casasnovas, 2006; Ramos et al., 2006). Grass filter strips can be effective in removing both particulate P and bacteria from runoff over the soil surface (Coyne et al., 1998; Stutter et al., 2009). Controlling runoff and having a buffer zone between water bodies and the land receiving organic amendment offer other means of reducing risks of surface water contamination. Mineral fertilizer practices can also have an effect on pathogen movement under infiltration. Gagliardi and Karns (2000) found that *E. coli* O157:H7 strain B6914 levels were correlated with the content of nitrogen in leachate from manure-amended soil. They suggested that by limiting the amount of nitrogen applied to fields to the amount required by a crop in a season, the leaching if *E. coli* through the soil could be reduced.

Hammesfahr et al. (2011) concluded that conventional manure storage is unsuited to reduce risks from sulfonamide antibiotics in the soil environment. Given that estimates suggest that human intake of individual antibiotics from food and water represents the equivalent of 20% of the acceptable daily dose (Boxall, 2010), the greatest risk from the release of antibiotics into the environment is considered to be the increase in the number of pathogenic organisms that have multiple resistances to effective control drugs. While the EU banned the use of antibiotics in agriculture for growth promotion their use is widespread in other jurisdictions. Restricting their use for disease therapy in livestock and poultry would greatly reduce the total mass entering the environment.

Endocrine disruptors present in organic amendments appear most likely to reach humans through contaminated water resources that are used for potable supplies, recreation or fishing (commercial or private consumption). Although the levels of disruptors observed in streams receiving effluent from the sewage treatment plants can produce changes in growth and development of fish, a causal relationship has yet to be established for animals in the wild (Mills and Chichester, 2005). Similarly, there is evidence that endocrine disruptors from organic amendments can move from fields to water resources but no satisfactory reports have been made that link this to impacts on humans.

At present, the choices available to most producers to control the release of contaminants into the environment from the land application of organic amendments depend on the opportunities for employing preapplication treatments to minimize the loadings of pathogens, endocrine disrupting compounds and antibiotics, while maintaining those of mineral nutrients. The timing and quantities of amendments appropriate for applying to a given field must depend on the content of crop nutrients, the nature of the contaminants present, weather, the application methods involved, the soil and geological setting together with the location in the landscape.

8.1. Hazard Analysis Critical Control Point, Good Agricultural Practices and Quantitative Risk Assessment for Safe Food Production Using Organic Amendments

One method of assessing the risk of health effects from the use of organic amendments is to apply the Hazard Analysis Critical Control Point (HACCP) approach, which has been adopted by the food-handling industry worldwide (Hulebak and Schlosser, 2002). As a process based approach, HACCP focuses on the reduction of contamination by pathogenic microorganisms to a degree that greatly reduces the risk of foodborne illness, primarily by identifying control points where pathogens can be introduced, grown and be eliminated. Since its development in 1960, it has been adopted internationally to ensure the suitability and safety of various commodities intended for human consumption, such as meat (Antle, 2000), poultry (Roy et al., 2002), fruits and vegetables (Leifert et al., 2008; Suslow, 2003), and seafood (Alberini et al., 2008), all of which could be impacted by contaminants originating in organic amendments. Furthermore, HACCP has been endorsed and adopted by the: National Advisory Committee on Microbiological Criteria for Foods in the United States (1992); European Union for meat products and for foodstuffs (1992–93); and in 1993 by the food standards body of the United Nations, the Codex Alimentarius Commission (Codex) (Caswell and Hooker, 1996).

The principles of Good Agricultural Practices (GAPs) with respect to soil amendments are the proper management, including record keeping, of their storage, handling and application with careful management of soil and water, taking account of the geographical location and physical environment. They also stress the need for personal hygiene to prevent the direct transfer of contaminants to animals or from animals to crops or other humans (GAP, 2012). Variability in environment, agronomic and cultural practices associated with the application of organic amendments almost certainly prevents the identification of individual practices that have universal applicability. However, guiding principles for the prevention of contamination and cross-contamination together with the reduction in pathways of exposure can be compiled (e.g. UC-GAP, 2012).

Quantitative risk analysis (QRA) is a stepwise analysis of hazards that may be associated with a particular organism or food product and encompasses hazard identification, exposure assessment, dose response assessment, characterization of risk and risk management (Mayes, 1998). Pintar et al., (2012) concluded, from an analysis of the risk of infection by *Cryptosporidium* through consumption of municipally treated water, that QRA can be successfully applied at the community level to identify data gaps, rank relative public health risks, and forecast future risk scenarios.

Essentially, each of these approaches addresses different aspects of the management of risk in the production of food. All start with recognition of the specific hazards involved and an identification of potential pathways of human exposure. GAPs can be seen as providing basic agronomic and animal management advice to address concerns associated with primary production. HACCP establishes the locations where intervention is needed to block pathways of exposure, while QRA establishes the relative importance of different pathways and the effectiveness of possible interventions. Presently, there is a need to clearly establish all the pathways and the critical control points in the production of food that involves the use of organic amendments. Some of the issues around integrating the three approaches have been discussed by Mayes (1998), Notermans and Mead (1996) and Notermans et al. (1995).

Notermans et al. (1995) identified that the ability to set the limits for activities at a specified control point was the important aspect of HACCP. Although it was acknowledged that, as a result of establishing good management practices in food processing facilities, HACCP provided a mechanism for ensuring production of safe food, ideally it would desirable to incorporate QRA. Notermans and Mead (1996) argued that incorporating QRA would permit the establishment of a better estimate of the probability of illness resulting in the human population from the presence of a specific contaminant in an item of food. Mayes (1998) also considered the advantages of incorporating QRA in the HACCP system. Based on risks from microbial contamination, Mayes concluded that potential benefits included an enhanced scientific basis for the hazard analysis, establishment of clear relationships between the critical limits, public health impacts and microbiological criteria. Furthermore, it allowed greater transparency in the decision making process around food safety. The additional challenge, when primary production is included in such an assessment, comes from balancing the requirements for control points appropriate to the different classes of potential hazards as well as the individual species within those classes.

9. GENERAL CONCLUSIONS

The use of organic amendments not only makes good use of often-finite nutrient resources but can also offset changes in soil conditions that result from essential practices for crop production. Materials that have commonly been thought of as waste can provide macro- and micronutrients together with beneficial carbon compounds. The latter can restore soil physical and chemical properties. But the organic amendments in greatest abundance, animal manure and sewage biosolids, also contain components that provide a challenge to ensure that they do not pose a potential hazard to humans and other animals and plants in the wider environment. Of these contaminants, pathogenic bacteria, viruses and protozoan parasites pose the greatest immediate risk. Antibiotics and other pharmaceutical compounds may threaten humans if they increase the number of disease organisms that carry resistance genes or plasmids and cannot be controlled by known antimicrobial drugs. Natural hormones from domestic animals and humans together with manufactured compounds, which can mimic or interfere with their action, appear to threaten the reproductive health of other organisms in the land and water environments. While nutrients are critical for good crop production, they can also threaten human health, either directly or indirectly.

Another challenge is associated with the application of liquid organic amendments to land with tile drains. Bacteria and other contaminants move rapidly to the drains, after application. The likelihood of bacteria moving into water resources declines with time because of the die-off of organisms. This process takes longer if manure is applied in late fall, shortly before freeze-up. The shortest period of survival would be expected for bacteria in manure applied as a side dressing for maize (late spring). Later applications might further reduce the likelihood of bacterial contamination, but will increase the risk of nitrate contamination of groundwater because the crop has insufficient time to acquire the nutrient from the soil. It is coarser-textured soils that provide the most likely route for NO_3^-–N to leach to groundwater resources, whereas it is in well-structured loams and clayey soils that are more likely to have bacteria penetrate to depth.

The issue of pharmaceuticals, especially antibiotics and endocrine-disrupting compounds, poses new challenges. These compounds are present in small concentrations and the techniques required for their quantification are only now emerging. Their presence calls into question the suitability of the processes in use to treat the raw materials of organic amendments before they are applied on the land. It is in runoff water that the largest concentrations are likely to enter drinking water resources, although preferential flow does allow their movement into shallow groundwater.

There is insufficient information to generate an accurate risk assessment associated with the use of any individual organic amendment. For each source of an organic amendment, the exact composition is subject to uncertainty. This is true with respect to pathogen loading, particularly because so many organisms produce no symptoms within farm animals. Protecting drinking water resources presents challenges, since the greatest risk for NO_3^-–N to leach is associated with sandy soils but bacteria and pharmaceuticals move through soil in preferential flow paths, which are more prevalent in loams and finer-textured soil. For organic amendments derived from animal manure or sewage biosolids. the risks are mainly through food and water supplies.

ACKNOWLEDGMENTS

We are grateful to Klaas van der Hoek and Tom Bruulsema for their constructive comments and thank Adrian Unc for helpful discussions.

REFERENCES

Abbasi, M.K., Tahir, M.M., 2012. Economizing nitrogen fertilizer in wheat through combinations with organic manures in Kashmir, Pakistan. Agron. J. 104, 169–177.

Ackman, D., Marks, S., Mack, P., Caldwell, M., Root, T., Birkhead, G., 1997. Swimming-associated haemorrhagic colitis due to *Escherichia coli* O157:H7 infection: evidence of prolonged contamination of a fresh water lake. Epidemiol. Infect. 119, 1–8.

ADAS (2001). "Safe Sludge Matrix. third ed." Available at: http://www.adas.co.uk/Home/Publications/DocumentStore/tabid/211/Default.aspx (Cited 25th April 2011; verified 1st August, 2012).

Addis, P.B., Blaha, T., Crooker, B., Diez, F., Feirtag, J., Goyal, S., Greaves, I., Hathaway, M., Janni, K., Kirkhorn, S., Moon, R., Morse, D.E., Phillips, C., Reneau, J., Shutske, J., Wells, S., 1999. Generic Environmental Impact Statement on Animal Agriculture: A Summary of the Literature Related to the Effects of Animal Agriculture on Human Health. University of Minnesota, Minneapolis, Minnesota, USA.

Addiscott, T.M., Benjamin, N., 2004. Nitrate and human health. Soil Use Manage. 20, 98–104.

Adeli, A., Shankle, M.W., Tewolde, H., Brooks, J.P., Sistani, K.R., McLaughlin, M.R., Rowe, D.E., 2011. Effect of surface incorporation of broiler litter applied to no-till cotton on runoff quality. J. Environ. Qual. 40, 566–574.

Akhtar, S., Riemann, H.P., Thurmond, M.C., Farver, T.B., Franti, C.E., 1990. The use of log-linear model to evaluate factors associated with sero-prevelence of *Campylobacter fetus* in dairy cattle. Theriogenology 34, 989–1001.

Al-Rajab, A.J., Sabourin, L., Scott, A., Lapen, D.R., Topp, E., 2009. Impact of biosolids on the persistence and dissipation pathways of triclosan and triclocarban in an agricultural soil. Sci. Total Environ. 407, 5978–5985.

Alberini, A., Lichtenberg, E., Mancini, D., Galinato, G.I., 2008. Was it something I ate? Implementation of the FDA Seafood HACCP Program. Am. J. Agric. Econ. 90 (1), 28–41.

Allen, B.L., Mallarino, A.P., 2008. Effect of liquid swine manure rate, incorporation, and timing of rainfall on phosphorus loss with surface runoff. J. Environ. Qual. 37, 125–137.

Allison, R.V., 1944. The original field plot studies at Rothamsted. Soil Sci. Soc. Am. Proc. 8, 6–11.

Alter, T., Gaull, F., Kasimir, S., Gürtler, M., Mielke, H., Linnebur, M., Fehlhaber, K., 2005. Prevalences and transmission routes of *Campylobacter* spp. strains within multiple pig farms. Vet. Microbiol. 108, 251–261.

Altrock, A.V., Louis, A.L., Roesler, U., Alter, T., Beyerbach, M., Kreienbrock, L., Waldmann, K.H., 2007. Prevalence of *Campylobacter* spp. and *Yersinia enterocolitica* in fattening pig herds in Lower Saxony, Germany. In: "Proceedings of the 7th International Symposium on the Epidemiology and Control of Food-borne Pathogens in Pork", Verona, Italy, pp. 101–104.

Amon, B., Amon, T., Boxberger, J., Alt, C., 2001. Emissions of NH_3, N_2O and CH_4 from dairy cows housed in a farmyard manure tying stall (housing, manure storage, manure spreading). Nutr. Cycling Agroecosyst. 60, 103–113.

Andersen, H., Siegrist, H., Sørensen, B., Ternes, T., 2003. Fate of estrogens in a municipal sewage treatment plant. Environ. Sci. Technol. 37, 4021–4026.

Anderson, J., Horn, B.J., Gilpin, B.J., 2012. The prevalence and genetic diversity of *Campylobacter* spp. in domestic 'backyard' poultry in Canterbury, New Zealand. Zoonoses Public Health 59, 52–60.

Angers, D.A., Chantigny, M.H., MacDonald, J.D., Rochette, P., Coté, D., 2010. Differential retention of carbon, nitrogen and phosphorus in grassland soil profiles with long-term manure application. Nutr. Cycling Agroecosyst. 86, 225–229.

Antle, J.M., 2000. No such thing as a free safe lunch: the cost of food safety regulation in the meat industry. Am. J. Agric. Econ. 82 (2), 310–322.

Apslund, K., Tuoninen, V., Veijalainen, P., Hirn, J., 1990. The prevalence of *Yersinia enterocolitica* 0:3 in Finnish pigs and pork. Acta Vet. Scand. 31, 39–43.

Aramini, J.J., McLean, M., Wilson, J., Holt, J., Copes, R., Allen, B., Sears, W., 2000. Drinking water quality and health-care utilization for gastrointestinal illness in greater Vancouver. Can. Commun. Dis. Rep. 26, 211–214.

Arikan, O.A., Mulbry, W., Ingram, D., Millner, P., 2009a. Minimally managed composting of beef manure at the pilot scale: effect of manure pile construction on pile temperature profiles and on the fate of oxytetracycline and chlortetracycline. Biores. Technol. 100, 4447–4453.

Arikan, O.A., Mulbry, W., Rice, C., 2009b. Management of antibiotic residues from agricultural sources: use of composting to reduce chlortetracycline residues in beef manure from treated animals. J. Hazard. Mater. 164, 483–489.

Arogo, J., Westerman, P.W., Heber, A.J., Robarge, W.P., Classen, J.J., 2006. Ammonia emissions from animal feeding operations. In: Rice, J.M. (Ed.), Animal Agriculture and the Environment: National Center for Manure and Animal Waste Management White Papers, American Society of Agricultural and Biological Engineers (ASABE), St. Joseph, Michigan. USA, pp. 41–88.

Arrus, K.M., Holley, R.A., Ominski, K.H., Tenuta, M., Blank, G., 2006. Influence of temperature on Salmonella survival in hog manure slurry and seasonal temperature profiles in farm manure storage reservoirs. Livestock Sci. 102, 226–236.

Aslam, M., Stanford, K., McAllister, T.A., 2010. Characterization of antimicrobial resistance and seasonal prevalence of *Escherichia coli* O157:H7 recovered from commercial feedlots in Alberta, Canada. Lett. Appl. Microbiol. 50, 320–326.

Atabay, H.I., Corry, J.E.L., 1997. The prevalence of campylobacters and arcobacters in broiler chickens. J. Appl. Microbiol. 83, 619–626.
Atabay, H.I., Corry, J.E.L., 1998. The isolation and prevalence of campylobacters from dairy cattle using a variety of methods. J. Appl. Microbiol. 84, 733–740.
Aust, M.-O., Godlinski, F., Travis, G.R., Hao, X., McAllister, T.A., Leinweber, P., Thiele-Bruhn, S., 2008. Distribution of sulfamethazine, chlorotetracycline and tylosin in manure and soil of Canadian feedlots after subtherapeutic use in cattle. Environ. Pollut. 156, 1243–1251.
Avery, L.M., Hill, P., Killham, K., Jones, D.L., 2004. *Escherichia coli* O157 survival following the surface and sub-surface application of human pathogen contaminated organic waste to soil. Soil Biol. Biochem. 36, 2101–2103.
Bae, W., Kaya, K.N., Hancock, D.D., Call, D.R., Park, Y.H., Besser, T.E., 2005. Prevalence and antimicrobial resistance of thermophilic *Campylobacter* spp. from cattle farms in Washington State. Appl. Environ. Microbiol. 71, 169–174.
Ball Coelho, B.R., Roy, R.C., Topp, E., Lapen, D.R., 2007. Tile water quality following liquid swine manure application into standing corn. J. Environ. Qual. 36, 580–587.
Baker, A.A., Davis, E., Rehberger, T., Rosener, D., 2010. Prevalence and diversity of toxigenic *Clostridium perfringens* and *Clostridium difficile* among swine herds in the Midwest. Appl. Environ. Microbiol. 76, 2961–2967.
Bao, Y., Zhou, Q., Guan, L., Wang, Y., 2009. Depletion of chlortetracycline during composting of aged and spiked manures. Waste Manage. 29, 1416–1423.
Baptista, F.M., Alban, L., Nielsen, L.R., Domingos, I., Pomba, C., V. Almeida, V., 2010. Use of herd information for predicting *Salmonella* status in pig herds. Zoonoses Public Health 57 (Suppl. 1), 49–59.
Bardon, J., Kolar, M., Cekanova, L., Hejnar, P., Koukalova, D., 2008. Prevalence of *Campylobacter jejuni* and its resistance to antibiotics in poultry in the Czech Republic. Zoonoses Public Health 56, 111–116.
Barnett, G.M., 1994. Phosphorus forms in animal manure. Biores. Technol. 49, 139–147.
Barwick, R.S., Mohammed, H.O., White, M.E., Bryant, R.B., 2003. Prevalence of *Giardia* spp. and *Cryptosporidium* spp. on dairy farms in southeastern New York state. Prev. Vet. Med. 59, 1–11.
Beaton, J., 2009. History of fertilizer. Efficient Fertilizer Use Manual, Mosaic Back to Basics, Plymouth, Minnesota, USA Available at: http://www.back-to-basics.net/HistoryofFertilizers.pdf (Cited 25th Jan. 2010; verified 1st August 2012).
Beauchamp, E.G., 1983. Response of corn to nitrogen in preplant and sidedress applications of liquid dairy cattle manure. Can. J. Soil Sci. 63, 377–386.
Beauchamp, E.G., 1986. Availability of nitrogen from three manures to corn in the field. Can. J. Soil Sci. 66, 713–720.
Beauchamp, E.G., Paul, J.W., 1989. A simple model to predict manure nitrogen availability to crops. In: Hansen, J.A., Henrikson, K. (Eds.), Nitrogen in Organic Wastes Applied to Soils, Academic Press, London, UK, pp. 140–149.
Beauchamp, E.G., Kidd, G.E., Thurtell, G., 1982. Ammonia volatilization from liquid dairy cattle manure in the field. Can. J. Soil Sci. 62, 11–19.
Bech, T.B., Johnsen, K., Dalsgaard, A., Laegdsmand, M., Jacobsen, O.H., Jacobsen, C.S., 2010. Transport and distribution of *Salmonella enterica* serovar Typhimurium in loamy and sandy soil monoliths with applied liquid manure. Appl. Environ. Microbiol. 76, 710–714.
Beck-Friis, B., Pell, M., Sonesson, U., Jönsson, H., Kirchmann, H., 2000. Formation and emission of N_2O and CH_4 from compost heaps of organic household waste. Environ. Monit. Assess. 62, 317–331.
Benjamin, N., O'Driscoll, F., Dougall, H., Duncan, C., Smith, L., Golden, M., 1994. Stomach NO synthesis. Nature 268, 502.

Benke, M.B., Indraratne, S.P., Hao, X., Chang, C., Goh, T.B., 2008. Trace element changes in soil after long-term cattle manure applications. J. Environ. Qual. 37, 798–807.
Bergström, L.F., Kirchman, H., 1999. Leaching of total nitrogen from nitrogen-15-labeled poultry manure and inorganic nitrogen fertilizer. J. Environ. Qual. 28, 1283–1290.
Bergström, L.F., Kirchman, H., 2006. Leaching and crop uptake of nitrogen and phosphorus from pig slurry as affected by different application rates. J. Environ. Qual. 35, 1803–1811.
Bernal, M.P., Roig, A., Lax, A., Navarro, A.F., 1992. Effect of the application of pig slurry on some physico-chemical and physical properties of calcareous soils. Biores. Technol. 42, 233–239.
Berndtson, E., Emanuelsson, U., Engvall, A., Danielsson-Tham, M.L., 1996. A one-year epidemiological study of *Campylobacter* in 18 Swedish chicken farms. Prev. Vet. Med. 26, 167–185.
Berry, E.D., Wells, J.E., Archibeque, S.L., Ferrell, C.L., Freetly, H.C., Miller, D.N., 2006. Influence of genotype and diet on steer performance, manure odor, and carriage of pathogenic and other fecal bacteria. II. Pathogenic and other fecal bacteria. J. Anim. Sci. 84, 2523–2532.
Beusen, A.H.W., Bouwman, A.F., Heuberger, P.S.C., Van Drecht, G., Van Der Hoek, K.W., 2008. Bottom-up uncertainty estimates of global ammonia emissions from global agricultural production systems. Atmos. Environ. 42, 6067–6077.
Beutin, L., Geier, D., Steinruck, H., Zimmermann, S., Scheutz, F., 1993. Prevalence and some properties of verotoxin (shiga-like toxin)-producing *Escherichia coli* in seven different species of healthy domestic animals. J. Clin. Microbiol. 31, 2483–2488.
Bicudo, J.R., Goyal, S.M., 2003. Pathogens and manure management systems: a review. Environ. Technol. 24, 115–130.
Bicudo, J.R., Schmidt, D.R., Jacobson, L.D., 2004. Using Covers to Minimize Odor and Gas Emissions from Manure Storages. University of Kentucky – College of Agriculture Cooperative Extension Service, 5 pp. Available at: http://dept.ca.uky.edu/agc/pub_author.asp?author=Bicudo,+Jose (Cited 7th April, 2011; verified 1st August, 2012).
Bipfubusa, M., Angers, D.A., N'Dayegamiye, A., Antoun, H., 2008. Soil aggregation and biochemical properties following the application of fresh and composted organic amendments. Soil Sci. Soc. Am. J. 72, 160–166.
Bitton, G., Harvey, R.W., 1992. Transport of pathogens through soils and aquifers. In: Mitchell, R. (Ed.), Environmental Microbiology, Wiley-Liss, Inc., New York, NY, USA, pp. 103–124.
Blackwell, P.A., Kay, P., Boxall, A.B.A., 2007. The dissipation and transport of veterinary antibiotics in a sandy loam soil. Chemosphere 67, 292–299.
Blevins, D.W., Wilkison, D.H., Kelly, B.P., Silva, S.R., 1996. Movement of nitrate fertilizer to glacial till and runoff from a claypan soil. J. Environ. Qual. 25, 584–593.
Bolan, N.S., Naidu, R., Mahimairaja, S., Baskaran, S., 1994. Influence of low-molecular-weight organic acids on the solubilization of phosphates. Biol. Fertil. Soils 18, 311–319.
Bolan, N.S., Khan, M.A., Donaldson, J., Adriano, D.C., Matthew, C., 2003. Distribution and bioavailability of copper in farm effluent. Sci. Total Environ. 309, 225–236.
Bolan, N.S., Adriano, D.C., Mahimairaja, S., 2004. Distribution and bioavailability of trace elements in livestock and poultry manure by-products. Crit. Rev. Environ. Sci. Technol. 34, 291–338.
Bolton, D.J., Byrne, C.M., Sheridan, J.J., McDowell, D.A., Blair, I.S., 1999. The survival characteristics of a non-toxigenic strain of *Escherichia coli* 0157:H7. J. Appl. Microbiol. 86, 407–411.

Bolton, D.J., O'Neill, C.J., Fanning, S., 2012. A preliminary study of Salmonella, Verocytotoxigenic *Escherichia coli/Escherichia coli* O157 and *Campylobacter* on four mixed farms. Zoonoses Public Health 59, 217–228.

Bonanomi, G., Antignani, V., Capodilupo, M., Scala, F., 2010. Identifying the characteristics of organic soil amendments that suppress soilborne plant diseases. Soil Biol. Biochem. 42, 136–144.

Botzler, R.G., Cowan, A.B., Wetzler, T.F., 1974. Survival of Listeria moncytogenes in soil and water. J. Wildl. Dis. 10, 204–210.

Boutin, P., Torre, M., Serceau, R., Rideau, P.J., 1988. Atmospheric bacterial contamination from landspreading of animal wastes: evaluation of the respiratory risk for people nearby. J. Agric. Eng. Res. 39, 149–160.

Boxall, A.B.A., 2010. Veterinary medicines and the environment. In: Cunningham, F. (Ed.), Comparative and Veterinary Pharmacology, Handbook of Experimental Pharmacology, vol. 99. Springer-Verlag, Berlin, Heidelberg, Germany, pp. 291–314.

Boxall, A.B.A., Johnson, P., Smith, E.J., Sinclair, C.J., Stutt, E., Levy, L.S., 2006. Uptake of veterinary medicines from soils into plants. J. Agric. Food Chem. 54, 2288–2297.

Bradford, S., Segal, E., 2009. Fate of indicator microorganisms under nutrient management plan conditions. J. Environ. Qual. 38, 1728–1738.

Brandt, K.K., Holm, P.E., Nybroe, O., 2008. Evidence for bioavailable copper-dissolved organic matter complexes and transiently increased copper bioavailability in manure-amended soils as determined by bioluminescent bacterial biosensors. Environ. Sci. Technol. 42, 3102–3108.

Breland, T.A., 1995. Green manuring with clover and ryegrass catch crops undersown in spring wheat: effects on soil structure. Soil Use Manage. 11, 163–167.

Breland, T.A., 1996. Green manuring with clover and ryegrass catch crops undersown in small grains: effects on soil mineral nitrogen in field and laboratory experiments. Acta Agric. Scand. 46, 178–185.

Bresson, L.M., Koch, C., Le Bissonnais, Y., Barriuso, E., Lecomte, V., 2001. Soil surface structure stabilization by municipal waste compost application. Soil Sci. Soc. Am. J. 72, 160–166.

Brewer, R.A., Corbel, M.J., 1983. Characterization of *Yersinia enterocolitica* strains isolated from cattle, sheep and pigs in the United Kingdom. J. Hyg. (Lond.) 90, 425–433.

Brix, R., Postigo, C., González, S., Villagrasa, M., Navarro, A., Kuster, M., Lopez de Alda, M.J., Barceló, D., 2010. Analysis and occurrence of alkylphenolic compounds and estrogens in a European river basin and an evaluation of their importance as priority pollutants. Anal. Bioanal. Chem. 396, 1301–1309.

Brooks, J.P., Tanner, B.D., Josephson, K.L., Gerba, C.P., Haas, C.N., Pepper, I.L., 2005. A national study on the residential impact of biological aerosols from the land application of biosolids. J. Appl. Microbiol. 99, 310–322.

Brown, C., 2008. Available Nutrients and Value for Manure from Various Livestock Types. Agdex Factsheet # 538 The Queen's Printer, Toronto, Ontario, Canada 7 pp. Available at: http://www.omafra.gov.on.ca/english/crops/facts/08-041.pdf (Cited 20th January, 2010; verified 1st August 2012).

Brown, I.H., Alexander, D.J., 1998. Influenza. In: Soulsby, Lord, Simpson, D.I.H. (Eds.), Zoonoses. Biology, Clinical Practice, and Public Health Control, Oxford University Press, Oxford, UK, pp. 365–386.

Brunetti, G., Plaza, C., Senesi, N., 2005. Olive pomace amendment in Mediterranean conditions: effect on soil and humic acid properties and wheat (*Triticum turgidum* L.) yield. J. Agric. Food Chem. 53, 6730–6737.

Brush, C.F., Ghiorse, W.C., Annguish, L.J., Parlange, J.Y., Grimes, H.G., 1999. Transport of *Cryptosporidium parvum* oocysts through saturated columns. J. Environ. Qual. 28, 809–815.

Bruulsema, T.W., Ketterings, Q., 2008. Best Management Practices for Fertilizers on Northeastern Dairy Farms. Fertilizer BMPs Item 30-3220; Ref. 08052 International Plant Nutrition Institute, Norcross, Georgia, USA Available at: http://www.ipni.net/ipniweb/portal.nsf/0/9bbc50427c6469ae852574f200162796/$FILE/Dairy%20BMP.pdf (Cited 7th June, 2011; verified 1st August 2012).

Bruulsema, T., Lemunyon, J., Herz, B., 2009. Know your fertilizer rights. Crops and Soil, 13–18 March–April.

Bryan, N.S., van Grinsven, H.J.M., 2012. Adv. Agron.

Buck Louis, G.M., Sundaram, R., Schisterman, E.F., Sweeney, A.M., Lynch, C.D., Gore-Langton, R.E., Chen, Z., Kim, S., Caldwell, K.L., Boyd Barr, D., 2012. Heavy metals and couple fecundity, the LIFE Study. Chemosphere 87, 1201–1207.

Bulluck, L.R., Brosius, M., Evanylo, G.K., Ristaino, J.B., 2002. Organic and synthetic fertility amendments influence soil microbial, physical and chemical properties on organic and conventional farms. Appl. Soil Ecol. 19, 147–160.

Burgoa, B., Hubbard, R.K., Wauchope, R.D., Davis-Carter, J.G., 1993. Simultaneous measurement of runoff and leaching losses of bromide and phosphate using tilted beds and simulated rainfall. Commun. Soil Sci. Plant Anal. 24, 2689–2699.

Burkhardt, M., Stamm, C., Waul, C., Singer, H., Müller, S., 2005. Surface runoff and transport of sulfonamide antibiotics and tracers on manured grassland. J. Environ. Qual. 34, 1363–1371.

Burnison, B.K., Neheli, T., Nuttley, D., Hartmann, A., McInnis, R., Jurkovic, A., Terry, K., Ternes, T., Servos, M., 2000. Identification of Estrogenic Substances in Animal and Human Waste. 27th Annual Aquatic Toxicity Workshop: St. John's, Newfoundland, Oct. 1–4, 2000.

Busheé, E.L., Edwards, D.R., Moore, P.A., 1998. Quality of runoff from plots treated with municipal sludge and horse bedding. Trans. ASAE 41, 1035–1041.

Cabrera, V.E., Stavast, L.J., Baker, T.T., Wood, M.K., Cram, D.S., Flynn, R.P., Ulery, A.L., 2009. Soil and runoff response to dairy manure application on New Mexico rangeland. Agric. Ecosyst. Environ. 131, 255–262.

Callaway, T.R., Keen, J.E., Edrington, T.S., Baumgard, L.H., Spicer, L., Fonda, E.S., Griswold, K.E., Overton, T.R., VanAmburgh, M.E., Anderson, R.C., Genovese, K.J., Poole, T.L., Harvey, R.B., Nisbet, D.J., 2005. Fecal prevalence and diversity of Salmonella Species in lactating dairy cattle in four States. J. Dairy Sci. 88, 3603–3608.

Calvert, C.C., Smith, L.W., Wrenn, T.R., 1978. Hormonal activity of poultry excreta processed for livestock feed. Poult. Sci. 57, 265–270.

Camargo, J.A., Alonso, Á., 2006. Ecological and toxicological effects of inorganic nitrogen pollution in aquatic ecosystems: a global assessment. Environ. Int. 32, 831–849.

Campbell, L.B., Racz, G.J., 1975. Organic and inorganic P content, movement and mineralization of P in soil beneath a feedlot. Can. J. Soil Sci. 55, 457–466.

Carmichael, W.W., Jones, C.L.A., Mahmood, N.A., Theiss, W.C., 1985. Algal toxins and water-based diseases. Crit. Rev. Environ. Control 15, 275–313.

Castellanos, T.Z., Pratt, P.F., 1981. Mineralization of manure nitrogen-correlation with laboratory indexes. Soil Sci. Soc. Am. J. 45, 354–357.

Castro-Hermida, J.A., González-Losada, Y.A., Mezo-Menéndez, M., Ares-Mazás, E., 2002. A study of cryptosporidiosis in a cohort of neonatal calves. Vet. Parasitol. 106, 11–17.

Caswell, J.A., Hooker, N.H., 1996. HACCP as an international trade standard. Am. J. Agric. Econ. 78 (3), 775–779.

Causapé, A.C., Quílez, J., Sánchez-Acedo, C., del Cacho, E., López-Bernad, F., 2002. Prevalence and analysis of potential risk factors for *Cryptosporidium parvum* infection in lambs in Zaragoza (northeastern Spain). Vet. Parasitol. 104, 287–298.

Cela, S., Berenguer, P., Santiveri, F., Lloveras, J., 2010. Potential phosphorus, potassium, and magnesium surpluses in an irrigated maize monoculture fertilized with pig slurry. Agron. J. 102, 96–102.

Cessna, J., Larney, F.J., Kuchta, S.L., Hao, X., Entz, T., Topp, E., McAllister, T.A., 2011. Veterinary antimicrobials in feedlot manure: dissipation during composting and effects on composting processes. J. Environ. Qual. 40, 188–198.
Chambers, P., Dekimpe, C., Foster, N., Goss, M., Miller, J., Prepas, E., 2001. Agricultural and forestry land use impacts. (Chapter 13) Threats to Sources of Drinking Water and Aquatic Ecosystem Health in Canada, Environment Canada, National Water Research Institute, Burlington, Ontario, Canada, pp. 57–62.
Chaney, R.L., 2012. Food safety issues for mineral and organic fertilizers. Adv. Agron.
Chang, C., Sommerfeldt, T.G., Entz, T., 1991. Soil chemistry after eleven annual applications of cattle feedlot manure. J. Environ. Qual. 20, 475–480.
Chantigny, M.H., Angers, D.A., Morvan, T., Pomar, C., 2004a. Dynamics of pig slurry nitrogen in soil and plant as determined with ^{15}N. Soil Sci. Soc. Am. J. 68, 637–643.
Chantigny, M.H., Rochette, P., Angers, D.A., Massé, D., Côte, D., 2004b. Ammonia volatilization and selected soil characteristics following application of anaerobically digested pig slurry. Soil Sci. Soc. Am. J. 68, 306–312.
Chapin, A., Rule, A., Gibson, K., Buckley, T., Schwab, K., 2005. Airborne multidrug-resistant bacteria isolated from a concentrated swine feeding operation. Environ. Health Perspect. 113, 137–142.
Chapman, P.A., Siddons, C.A., Cerdan Malo, A.T., Harkin, M.A., 1997. A 1-year study of *Escherichia coli* O157 in cattle, sheep, pigs and poultry. Epidemiol. Infect. 119, 245–250.
Chee-Sandford, J.C., Mackie, R.I., Koike, S., Krapac, I.G., Lin, Y.F., Yannarell, A.C., Maxwell, S., Aminov, R.I., 2009. Fate and transport of antibiotic residues and antibiotic resistance genes following land application of manure waste. J. Environ. Qual. 38, 1086–1108.
Christian, T., Schneider, R.J., Farber, H.A., Skutlarek, D., Meyer, M.T., Goldbach, H.E., 2003. Determination of antibiotic residues in manure, soil and surface waters. Acta Hydrochim. Hydrobiol. 31, 36–44.
Cisneros, R., Bytnerowicz, A., Schweizer, D., Zhong, S., Traina, S., Bennett, D.H., 2010. Ozone, nitric acid, and ammonia air pollution is unhealthy for people and ecosystems in southern Sierra Nevada, California. Environ. Pollut. 158, 3261–3271.
Clinton, N.A., Weaver, R.W., Zibilske, L.M., Hidalgo, R.J., 1979. Incidence of salmonellae in feedlot manure. J. Environ. Qual. 8, 480–481.
Coklin, T., Farber, J., Parrington, L., Dixon, B., 2007. Prevalence and molecular characterization of *Giardia duodenalis* and *Cryptosporidium* spp. in dairy cattle in Ontario, Canada. Vet. Parasitol. 150, 297–305.
Cools, D., Merckx, R., Vlassak, K., Verhaegen, J., 2001. Survival of *E. coli* and *Enterococcus* spp. derived from pig slurry in soils of different texture. Appl. Soil Ecol. 17, 53–62.
Cosby, K., Partovi, K.S., Crawford, J.H., Patel, R.K., Reiter, C.D., Martyr, S., Yang, B.K., Waclawiw, M.A., Zalos, G., Xu, X., Huang, K.T., Shields, H., Kim-Shapiro, D.B., Schechter, A.N., Cannon, R.O., Gladwin, M.T., 2003. Nitrite reduction to nitric oxide by deoxyhemoglobin vasodilates the human circulation. Nat. Med. 9, 1498–1505.
Côté, C., Villeneuve, A., Lessard, L., Quessy, S., 2006. Fate of pathogenic and nonpathogenic microorganisms during storage of liquid hog manure in Québec. Livestock Sci. 102, 204–210.
Coyne, M.S., Gilfillen, R.A., Villalba, A., Zhang, Z., Rhodes, R., Dunn, L., Blevins, R.L., 1998. Fecal bacteria trapping by grass filter strips during simulated rain. J. Soil Water Conserv. 53, 140–145.
Crane, S.R., Moore, J.A., Grismer, M.E., Miner, J.R., 1983. Bacterial pollution from agricultural sources: a review. Trans. ASAE 26, 858–872.

Cummings, K.J., Warnick, L.D., Alexander, K.A., Cripps, C.J., Gröhn, Y.T., McDonough, P.L., Nydam, D.V., Reed, K.E., 2009. The incidence of salmonellosis among dairy herds in the northeastern United States. J. Dairy Sci. 88, 3766–3774.
Davies, R.H., Heath, P.J., Coxon, S.M., Sayers, A.R., 2003. Evaluation of the use of pooled serum, pooled muscle tissue fluid (meat juice) and pooled faeces for monitoring pig herds for Salmonella. J. Appl. Microbiol. 95, 1016–1025.
Davis, J.G., Truman, C.C., Kim, S.C., Ascough, J.C., Carlson, K., 2006. Antibiotic transport via runoff and soil loss. J. Environ. Qual. 35, 2250–2260.
De Liguoro, M., Cibin, V., Capolongo, F., Halling-Sørensen, B., Montesissa, C., 2003. Use of oxytetracycline and tylosin in intensive calf farming: evaluation of transfer to manure and soil. Chemosphere 52, 203–212.
Del Castilho, P., Chardon, W.J., Salomons, W., 1993a. Influence of cattle-manure slurry application on the solubility of cadmium, copper and zinc in a manured acidic, loamy-sand soil. J. Environ. Qual. 22, 686–697.
Del Castilho, P., Dalenberg, J.W., Brunt, K., Bruins, A.P., 1993b. Dissolved organic matter, cadmium, copper and zinc in pig slurry- and soil solution-size exclusion chromatography fractions. Int. J. Environ. Anal. Chem. 50, 91–107.
Demirer, G., Chen, S., 2004. Effect of retention time and organic loading rate on anaerobic acidification and biogasification of dairy manure. J. Chem. Technol. Biotechnol. 79, 1381–1387.
Denis, M., Henrique, E., Chidaine, B., Tircot, A., Bougeard, S., Fravalo, P., 2011. *Campylobacter* from sows in farrow-to-finish pig farms: risk indicators and genetic diversity. Vet. Microbiol. 154, 163–170.
De Vries, W., Römkens, P.F.A.M., Schütze, G., 2007. Critical soil concentrations of cadmium, lead, and mercury in view of health effects on humans and animals. Rev. Environ. Contam. Toxicol. 191, 91–130.
Dickinson, A.G., Stamp, J.T., Renwick, C.C., 1974. Maternal and lateral transmission of scrapie in sheep. J. Comp. Pathol. 84, 19–25.
Dils, R.M., Heathwaite, A.L., 1997. Phosphorus fractionation in grassland hill-slope hydrological pathways. In: Tunney, H. (Ed.), Phosphorus Loss from Soil to Water, CAB International, Wallingford, UK, pp. 349–351.
Dolliver, H., Gupta, S., 2008. Antibiotic losses in leaching and surface runoff from manure-amended agricultural land. J. Environ. Qual. 37, 1227–1237.
Dolliver, H., Gupta, S., Noll, S., 2008. Antibiotic degradation during manure composting. J. Environ. Qual. 37, 1245–1253.
Donaldson, K., Seaton, A., 2012. A short history of the toxicology of inhaled particles. Available at: Part. Fibre Toxicol. 9, 13. http://www.particleandfibretoxicology.com/content/9/1/13.
Donoho, A.J., 1984. Biochemical studies on the fate of monensin in animal and in the environment. J. Anim. Sci. 58, 1528–1539.
Douesnard-Malo, F., Daigle, F., 2011. Increased persistence of *Salmonella enterica* serovar Typhi in the presence of *Acanthamoeba castellanii*. Appl. Environ. Microbiol. 77, 7640–7646.
Douglas, J.T., Aitken, M.N., Smith, C.A., 2003. Effects of five non-agricultural organic wastes on soil composition, and on the yield and nitrogen recovery of Italian ryegrass. Soil Use Manage. 19, 135–138.
Duncan, C., Dougall, H., Johnston, P., Green, S., Brogan, R., Leifert, C., Smith, L., Golden, M., Benjamin, N., 1995. Chemical generation of nitric oxide in the mouth from the enterosalivary circulation of dietary nitrate. Nat. Med. 1, 546–551.
Dungan, R.S., 2010. Board-invited review: fate and transport of bioaerosols associated with livestock operations and manures. J. Anim. Sci. 88, 3693–3706.

Duranti, A., Cacciò, S.M., Pozio, E., Di Egidio, A., De Curtis, M., Battisti, A., Scaramozzino, P., 2009. Risk factors associated with *Cryptosporidium parvum* infection in cattle. Zoonoses Public Health 56, 176–182.

Dutta, S., Inamdar, S., Tso, J., Aga, D.S., Sims, T., 2010. Free and conjugated estrogen exports in surface-runoff from poultry litter-amended soil. J. Environ. Qual. 39, 1688–1698.

Dyke, G.V., 1993. John Lawes of Rothamsted. Pioneer of Science, Farming and Industry. Hoos Press, Harpenden, UK.

Edmeades, D.C., 2003. The long-term effects of manures and fertilisers on soil productivity and quality: a review. Nutr. Cycling Agroecosyst. 66, 165–180.

Edrington, T.S., Hume, M.E., Looper, M.L., Schultz, C.L., Fitzgerald, A.C., Callaway, T.R., Genovese, K.J., Bischoff, K.M., McReynolds, J.L., Anderson, R.C., Nisbet, D.J., 2004. Variation in the faecal shedding of *Salmonella* and *E. coli* O157:H7 in lactating dairy cattle and examination of Salmonella genotypes using pulsed-field gel electrophoresis. Lett. Appl. Microbiol. 38, 366–372.

Edwards, D.R., Daniel, T.C., 1993. Effects of poultry litter application rate and rainfall intensity on quality of runoff from fescue grass plots. J. Environ. Qual. 22, 361–365.

Edwards, M., Topp, E., Metcalf, C.D., Li, H., Gottschall, N., Bolton, P., Curnoe, W., Payne, M., Beck, A., Kleywegt, S., Lapen, D.R., 2009. Pharmaceutical and personal care products in tile drainage following surface application and injection of dewatered municipal biosolids to an agricultural field. Sci. Total Environ. 407, 4220–4230.

Eghball, B., Binford, G.D., Baltensperger, D.D., 1996. Phosphorus movement and adsorption in a soil receiving long-term manure and fertilizer application. J. Environ. Qual. 25, 1339–1343.

El-Shibiny, A., Connerton, P.L., Connerton, I.F., 2005. Enumeration and diversity of campylobacters and bacteriophages isolated during the rearing cycles of free-range and organic chickens. Appl. Environ. Microbiol. 71, 1259–1266.

Elder, R.O., Keen, J.E., Siragusa, G.R., Barkocy-Gallagher, G.A., Koohmaraie, M., Laegreid, W.W., 2000. Correlation of enterohemorrhagic *Escherichia coli* O157 prevalence in feces, hides, and carcasses of beef cattle during processing. Proceedings of the National Academy of Sciences of the United States of America 97, 2999–3003.

Ellerbroek, L.I., Lienau, J.-A., Klein, G., 2010. *Campylobacter* spp. in broiler flocks at farm level and the potential for cross-contamination during slaughter. Zoonoses Public Health 57, e81–e88.

Ellis-Iversen, J., Geoff C. Pritchard, G.C., Wooldridge, M., Nielen, M., 2009. Risk factors for *Campylobacter jejuni* and *Campylobacter coli* in young cattle on English and Welsh farms. Prev. Vet. Med. 88, 42–48.

Enemark, H.L., Ahrens, P., Lowery, C.J., Thamsborg, S.M., Enemark, J.M.D., Bille-Hansen, V., Lind, P., 2002. *Cryptosporidium andersoni* from a Danish cattle herd: identification and preliminary characterisation. Vet. Parasitol. 107, 37–49.

Epstein, E., 2003. Land Application of Sewage Sludge and Biosolids. Lewis Publishers, CRC Press, Boca Raton, Florida, USA.

Erickson, M.C., Webb, C.C., Diaz-Perez, J.C., Phatak, S.C., Silvoy, J.J., Davey, L., Payton, A.S., Liao, J., Ma, L., Doyle, M.P., 2010. Infrequent internalization of *Escherichia coli* O157:H7 into field-grown leafy greens. J. Food Prot. 73, 500–506.

van Es, H.M., Sogbedji, J.M., Schindelbeck, R.R., 2006. Effect of manure application timing, crop, and soil type on nitrate leaching. J. Environ. Qual. 35, 670–679.

Esteban, J.I., Oporto, B., Aduriz, G., Juste, R.A., Hurtado, A., 2009. Faecal shedding and strain diversity of Listeria monocytogenes in healthy ruminants and swine in Northern Spain. BMC Vet. Res. 5, 2.

Estes, M.K., Cohen, J., 1989. Rotavirus gene structure and function. Microbiol. Rev. 53, 165–169.

Falconer, I.R., 1991. Tumor promotion and liver injury caused by oral consumption of cyanobacteria. Environ. Toxicol. Water Qual. 6, 177–184.

Falconer, I.R., 2001. Toxic cyanobacterial bloom problems in Australian waters: risks and impacts on human health. Phycologia 40 (3), 228–233.

Falconer, I.R., Humpage, A.R., 2001. Preliminary evidence for in vivo tumour initiation by oral administration of extracts of the blue-green alga *Cylindrospermopsis raciborskii* containing the toxin cylindrospermopsin. Environ. Toxicol. 16, 192–195.

Falconer, I.R., Hardy, S.J., Humpage, A.R., Froscio, S.M., Tozer, G.J., Hawkins, P.R., 1999. Hepatic and renal toxicity of the blue–green alga (cyanobacterium) *Cylindrospermopsis raciborskii* in male Swiss albino mice. Environ. Toxicol. 14, 143–150.

Farzan, A., Friendship, R.M., Dewey, C.E., Warriner, K., Poppe, C., Klotins, K., 2006. Prevalence of Salmonella spp. on Canadian pig farms using liquid or dry-feeding. Prev. Vet. Med. 73, 241–254.

Farzan, A., Friendship, R.M., Cook, A., Pollari, F., 2010. Occurrence of *Salmonella*, *Campylobacter*, *Yersinia enterocolitica*, *Escherichia coli* O157 and Listeria monocytogenes in swine. Zoonoses Public Health 57, 388–396.

Feder, I., Wallace, F.M., Gray, J.T., Fratamico, P., Fedorka-Cray, P.J., Pearce, R.A., Call, J.E., Perrine, R., Luchansky, J.B., 2003. Isolation of *Escherichia coli* O157:H7 from intact colon fecal samples of swine. Emerg. Infect. Dis. 9, 380–383.

Fegan, N., Higgs, G., Vanderlinde, P., Desmarchelier, P., 2003. Enumeration of *Escherichia coli* O157:H7 in cattle faeces using most probable number technique and automated immunomagnetic separation. Lett. Appl. Microbiol. 38, 56–59.

Fink, R.C., Black, E.P., Hou, Z., Sugawara, M., Sadowsky, M.J., Diez-Gonzalez, F., 2012. Tanscriptional responses of *Escherichia coli* K-12 and O157:H7 associated with lettuce leaves. Appl. Environ. Microbiol. 78, 1752–1764. 10.1128/AEM.07454-11.

Finlay-Moore, O., Hartel, P.G., Cabrera, M.L., 2000. 17β-Estradiol and testosterone in soil and runoff from grasslands amended with broiler litter. J. Environ. Qual. 29, 1604–1611.

Finlayson-Pitts, B.J., Ezell, M.J., Pitts, J.N., 1989. Formation of chemically active chlorine compounds by reactions of atmospheric NaCl particles with gaseous N_2O_5 and $ClONO_2$. Nature 337, 241–244.

Fitz, D.R., Pisano, J.T., Malkina, I.L., Goorahoo, D., Krauter, C.F., 2003. A passive flux denuder for evaluating emissions of ammonia at a dairy Farm. J. Air Waste Manage. Assoc. 53, 937–945.

Fleming, R., Fraser, H., 2000. Impacts of Winter Spreading of Manure on Water Quality – Literature Review. Ridgetown College, University of Guelph, Ridgetown, Ontario, Canada Available at: http://www.ridgetownc.uoguelph.ca/research/research_reports_topic.cfm?ref=MANURE (Cited 30th March, 2010; verified 1st August 2012).

Flowers, T.H., Arnold, P.W., 1983. Immobilization and mineralization of nitrogen in soils incubated with pig slurry or ammonium sulphate. Soil Biol. Biochem. 15, 329–335.

Foley, B.J., Cooperband, L.R., 2002. Paper mill residuals and compost effects on soil carbon and physical properties. J. Environ. Qual. 31, 2086–2095.

Foran, M.E., Dean, D.M., Taylor, H.E., 1993. The land application of liquid manure and its effect on tile drain water and groundwater quality. Agricultural Research to Protect Water Quality: Proceedings of the Conference. February 21–24. Minneapolis, MN, Soil and Water Conservation Society, Ankeny, IA, USA, pp. 279–281.

Forman, D., Al-Dabbagh, S., Doll, R., 1985. Nitrates, nitrites and gastric cancer in Great Britain. Nature 313, 620–625.

Fosse, J., Seegers, H., Magras, C., 2008. Prevalence and risk factors for bacterial food-borne zoonotic hazards in slaughter pigs: a review. Zoonoses Public Health 56, 429–454.

Franco, A., Lovari, S., Cordaro, G., Di Matteo, P., Sorbara, L., Iurescia, M., Donati, V., Buccella, C., Battisti, A., 2009. Prevalence and concentration of verotoxigenic *Escherichia coli* O157:H7 in adult sheep at slaughter from Italy. Zoonoses Public Health 56, 215–220.

Franz, E., Klerks, M.M., De Vos, O.J., Termorshuizen, A.J., van Bruggen, H.C., 2007. Prevalence of shiga toxin-producing *Escherichia coli* stx1, stx2, eaeA, and rfbE genes and survival of *E. coli* O157:H7 in manure from organic and low-input conventional dairy farms. Appl. Environ. Microbiol. 73, 2180–2190.

Franz, E., Semenov, A.V., Termorshuizen, A.J., de Vos, O.J., Bokhorst, J.G., van Bruggen, A.H.C., 2008. Manure-amended soil characteristics affecting the survival of *E. coli* O157:H7 in 36 Dutch soils. Environ. Microbiol. 10, 313–327.
Franz, E., van Hoek, A.H.A.M., Bouw, E., Aarts, H.J.M., 2011. Variability of Escherichia coliO157 strain survival in manure-amended soil in relation to strain origin, virulence profile and carbon nutrition profile. Appl. Environ. Microbiol. 77, 8088–8096.
Fukushima, H., Nakamura, R., Ito, Y., Saito, K., 1983. Ecological studies of *Yersinia enterocolitica* I. Dissemination of *Y. enterocolitica* in pigs. Vet. Microbiol. 8, 469–483.
Furuichi, T., Kannan, K., Suzuki, K., Tanaka, S., Giesy, J.P., Masunaga, S., 2006. Occurrence of estrogenic compounds in and removal by a swine farm waste treatment plant. Environ. Sci. Technol. 40, 7896–7902.
Gadd, J.B., Tremblay, L.A., Northcott, G.L., 2010. Steroid estrogens, conjugated estrogens and estrogenic activity in farm dairy shed effluents. Environ. Pollut. 158, 730–736.
Gagliardi, J.V., Karns, J.S., 2000. Leaching of *Escherichia coli* O157:H7 in diverse soils under various agricultural management practices. Appl. Environ. Microbiol. 66, 877–883.
Gale, P., 2007. The prion/lipid hypothesis – further evidence to support the molecular basis for transmissible spongiform encephalopathy risk assessment. J. Appl. Microbiol. 103, 2033–2045.
Gale, P., Stanfield, G., 2001. Towards a quantitative risk assessment for BSE in sewage sludge. J. Appl. Microbiol. 91, 563–569.
Gale, P., Young, C., Stanfield, G., Oaks, D., 1998. Development of a risk assessment for BSE in the aquatic environment. J. Appl. Microbiol. 84, 467–477.
Gannon, J.T., Manilal, V.B., Alexander, M., 1991. Relationship between cell surface properties and transport of bacteria through soil. Appl. Environ. Microbiol. 57, 190–193.
Gannon, V.P.J., Graham, T.A., King, R., Michel, P., Read, S., Ziebell, K., Johnson, R.P., 2002. *Escherichia coli* O157:H7 infection in cows and calves in a beef cattle herd in Alberta, Canada. Epidemiol. Infect. 129, 163–172.
Garbeva, P., van Veen, J.A., van Elsas, J.D., 2004. Microbial diversity in soil: selection of microbial populations by plant and soil type and implications for disease suppressiveness. Annu. Rev. Phytopathol. 42, 243–270.
García, R., Bælum, J., Fredslund, L., Santorum, P., Jacobsen, C.S., 2010. Influence of temperature and predation on survival of *Salmonella enterica* serovar Typhimurium and expression of invA in soil and manure-amended soil. Appl. Environ. Microbiol. 76, 5025–5031.
García-Feliz, C., Collazos, J.A., Carvajal, A., Vidal, A.B., Aladueña, A., Ramiro, R., de la Fuente, M., Echeita, M.A., Rubio, P., 2007. *Salmonella enterica* infections in Spanish swine fattening units. Zoonoses Public Health 54, 294–300.
Garfield, L.M., Walker, M.J., 2008. Water potential changes in faecal matter and *Escherichia coli* survival. J. Appl. Microbiol. 105, 1009–1016.
Gaynor, J.D., Findlay, W.I., 1995. Soil phosphorus loss from conservation and conventional tillage in corn production. J. Environ. Qual. 24, 734–741.
Gebreyes, W.A., Thakur, S., Morrow, W.E.M., 2005. *Campylobacter coli*: prevalence and antimicrobial resistance in antimicrobial-free (ABF) swine production systems. J. Antimicrob. Chemother. 56, 765–768.
Gerba, C.P., Bitton, G., 1984. Microbial pollutants: their survival and transport pattern to groundwater. In: Gerba, C.P., Bitton, G. (Eds.), Groundwater Pollution Microbiology, John Willey & Sons, New York, pp. 66–88.
Gerba, C.P., Smith, J.E., 2005. Sources of pathogenic microorganisms and their fate during land application of wastes. J. Environ. Qual. 34, 42–48.
Gerba, C.P., Wallis, C., Melnick, J.L., 1975. Fate of wastewater bacteria and viruses in soil. J. Irrig. Drain. Eng. 54, 157–174.
Gielen, G.J.H.P., van den Heuvel, M.R., Clinton, P.W., Greenfield, L.G., 2009. Factors impacting on pharmaceutical leaching following sewage application to land. Chemosphere 74, 537–542.

González, M.D., Moreno, E., Quevedo-Sarmiento, J., Ramos-Cormenzana, A., 1990. Studies on antibacterial activity of waste waters from olive oil mills (alpechin): Inhibitory activity of phenolic and fatty acids. Chemosphere 20, 423–432.

Gorbach, S.L., 2001. Antimicrobial use in animal feed – time to stop. N. Engl. J. Med. 345, 1202–1203.

Goss, M.J., Kay, B.D., 2005. Soil aggregation. In: Wright, S.F. (Ed.), Roots and Soil Management: Interactions between Root and the Soil, In: Zobel, R.W. (Ed.), Agronomy Monograph No 48, ASA, CSSA, and SSSA, Madison, WI. USA, pp. 163–180.

Goss, M.J., Richards, C., 2008. Development of a risk-based index for source water protection planning, which supports the reduction of pathogens from agricultural activity entering water resources. J. Environ. Manage. 87, 623–632.

Goss, M.J., Harris, G.L., Howse, K.R., 1983. Functioning of mole drains in a clay soil. Agric. Water Manage. 6, 27–30.

Goss, M.J., Curnoe, W.E., Beauchamp, E.G., Smith, P.S., Nunn, B.D.C., Barry, D.A.J., 1995. An Investigation into the Management of Manure-nitrogen to Safeguard the Quality of Groundwater. COESA Report No.: LMAP – 013/95. Research Branch Agriculture and Agri-Food Canada, Pest Management Research Centre (London), London, Ontario, Canada.

Goss, M.J., Barry, D.A.J., Rudolph, D.L., 1998. Contamination in Ontario farmstead domestic wells and its association with agriculture: 1. Results from drinking water wells. J. Contam. Hydrol. 32, 267–293.

Goss, M.J., Ehlers, W., Unc, A., 2010. The role of lysimeters in the development of our understanding of contamination of groundwater aquifers. Phys. Chem. Earth 35, 913–926.

Gottschall, N., Topp, E., Metcalfe, C., Payne, M., Kleywegt, S., Russell, P., Lapen, D.R., 2012. Pharmaceutical and personal care products in groundwater, subsurface drainage, soil, and wheat grain, following a high single application of municipal biosolids to a field. Chemosphere 87, 194–203.

Grinberg, A., Pomroy, W.E., Weston, J.F., Ayanegui-Alcerreca, A., Knight, D., 2005. The occurrence of *Cryptosporidium parvum*, *Campylobacter* and *Salmonella* in newborn dairy calves in the Manawatu region of New Zealand. N. Z. Vet. J. 53, 315–320.

Grove-White, D.H., Leatherbarrow, A.J.H., Cripps, P.J., Diggle, P.J., French, N.P., 2010. Temporal and farm-management-associated variation in the faecal-pat prevalence of *Campylobacter jejuni* in ruminants. Epidemiol. Infect. 138, 549–558.

Guber, A.K., Pachepsky, Y.A., Shelton, D.R., Yu, O., 2007. Effect of bovine manure on fecal coliform attachment to soil and soil particles of different sizes. Appl. Environ. Microbiol. 73, 3363–3370.

Guévremont, E., Higgins, R., Quessy, S., 2004. Characterization of *Campylobacter* isolates recovered from clinically healthy pigs and from sporadic cases of campylobacteriosis in humans. J. Food Prot. 67, 228–234.

Gupta, U.C., Gupta, S.C., 1998. Trace element toxicity relationships to crop production and livestock and human health: implications for management. Commun. Soil Sci. Plant Anal. 29, 1491–1522.

Gupta, U.C., Kening, W.U., Siyuan, L., 2008. Macronutrients in soils, crops, and livestock. Earth Sci. Front. 15, 110–125.

Gürtler, M., Alter, T., Kasimir, S., Linnebur, M., Fehlhaber, K., 2005. Prevalence of *Yersinia enterocolitica* in fattening pigs. J. Food Prot. 68, 850–854.

Hachicha, R., Rekik, O., Hachicha, S., Ferchichi, M., Woodward, S., Moncef, N., Cegarra, J., Mechichi, T., 2012. Co-composting of spent coffee ground with olive mill wastewater sludge and poultry manure and effect of Trametes versicolor inoculation on the compost maturity. Chemosphere 88, 677–682.

Hakkinen, M., Hänninen, M.-L., 2009. Shedding of *Campylobacter* spp. in Finnish cattle on dairy farms. J. Appl. Microbiol. 107, 898–905.

Haller, M.Y., Muller, S.R., McArdell, C.S., Alder, A.C., Suter, M.J.F., 2002. Quantification of veterinary antibiotics (sulfonamides and trimethoprim) in animal manure by liquid chromatography-mass spectrometry. J. Chromatogr. 952, 111–120.

Hammesfahr, H., Kotzerke, A., Lamshöft, M., Wilke, B.M., Kandeler, E., Thiele-Bruhn, S., 2011. Effects of sulfadiazine-contaminated fresh and stored manure on a soil microbial community. Eur. J. Soil Biol. 47, 61–68.

Hamscher, G., Sczesny, S., Höper, H., Nau, H., 2002. Determination of persistent tetracycline residues in soil fertilized with liquid manure by a high-performance liquid chromatography with electrospray ionization tandem mass spectrometry. Anal. Chem. 74, 1509–1518.

Hancock, D.D., Besser, T.E., Kinsel, M.L., Tarr, P.I., Rice, D.H., Paros, M.G., 1994. The prevalence of *Escherichia coli* 0157. H7 in dairy and beef cattle in Washington State. Epidemiol. Infect. 113, 199–207.

Hancock, D.D., Besser, T.E., Rice, D.H., Herriott, D.E., Tarr, P.I., 1997a. A longitudinal study of *Escherichia coli* 0157 in fourteen cattle herds. Epidemiol. Infect. 118, 193–195.

Hancock, D.D., Rice, D.H., Thomas, L.A., Dargatz, D.A., Besser, T.E., 1997b. Epidemiology of *Escherichia coli* 0157 in feedlot cattle. J. Food Prot. 60, 462–465.

Hanifi, S., El Hadrami, I., 2008. Phytotoxicity and fertilising potential of olive mill wastewaters for maize cultivation. Agron. Sustain. Dev. 28, 313–319.

Hanselman, T.A., Graetz, D.A., Wilkie, A.C., 2003. Manure-borne estrogens as a potential environmental contaminant: a review. Environ. Sci. Technol. 37, 5471–5478.

Hansen, M., Krogh, K.A., Björklund, E., Brandt, A., Halling-Sørensen, B., 2009. Environmental risk assessment of ionophores. Trends Anal. Chem. 28, 534–542.

Harrison, A.F., 1987. Soil Organic Phosphorus: A Review of World Literature. CAB International, Wallingford, UK.

Hartnach, S., Doherr, M.G., Alter, T., Toutounian-Mashad, K., Greiner, M., 2009. *Campylobacter* monitoring in German broiler flocks: an explorative time series analysis. Zoonoses Public Health 56, 117–128.

Haruna, M., Sasaki, Y., Murakami, M., Ikeda, A., Kusukawa, M., Tsujiyama, Y., Ito, K., Asai, T., Yamada, Y., 2012. Prevalence and antimicrobial susceptibility of *Campylobacter* in broiler flocks in Japan. Zoonoses Public Health 59, 241–245.

Harvey, R.W., 1991. Parameters involved in modelling movement of bacteria in groundwater. In: Hurst, C.J. (Ed.), Modelling the Environmental Fate of Microorganisms, American Society for Microbiology, Washington DC, U.S.A. pp. 89–114.

Heckrath, G., Brookes, P.C., Poulton, P.R., Goulding, K.W.T., 1997. Phosphorus losses in drainage water from an arable silty clay loam. In: Tunney, H. (Ed.), Phosphorus Loss from Soil to Water, CAB International, Wallingford, UK, pp. 367–369.

Hegesh, E., Shiloah, J., 1982. Blood nitrates and infantile methemoglobinemia. Clin. Chim. Acta 125, 107–115.

Heinonen-Tanski, H., Mohaibes, M., Karinen, P., Koivunen, J., 2006. Methods to reduce pathogen microorganisms in manure. Livestock Sci. 102, 248–255.

Hergert, G.W., Bouldin, D.R., Klausner, S.D., Zwerman, P.J., 1981. Phosphorus concentration-water flow interactions in tile effluent from manured land. J. Environ. Qual. 10, 338–344.

Hernandez-Ramirez, G., Brouder, S.M., Ruark, M.D., Turco, R.F., 2011. Nitrate, phosphate, and ammonium loads at subsurface drains: agroecosystems and nitrogen management. J. Environ. Qual. 40, 1229–1240.

Herriott, D.E., Hancock, D.D., Ebel, E.D., Carpenter, L.V., Rice, D.H., Besser, T.E., 1998. Association of herd management factors with colonization of dairy cattle by shiga toxin-positive *Escherichia coli* O157. J. Food Prot. 61, 802–807.

Heuvelink, A.E., Zwartkruis-Nahuis, J.T.M., van den Biggelaar, F.L.A.M., van Leeuwen, W.J., de Boer, E., 1999. Isolation and characterization of verocytotoxin-producing *Escherichia coli* O157 from slaughter pigs and poultry. Int. J. Food Microbiol. 52, 67–75.

Hilborn, E.D., Mermin, J.H., Mshar, P.A., Hadler, J.L., Voetsch, A., Wojtkunski, C., Swartz, M., Mshar, R., Lambert-Fair, M.-A., Farrar, J.A., Glynn, M.K., Slutsker, L., 1999. A

multistate outbreak of *Escherichia coli* O157:H7 infections associated with consumption of Mesclun lettuce. Arch. Intern. Med. 159, 1758–1764.
Himathongkham, S., Bahari, S., Riemann, H., Cliver, D., 1999. Survival of *Escherichia coli* O157:H7 and *Salmonella typhimurium* in cow manure and cow manure slurry. FEMS Microbiol. Lett. 178, 251–257.
Hoese, A., Clay, S.A., Clay, D.E., Oswald, J., Trooien, T., Thaler, R., Carlson, C.G., 2009. Chlotetracycline and tylosin runoff from soils treated with antimicrobial containing manure. Environ. Sci. Health, Part B 44, 371–378.
Hogan, J.S., Smith, K.L., Hoblet, K.H., Todhunter, D.A., Schoenberger, P.S., Hueston, W.D., Pritchard, D.E., Bowman, G.L., Heider, L.E., Brockett, B.L., Conrad, H.R., 1989. Bacterial counts in bedding materials used on nine commercial dairies. J. Dairy Sci. 72, 250–258.
Hölzel, C., Bauer, J., 2008. Salmonella spp. in Bavarian liquid pig manure: occurrence and relevance for the distribution of antibiotic resistance. Zoonoses Public Health 55, 133–138.
Hölzel, C.S., Schwaiger, K., Harms, K., Küchenhoff, H., Kunz, A., Meyer, K., Müller, C., Bauer, J., 2010. Sewage sludge and liquid pig manure as possible sources of antibiotic resistant bacteria. Environ. Res. 110, 318–326.
Hristov, A.N., Hanigan, M., Cole, A., Todd, R., McAllister, T.A., Ndegwa, P.M., Rotz, A., 2011. Review: ammonia emissions from dairy farms and beef feedlots. Can. J. Anim. Sci. 91, 1–35.
Hu, P., Wang, A.S., Engledow, A.S., Hollister, E.B., Rothlisberger, K.L., Matocha, J.E., Zuberer, D.A., Provin, T.L., Hons, F.M., Gentry, T.J., 2011. Inhibition of the germination and growth of *Phymatotrichopsis omnivora* (cotton root rot) by oilseed meals and isothiocyanates. Appl. Soil Ecol. 49, 68–75.
Hubbard, R.K., Sheridan, R.G., 1983. Water and nitrate losses from a small upland coastal plain watershed. J. Environ. Qual. 12, 291–295.
Hubbard, R.K., Leonard, R.A., Johnson, A.W., 1991. Nitrate transport on a sandy coastal plain soil underlain by plinthite. Trans. ASAE. 34, 802–808.
Huetink, R.E.C., van der Giessen, J.W.B., Noordhuizen, J.P.T.M., Ploeger, H.W., 2001. Epidemiology of *Cryptosporidium* spp. and *Giardia duodenalis* on a dairy farm. Vet. Parasitol. 102, 53–67.
Hulebak, K.L., Schlosser, W., 2002. Hazard analysis and critical control point (HACCP): history and conceptual overview. Risk Anal. 22, 547–552.
Humpage, A.R., Fontaine, F., Froscio, S., Burcham, P., Falconer, I.R., 2005. Cylindrospermopsin genotoxicity and cytotoxicity: role of cytochrome p-450 and oxidative stress. J. Toxicol. Environ. Health, Part A 68, 739–753.
Hurd, H.S., McKean, J.D., Griffith, R.D., Rostagno, M.H., 2003. Estimation of the *Salmonella enterica* prevalence in finishing swine. Epidemiol. Infect. 132, 127–135.
Hutchins, S.R., White, M.V., Hudson, F.M., Fine, D.D., 2007. Analysis of lagoon samples from different concentrated animal feeding operations for estrogens and estrogen conjugates. Environ. Sci. Technol. 41, 738–744.
Hutchison, M.L., Ashmore, A.K., Crookes, K.M., Wilson, D.W., Grives S.J., Chambers, B.T., Keevil, C.W., Moore, A., 2002. Enumeration of pathogens in livestock wastes and factors affecting their survival. In: "Proceedings of the Joint CIWEM and Aqua Enviro Technology Transfer 7th European Biosolids and Organic Residuals Conference", September 2000, Wakefield, UK, pp. S3.15.1–S3.15.7.
Hutchison, M.L., Walters, L.D., Avery, S.M., Synge, B.A., Moore, A., 2004a. Levels of zoonotic agents in British livestock manures. Lett. Appl. Microbiol. 39, 207–214.
Hutchison, M.L., Walters, L.D., Moore, A., Crookes, K.M., Avery, S.M., 2004b. Effect of length of time before incorporation on survival of pathogenic bacteria present in livestock wastes applied to agricultural soil. Appl. Environ. Microbiol. 70, 5111–5118.
Hutchison, M.L., Walters, L.D., Avery, S.M., Munro, F., Moore, A., 2005a. Analyses of livestock production, waste storage, and pathogen levels and prevalences in farm manures. Appl. Environ. Microbiol. 71, 1231–1236.

Hutchison, M.L., Walters, L.D., Avery, S.M., Moore, A., 2005b. Decline of zoonotic agents in livestock waste and bedding heaps. J. Appl. Microbiol. 99, 354–362.

Hutchison, M.L., Walters, L.D., Moore, A., Avery, S.M., 2005c. Decline of zoonotic agents in liquid livestock wastes stored in batches on-farm. J. Appl. Microbiol. 99, 58–65.

Hutchison, M.L., Avery, S.M., Monaghan, J.M., 2008. The air-borne distribution of zoonotic agents from livestock waste spreading and microbiological risk to fresh produce from contaminated irrigation sources. J. Appl. Microbiol. 105, 848–857.

Ivie, G.W., Christopher, R.J., Munger, C.E., Coppock, C.E., 1986. Fate and residues of (4-^{14}C) estradiol-17β after intramuscular injection into Holstein steer calves. J. Anim. Sci. 62, 681–690.

Jacobson, K.H., Lee, S., Somerville, R.A., McKenzie, D., Benson, C.H., Pedersen, J.A., 2010. Transport of the pathogenic prion protein through soils. J. Environ. Qual. 39, 1145–1152.

Jardine, P.M., Wilson, G.V., Luxmoore, R.J., 1990. Unsaturated solute transport through a forest soil during rain storm events. Geoderma 46, 103–118.

Jemison, J.M., Fox, R.H., 1994. Nitrate leaching from nitrogen B fertilized and manured corn measured with zero-tension pan lysimeters. J. Environ. Qual. 23, 337–343.

Jenkins, M.B., Endale, D.M., Schomberg, H.H., Sharpe, R.R., 2006. Fecal bacteria and sex hormones in soil and runoff from cropped watersheds amended with poultry litter. Sci. Total Environ. 358, 164–177.

Jenkins, M.B., Truman, C.C., Siragusa, G., Line, E., Bailey, J.S., Frye, J., Endale, D.M., Franklin, D.H., Schomberg, H.H., Fisher, D.S., Sharp, R.R., 2008. Rainfall and tillage effects on transport of fecal bacteria and sex hormones 17β-estradiol and testosterone from broiler litter applications to a Georgia Piedmont Ultisol. Sci. Total Environ. 403, 154–163.

Jenkinson, D.S., 1991. The Rothamsted long-term experiments: are they still of use? Agron. J. 83, 2–10.

Jenneman, G.E., McInerney, M.J., Knapp, R.M., 1985. Microbial penetration through nutrient-saturated Berea sandstone. Appl. Environ. Microbiol. 50, 383–391.

Jensen, A.N., Dalsgaard, A., Baggesen, D.L., Nielsen, E.M., 2006a. The occurrence and characterization of *Campylobacter jejuni* and *C. coli* in organic pigs and their outdoor environment. Vet. Microbiol. 116, 96–105.

Jensen, A.N., Dalsgaard, A., Stockmarr, A., Nielsen, E.M., Baggesen, D.L., 2006b. Survival and transmission of *Salmonella enterica* serovar typhimurium in an outdoor organic pig farming environment. Appl. Environ. Microbiol. 72, 1833–1842.

Jiang, X., Morgan, J., Doyle, M.P., 2002. Fate of *Escherichia coli* O157:H7 in manure-amended soil. Appl. Environ. Microbiol. 68, 2605–2609.

Jjemba, P.K., 2002a. The potential impact of veterinary and human therapeutic agents in manure and biosolids on plants grown on arable land: a review. Agric. Ecosyst. Environ. 93, 267–278.

Jjemba, P.K., 2002b. The effect of chloroquine, quinacrine, and metronidazole on both soybean plants and soil microbiota. Chemosphere 46, 1019–1025.

Johannessen, G.S., Bengtsson, G.B., Heier, B.T., Bredholt, S., Wasteson, Y., Rørvik, L.M., 2005. Potential uptake of *Escherichia coli* O157:H7 from organic manure into crisphead lettuce. Appl. Environ. Microbiol. 71, 2221–2225.

Johnson, A.W., 1985. Specific crop rotation effects combined with cultural practices and nematicides. In: Sasser, J.N., Carter, C.C. (Eds.), An Advanced Treatise on Meloidogyne: Biology and Control, North Carolina State University Graphics, Raleigh, North Carolina, USA, pp. 283–301.

Johnson, C.J., Pedersen, J.A., Chappell, R.J., McKenzie, D., Aiken, J.M., 2007. Oral transmissibility of prion disease is enhanced by binding to soil particles. PLoS Pathog. 3, e93. http://dx.doi.org/10.1371/journal.ppat.0030093.

Johnston, A.E., Poulton, P.R., 1997. The downward movement and retention of phosphorus in agricultural soils. In: Tunney, H. (Ed.), Phosphorus Loss from Soil to Water, CAB International, Wallingford, UK, pp. 422–425.

Johnston, A.E., Poulton, P.R., Coleman, K., 2009. Soil organic matter: Its importance in sustainable agriculture and carbon dioxide fluxes. Adv. Agron. 101, 1–57.

Jokela, W.E., 1992. Nitrogen fertilizer and dairy manure effects on corn yield and soil nitrate. Soil Sci. Soc. Am. J. 56, 148–154.

Joshi, R.C., Haokip, D.D., Singh, K.N., 1994. Effect of green manuring on the physical properties of soil under a rice–wheat cropping system. J. Agric. Sci. 122, 107–113.

Kaiser, D.E., Mallarino, A.P., Haq, M.U., Allen, B.L., 2009. Runoff phosphorus loss immediately after poultry manure application as influenced by the application rate and tillage. J. Environ. Qual. 38, 299–308.

Kantor, L.S., Lipton, K., Manchester, A., Oliveira, V., 1997. Estimating and addressing America's food loss. Food Rev. 20, 2–12.

Karci, A., Balcioğlu, I.K., 2009. Investigation of the tetracycline, sulfonamide, and fluoroquinolone antimicrobial compounds in animal manure and agricultural soils in Turkey. Sci. Total Environ. 407, 4652–4664.

Kay, P., Blackwell, P.A., Boxall, A.B.A., 2005a. Transport of veterinary antibiotics in overland flow following the application of slurry to arable land. Chemosphere 59, 951–959.

Kay, P., Blackwell, P.A., Boxall, A.B.A., 2005b. Column studies to investigate the fate of veterinary antibiotics in clay soils following slurry application to agricultural land. Chemosphere 60, 497–507.

Kay, P., Blackwell, P.A., Boxall, A.B.A., 2005c. A lysimeter experiment to investigate the leaching of veterinary antibiotics through a clay soil and comparison with field data. Environ. Pollut. 134, 333–341.

Kelessidis, A., Stasinakis, A.S., 2012. Comparative study of the methods used for treatment and final disposal of sewage sludge in European countries. Waste Manage. 32, 1186–1195.

Khaleel, R., Reddy, K.R., Overcash, M.R., 1981. Changes in soil physical properties due to organic waste applications: a review. J. Environ. Qual. 10, 133–141.

King, L.D., Safley, L.M., Spears, J.W., 1983. A greenhouse study on the response of corn (*Zea mays* L.) to manure from beef cattle fed antibiotics. Agric. Wastes 8, 185–190.

Kinney, C., Furlong, E., Kolpin, D.W., Burkhardt, M.R., Zaugg, S.D., Werner, S.L., Bossio, J.P., Benotti, M.J., 2008. Bioaccumulation of pharmaceuticals and other anthropogenic waste indicators in earthworms from agricultural soil amended with biosolid or swine manure. Environ. Sci. Technol. 42, 1863–1870.

Kirchmann, H., 1985. Losses, plant uptake and utilisation of manure nitrogen during a production cycle. Acta Agric. Scand. Suppl. 24, 1–77.

Kirchmann, H., Lundvall, A., 1993. Relationship between N immobilization and volatile fatty acids in soil after application of pig and cattle slurry. Biol. Fertil. Soils 15, 161–164.

Kirkhorn, S.R., Garry, V.F., 2000. Agricultural lung diseases. Environ. Health Perspect. 108, 705–712.

Kishima, M., Uchida, I., Namimatsu, T.T., Osumi, T., Takahashi, S.K., Tanaka, K., Aoki, H., Matsuura, K., Yamamoto, K., 2008. Nationwide surveillance of salmonella in the faeces of pigs in Japan. Zoonoses Public Health 55, 139–144.

Kjær, J., Olsen, P., Bach, K., Barlebo, H.C., Ingerslev, F., Hansen, M., Sørensen, B.H., 2007. Leaching of estrogenic hormones from manure-treated structured soils. Environ. Sci. Technol. 41, 3911–3917.

Kleinman, P.J.A., Sharpley, A.N., Veith, T.L., Maguire, R.O., Vadas, P.A., 2004. Evaluation of phosphorus transport in surface runoff from packed soil boxes. J. Environ. Qual. 33, 1413–1423.

Kolenbrander, G.J., 1969. Nitrate content and nitrogen loss in drainwater. Neth. J. Agric. Sci. 17, 246–255.

König, N., Baccini, P., Ulrich, B., 1986. The influence of natural organic substances on the distribution of metals between soil and soil solution in an acid forest soil. Z. Pflanz. Bodenkunde 149, 68–82 (in German).

Konold, T., Moore, S.J., Bellworthy, S.J., Simmons, A., 2008. Evidence of scrapie transmission via milk. BMC Vet. Res. 4, 14. http://dx.doi.org/10.1186/1746-6148-4-14.
Kowal, N.E., 1985. Health Effects of Land Application of Municipal Sludge. Health Effects Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, North Carolina, USA.
Kuchta, S.L., Cessna, A.J., Elliott, J.A., Peru, K.M., Headley, J.V., 2009. Transport of lincomycin to surface and ground water from manure-amended cropland. J. Environ. Qual. 38, 1719–1727.
Kukier, U., Chaney, R.L., Ryan, J.A., Daniels, W.L., Dowdy, R.H., Granto, T.C., 2010. Phytoavailability of cadmium in long-term biosolids-amended soils. J. Environ. Qual. 39, 519–530.
Kumar, K., Rosen, C.J., Gupta, S.C., McNearney, M., 2009. Land application of sugar beet by-products: effects on nitrogen mineralization and crop yields. J. Environ. Qual. 38, 319–328.
Laegreid, W.W., Elder, R.O., Keen, J.E., 1999. Prevalence of *Escherichia coli* O157:H7 in range beef calves at weaning. Epidemiol. Infect. 129, 163–172.
Lahti, E., Ruoho, O., Rantala, L., Hänninen, M.-L., Honkanen-Buzalski, T., 2003. Longitudinal study of *Escherichia coli* O157 in a cattle finishing unit. Appl. Environ. Microbiol. 73, 2180–2190.
Lal, R., 2005. Enhancing crop yields in the developing countries through restoration of the soil organic carbon pool in agricultural lands. Land Degrad. Dev. 17, 197–209.
Lampkin, N., 1990. Organic Farming. Farming Press Books, Ipswich, UK.
Lamshöft, M., Sukul, P., Zühlke, S., Spiteller, M., 2007. Metabolism of ^{14}C-labelled and non-labelled sulfadiazine after administration to pigs. Anal. Bioanal. Chem. 388, 1733–1745.
Lamshöft, M., Sukul, P., Zühlke, S., Spiteller, M., 2010. Behaviour of ^{14}C-sulfadiazine and ^{14}C-difloxacin during manure storage. Sci. Total Environ. 408, 1563–1568.
Lance, J.C., Gerba, C.P., 1984. Virus movement in soil during saturated and unsaturated flow. Appl. Environ. Microbiol. 47, 335–337.
Lance, J.C., Gerba, C.P., Melnick, J.L., 1976. Virus movement in soil columns flooded with secondary sewage effluent. Appl. Environ. Microbiol. 32, 520–526.
Lance, J.C., Gerba, C.P., Wang, D.S., 1982. Comparative movement of different enteroviruses in soil columns. J. Environ. Qual. 11, 347–351.
Lange, I.G., Daxenberger, A., Schiffer, B., Witters, H., Ibarreta, D., Meyer, H.H.D., 2002. Sex hormones originating from different livestock production systems: fate and potential disrupting activity in the environment. Anal. Chim. Acta 473, 27–37.
Lapen, D.R., Topp, E., Edwards, M., Sabourin, L., Curnoe, W., Gottschall, N., Bolton, P., Rahman, S., Ball-Coelho, B., Payne, M., Kleywegt, S., McLaughlin, N., 2008. Effect of liquid municipal biosolid application method on tile and ground water quality. J. Environ. Qual. 37, 925–936.
Larkin, R.P., Griffin, T.S., 2007. Control of soilborne potato diseases using Brassica green manures. Crop Prot. 26 (7), 1067–1077.
Larsbo, M., Lapen, D.R., Topp, E., Metcalfe, C., Abbaspour, K.C., Fenner, K., 2009. Simulation of pharmaceutical and personal care product transport to tile drains after biosolids application. J. Environ. Qual. 38, 1274–1285.
Lax, A., Diaz, E., Castillo, V., Albaladejo, J., 1994. Reclamation of physical and chemical properties of a salinized soil by organic amendment. Arid Soil Res. Rehabil. 8, 9–17.
Lear, B., 1956. Results of laboratory experiments with Vapam for control of nematodes. Plant Dis. Rep. 40, 847–852.
Leifert, C., Ball, K., Volakakis, N., Cooper, J.M., 2008. Control of enteric pathogens in ready-to-eat vegetable crops in organic and 'low input' production systems: a HACCP-based approach. J. Appl. Microbiol. 105, 931–950.

Létourneau, V., Duchaine, C., Côté, C., Letellier, A., Topp, E., Massé, D., 2010. Presence of zoonotic pathogens in physico-chemically characterized manures from hog finishing houses using different production systems. Biores. Technol. 101, 4048–4055.

Lewis, J.A., Papavizas, G.C., 1971. Effect of sulfur-containing volatile compounds and vapors from cabbage decomposition on *Aphanomyces euteiches*. Phytopathology 61, 208–214.

Leytem, A.B., Bjorneberg, D.L., Sheffield, R.E., de Haro Marti, M.E., 2009. Case study: on-farm evaluation of liquid dairy manure application methods to reduce ammonia losses. Prof. Anim. Sci. 25, 93–98.

Liang, J., Wang, X., Xiao, Y., Cui, Z., Xia, S., Hao, Q., Yang, J., Luo, L., Xu, J., Kan, B., Jing, H., 2012. Prevalence of *Yersinia enterocolitica* in pigs slaughtered in Chinese abattoirs. Appl. Environ. Microbiol. 78, 2949–2956.

Ligios, C., Cancedda, M.G., Carta, A., Santucciu, C., Maestrale, C., Demontis, F., Saba, M., Patta, C., DeMartini, J.C., Aguzzi, A., Sigurdson, C., 2011. Sheep with scrapie and mastitis transmit infectious prions through milk. J. Virol. 85, 1136–1139.

Litskas, V.D., Karamanlis, X.N., Batzias, G.C., Kamarianos, A.P., 2011. Sorption of the antiparasitic drug eprinomectin in three soils. Chemosphere 82, 193–198.

Liu, Z., Ogeja, A., Pruden, A., Knowlton, K.F., 2011. Occurrence, fate and removal of synthetic oral contraceptives (SOCs) in the natural environment: a review. Sci. Total Environ. 409, 5149–5161.

Lo Fo Wong, D.M.A., Dahl, J., van der Wolf, P.J., Wingstrand, A., Leontides, L., von Altrock, A., 2003. Recovery of *Salmonella enterica* from seropositive finishing pig herds. Vet. Microbiol. 97, 163–170.

Loke, M.L., Ingerslev, F., Sorensen, B.H., Tjornelund, J., 2000. Stability of Tylosin A in manure containing test systems determined by high performance liquid chromatography. Chemosphere 20, 759–765.

Lomaestro, B.M., Bailie, G.R., 1995. Absorption interactions with fluoroquinolones. 1995 update. Drug Saf. 12, 314–333.

Looper, M.L., Edrington, T.S., Callaway, T.R., Rosenkrans, C.F., 2009a. Fate of *Escherichia coli* O157:H7 and Salmonella from contaminated manure slurry applied to soil surrounding tall fescue. Lett. Appl. Microbiol. 48, 513–516.

Looper, M.L., Edrington, T.S., Rosenkrans, C.F., 2009b. Influence of body condition and forage type on prevalence of *Escherichia coli* O157:H7 and Salmonella in grazing beef cows. Lett. Appl. Microbiol. 49, 361–365.

Lorenzo Lorenzo, M.J., Ares-Mazás, E., Villacorta Martínez de Maturana, I., 1993. Detection of oocysts and IgG antibodies to *Cryptosporidium parvurn* in asymptomatic adult cattle. Vet. Parasitol. 47, 9–15.

Lowrance, R., 1992. Nitrogen outputs from a field-size agricultural watershed. J. Environ. Qual. 21, 602–607.

Luangtongkum, T., Morishita, T.Y., Ison, A.J., Huang, S., McDermott, P.F., Zhang, Q., 2006. Effect of conventional and organic production practices on the prevalence and antimicrobial resistance of *Campylobacter* spp. in poultry. Appl. Environ. Microbiol. 72, 3600–3607.

Lucas, S.D., Jones, D.L., 2006. Biodegradation of estrone and 17 β-estradiol in grassland soils amended with animal wastes. Soil Biol. Biochem. 38, 2803–2815.

Lucas, S.D., Jones, D.L., 2009. Urine enhances the leaching and persistence of estrogens in soils. Soil Biol. Biochem. 41, 236–242.

Lund, E., Nissen, B., 1983. The survival of Enteroviruses in aerated and unaerated cattle and pig slurry. Agric. Wastes 7, 221–233.

Lægdsmand, M., Andersen, H., Hørbye, O., Halling-Sørensen, B., 2009. Transport and fate of estrogenic hormones in slurry-treated soil monoliths. J. Environ. Qual. 38, 955–964.

L'hirondel, J., L'hirondel, J.-L., 2002. Nitrate and Man: Toxic, Harmless or Beneficial. CABI Publishing, Wallingford, UK.

Ma, X., Benson, C.H., McKenzie, D., Aiken, J.M., Pedersen, J.A., 2007. Adsorption of pathogenic prion protein to quartz sand. Environ. Sci. Technol. 41, 2324–2330.
MacLean, A.J., 1983. Pathogens of animals in manure: environmental impact and public health. Farm Animal Manures in the Canadian Environment, vol. NRCC/No. 189766. National Research Council of Canada Associate Committee on Scientific Criteria for Environmental Quality, Ottawa, Ontario, Canada, pp. 103–107.
MacRae, H.F., Zaharia, W., Common, R.H., 1959. Isolation of crystalline estradiol-17β from droppings of laying hens. Poult. Sci. 38, 318–321.
Maddison, B.C., Owen, J.P., Bishop, K., Shaw, G., Rees, H.C., Gough, K.C., 2010a. The interaction of ruminant PrPSc with soils is influenced by prion source and soil type. Environ. Sci. Technol. 44, 8503–8508.
Maddison, B.C., Rees, H.C., Baker, C.A., Taema, M., Bellworthy, J., Thorne, L., Terry, L.A., Gough, K.C., 2010b. Prions are secreted into the oral cavity in sheep with preclinical scrapie. J. Infect. Dis. 201 1672–1176.
Maddox-Hyttel, C., Langkjaer, R.B., Enemark, H.L., Vigre, H., 2006. *Cryptosporidium* and *Giardia* in different age groups of Danish cattle and pigs-occurrence and management associated risk factors. Vet. Parasitol. 141, 48–59.
Mafu, A.A., Higgins, R., Nadeau, M., Cousineau, G., 1989. The incidence of *Salmonella*, *Campylobacter*, and *Yersinia enterocolitica* in swine carcasses and the slaughterhouse environment. J. Food Prot. 52, 642–645.
Magdoff, F.R., Amadon, F.F., 1980. Nitrogen availability from sewage sludge. J. Environ. Qual. 9, 451–455.
Manser, P.A., Dalziel, R.W., 1985. A survey of *Campylobacter* in animals. J. Hyg. (Lond.) 95, 15–21.
Mashtare, M.L., Khan, B., Lee, L.S., 2011. Evaluating stereoselective sorption by soils of 17α-estradiol and 17β-estradiol. Chemosphere 82, 847–852.
Mathur, R.S., Common, R.H., 1969. A note on the daily urinary excretion of estradiol-17β and estrone by the hen. Poult. Sci. 48, 100–104.
Mathur, R.S., Anastassiadis, P.A., Common, R.H., 1966. Urinary excretion of estrone and of 16-epi-estriol plus 17-epi-estriol by the hen. Poult. Sci. 45, 946–952.
Matthess, G., Pekdeger, A., Schroeter, J., 1988. Persistence and transport of bacteria and viruses in groundwater - a conceptual evaluation. J. Contam. Hydrol. 2, 171–188.
Mawdsley, J.L., Badgett, R.D., Merry, R.J., Pain, B.F., Theodorou, M.K., 1995. Pathogens in livestock waste, their potential for movement through soil and environmental pollution. Appl. Soil Ecol. 2, 1–15.
Mayes, T., 1998. Risk analysis in HACCP: burden or benefit? Food Control 9, 171–176.
McClellan, K., Halden, R.U., 2010. Pharmaceuticals and personal care products in archived U.S. biosolids from the 2001 EPA national sewage sludge survey. Water Res. 44, 658–668.
McEvoy, J.M., Giddings, C.W., 2009. *Crytosporidium* in commercially produced turkeys on-farm and postslaughter. Lett. Appl. Microbiol. 48, 302–306.
McGinn, S.M., Janzen, H.H., Coates, T., 2003. Atmospheric ammonia, volatile fatty acids, and other odorants near beef feedlots. J. Environ. Qual. 32, 1173–1182.
McLeod, R.V., Hegg, R.O., 1984. Pasture runoff water quality from application of inorganic and organic nitrogen sources. J. Environ. Qual. 13, 122–126.
Mechie, S.C., Chapman, P.A., Siddons, C.A., 1997. A fifteen month study of *Escherichia coli* 0157:H7 in a dairy herd. Epidemiol. Infect. 118, 17–25.
Meisinger, J.J., Randall, G.W., 1991. Estimating nitrogen budgets for soil-crop system. In: Follet, R.F., Keeney, D.R., Cruse, R.M. (Eds.), Managing Nitrogen for Groundwater Quality and Farm Profitability, Soil Science Society of America Inc., Madison, Wisconsin, USA., pp. 85–124.
Mejía, W., Casal, J., Zapata, D., Sánchez, G.J., Martín, M., Mateu, E., 2006. Epidemiology of salmonella infections in pig units and antimicrobial susceptibility profiles of the strains of Salmonella species isolated. Vet. Rec. 159, 271–276.

Meldrum, R.J., Smith, R.M.M., Wilson, I.G., 2006. Three-year surveillance program examining the prevalence of *Campylobacter* and *Salmonella* in whole retail raw chicken. J. Food Prot. 69, 928–931.

Meng, X.J., Purcell, R.H., Halbur, P.G., Lehman, J.R., Webb, D.M., Tsareva, T.S., Haynes, J.S., Thacker, B.J., Emerson, S.U., 1997. A novel virus in swine is closely related to the human hepatitis E virus. Proc. Natn. Acad. Sci. U.S.A. 94, 9860–9865.

Menzi, H., Kessler, J., 1998. Heavy metal content of manures in Switzerland. In: Martinez, J., Maudet, M. (Eds.), RAMIRAN 98. Proc. 8th Int. Conference of the FAO ESCORENA Network on Recycling of Agricultural, Municipal, and Industrial Residues in Agriculture. Rennes, France, 26–29 May 1998, FAO and Cemagref, France, pp. 495–506.

Merialdi, G., Barigazzi, G., Bonilauri, P., Tittarelli, C., Bonci, M., D'incau, M., Dottori, M., 2008. Longitudinal study of Salmonella infection in Italian farrow-to-finish swine herds. Zoonoses Public Health 55, 222–226.

Miller, J.J., Olson, E.C.S., Chanasyk, D.S., Beasley, B.W., Larney, F.J., Olson, B.M., 2006. Phosphorus and nitrogen in rainfall simulation runoff after fresh and composted beef cattle manure application. J. Environ. Qual. 35, 1279–1290.

Millner, P.D., 2009. Bioaerosols associated with animal production operations. Biores. Technol. 100, 5379–5385.

Mills, L.J., Chichester, C., 2005. Review of evidence: are endocrine-disrupting chemicals in the aquatic environment impacting fish populations? Sci. Total Environ. 343, 1–34.

Misselbrook, T.H., Shepherd, M.A., Pain, B.F., 1996. Sewage sludge applications to grassland: influence of sludge type, time and method of application on nitrate leaching and herbage yield. J. Agric. Sci. (Camb.) 126, 343–352.

Mohammed, H.O., Atwill, E., Dunbar, L., Ward, T., McDonough, P., Gonzalez, R., Stipetic, K., 2010. The risk of Listeria monocytogenes infection in beef cattle operations. J. Appl. Microbiol. 108, 349–356.

Mojtahedi, H., Santo, G.S., Hang, A.N., Wilson, J.H., 1991. Suppression of root-knot nematode populations with selected rapeseed cultivars as green manure. J. Nematol. 23, 170–174.

Montenegro, M.A., Bülte, M., Trumpf, T., Aleksić, S., Reuter, G., Bulling, E., Helmuth, R., 1990. Detection and characterization of fecal verotoxin-producing *Escherichia coli* from healthy cattle. J. Clin. Microbiol. 28, 1417–1421.

Mosaddeghi, M.R., Mahboubi, A.A., Zandsalimi, S., Unc, A., 2009. Influence of organic waste type and soil structure on the bacterial filtration rates in unsaturated intact soil columns. J. Environ. Manage. 90, 730–739.

Mubiru, D.N., Coyne, M.S., Grove, J.H., 2000. Mortality of *Escherichia coli* O157:H7 in two soils with different physical and chemical properties. J. Environ. Qual. 29, 1821–1825.

Muller, M., Combalbert, S., Delgenès, N., Bergheaud, V., Rocher, V., Benoît, P., Delgenès, J.-P., Patureau, D., Hernandez-Raquet, G., 2010. Occurrence of estrogens in sewage sludge and their fate during plant-scale anaerobic digestion. Chemosphere 81, 65–71.

Munroe, D.L., Prescott, J.F., Penner, J.L., 1983. *Campylobacter jejuni* and *Campylobacter coli* serotypes isolated from chickens, cattle, and pigs. J. Clin. Microbiol. 18, 877–881.

Munyankusi, E., Gupta, S.C., Moncrief, J.F., Berry, E.C., 1994. Soil processes and chemical transport. J. Environ. Qual. 23, 773–784.

Murayama, Y., Yoshioka, M., Okado, H., Takata, M., Yokoyama, Y., Mohri, S., 2007. Urinary excretion and blood level of prions in scrapie-infected hamsters. J. Gen. Virol. 88, 2890–2898.

Musgrave, G.W., Nichols, M.L., 1943. III. Organic matter in relation to land use. Soil Sci. Soc. Am. J. 7, 22–28.

Narula, N., Deubel, A., Gransee, A., Behl, R.R., Merbach, W., 2002. Impact of fertilizers on total microbiological flora in planted and unplanted soils of long-term fertilization experiment. Arch. Agron. Soil Sci. 48, 171–180.

Nazir, J., Haumacher, R., Ike, A.C., Marschang, R.E., 2011. Persistence of avian influenza viruses in lake sediment, duck feces and duck meat. Appl. Environ. Microbiol. 77, 4981–4985.

Neeteson, J.J., 2000. Nitrogen and phosphorus management on Dutch dairy farms: legislation and strategies employed to meet the regulations. Biol. Fertil. Soils 30, 566–572.
Neilson, E.M., Tegtmeier, C., Andersen, H.J., Grønbæk, C., Andersen, J.S., 2002. Influence of age, sex and herd characteristics on the occurrence of verocytotoxin-producing *Escherichia coli* O157 in Danish dairy farms. Vet. Microbiol. 88, 245–257.
Newman, C.M., Rotenberg, D., Cooperband, L.R., 2005. Paper mill residuals and compost effects on particulate organic matter and related soil functions in a sandy soil. Soil Sci. 170, 788–801.
Nichols, D.J., Daniel, T.C., Moore, P.A., Edwards, D.R., Pote, D.H., 1997. Runoff of estrogen hormone 17β-Estradiol from poultry litter applied to pasture. J. Environ. Qual. 26, 1002–1006.
Nicholson, F.A., Chambers, B.J., Williams, J.R., Unwin, R.J., 1999. Heavy metal contents of livestock feeds and animal manures in England and Wales. Biores. Technol. 70, 23–31.
Nicholson, F.A., Groves, S.J., Chambers, B.J., 2005. Pathogen survival during livestock manure storage and following land application. Biores. Technol. 96, 135–143.
Nielsen, N.E., Jensen, H.E., 1990. Nitrate leaching from loamy soils as affected by crop rotation and nitrogen fertilizer application. Fert. Res. 26, 197–207.
Nielsen, T.D., Nielsen, L.R., Toft, N., 2011. Bayesian estimation of true between-herd and within-herd prevalence of Salmonella in Danish veal calves. Prev. Vet. Med. 100, 155–162.
Niu, Y.D., McAllister, T.A., Xu, Y., Johnson, R.P., Stephens, T.P., Stanford, K., 2009. Prevalence and impact of bacteriophages on the presence of *Escherichia coli* O157:H7 in feedlot cattle and their environment. Appl. Environ. Microbiol. 75, 1271–1278.
Notermans, S., Mead, G.C., 1996. Incorporation of elements of quantitative risk analysis in the HACCP system. Int. J. Food Microbiol. 30, 157–173.
Notermans, S., Gallhoff, G., Zwietering, M., Mead, G.C., 1995. The HACCP concept: specification of criteria using quantitative risk assessment. Food Microbiol. 12, 81–90.
Nyberg, K.A., Vinnerås, B., Ottoson, J.R., Aronsson, P., Albihn, A., 2010. Inactivation of *Escherichia coli* O157:H7 and Salmonella Typhimurium in manure-amended soils studied in outdoor lysimeters. Appl. Soil Ecol. 46, 398–404.
N'Dayegamiye, A., Tran, T.S., 2001. Effects of green manures on soil organic matter and wheat yields and N nutrition. Can. J. Soil Sci. 81, 371–382.
Obi, M.E., Ebo, P.O., 1995. The effects of organic and inorganic amendments on soil physical properties and maize production in a severely degraded sandy soil in southern Nigeria. Biores. Technol. 51, 117–123.
Obied, H.K., Bedgood Jr., D.R., Prenzler, P.D., Robards, K., 2007. Bioscreening of Australian olive mill waste extracts: biophenol content, antioxidant, antimicrobial and molluscicidal activities. Food Chem. Toxicol. 45, 1238–1248.
Ogden, I.D., MacRae, M., Strachan, N.J.C., 2004. Is the prevalence and shedding concentrations of *E. coli* 0157 in beef cattle in Scotland Seasonal? FEMS Microbiol. Lett. 233, 297–300.
Ogden, I.D., MacRae, M., Strachan, N.J.C., 2005. Concentration and prevalence of *Escherichia coli* 0157 in sheep faeces at pasture in Scotland. J. Appl. Microbiol. 98, 646–651.
Oliver, M.A., 1997. Soil and human health: a review. Eur. J. Soil Sci. 48, 573–592.
Olson, M., 2000. Transmission and survival of *Escherichia coli* O157:H7. Canadian Cattlemen's Association *E. coli* O157:H7 Workshop. November 27 and 28, 2000, Canadian Cattlemen's Association, Calgary, Alberta. Canada, pp. 28–33.
Olson, M.E., Thorlakson, C.L., Deselliers, L., Morck, D.W., D.W., McAllister, T.A., 1997. *Giardia* and *Cryptosporidium* in Canadian farm animals. Vet. Parasitol. 68, 375–381.
Olson, M.E., O'Handley, R.M., Ralston, B.J., McAllister, T.A., Thompson, R.C.A., 2004. Update on *Cryptosporidium* and *Giardia* infections in cattle. Trends Parasitol. 20, 185–191.
Omisakin, F., MacRae, M., Ogden, I.D., Strachan, N.J.C., 2003. Concentration and prevalence of *Escherichia coli* O157 in cattle feces at slaughter. Appl. Environ. Microbiol. 69, 2444–2447.

Oporto, B., Esteban, J.I., Aduriz, G., Juste, R.A., Hurtado, A., 2007. Prevalence and strain diversity of thermophilic campylobacters in cattle, sheep and swine farms. J. Appl. Microbiol. 103, 977–984.

Oporto, B., Esteban, J.I., Aduriz, G., Juste, R.A., Hurtado, A., 2008. *Escherichia coli* O157:H7 and non-O157 shiga toxin-producing *E. coli* in healthy cattle, sheep and swine herds in northern Spain. Zoonoses Public Health 55, 73–81.

Östling, C.E., Lindgren, S.E., 1991. Bacteria in manure and on manured and NPK fertilized silage crops. J. Sci. Food Agric. 55, 579–588.

Ott, P., Vogtmann, H., 1982. A Propos du Compostage du Fumier (Composting Farmyard Manure, Behandlung und Verwertung von Landwirtschaftsdungern). In: Hill, S.B., Ott, P. (Eds.), Basic Technics in Ecological Farming = Techniques de base en agriculture biologique = Grundsatzliche verfahren der okologischen landwirtschaft: papers presented at the 2nd international conference held by the IFOAM, Montreal, October 1–5, 1978; Le Maintien de la fertilite des sols = The Maintenance of soil fertility = Die Erhaltung der bodenfruchtbarkeit: exposes presentes a la 3eme conference internationale organisee par IFOAM, Bruxelles, du 3 au 5 Septembre 1980, International Federation of Organic Agriculture Movements, Birkhauser Verlag, Basel, Switzerland, pp. 357–365.

Paez-Rubio, T., Huab, X., Anderson, J., Peccia, J., 2006. Particulate matter composition and emission rates from the disk incorporation of class B biosolids into soil. Atmos. Environ. 40, 7034–7045.

Paez-Rubio, T., Ramarui, A., Sommer, J., Xin, H., Anderson, J., Peccia, J., 2007. Emission rates and characterization of aerosols produced during the spreading of dewatered Class B biosolids. Environ. Sci. Technol. 41, 3537–3544.

Paiba, G.A., Gibbens, J.C., Pascoe, S.J.S., Wilesmith, J.W., Kidd, S.A., Byrne, C., Ryan, J.B.M., Smith, R.P., McLaren, I.M., Futter, R.J., Kay, A.C.S., Jones, Y.E., Chappell, S.A., Willshaw, G.A., Cheasty, T., 2002. Faecal carriage of verocytotoxin-producing *Escherichia coli* O157 in cattle and sheep at slaughter in Great Britain. Vet. Rec. 150, 593–598.

Paiba, G.A., Wilesmith, J.W., Evans, S.J., Pascoe, S.J.S., Smith, R.P., Kidd, S.A., Ryan, J.B.M., McLaren, I.M., Chappell, S.A., Willshaw, G.A., Cheasty, T., French, N.P., Jones, T.W.H., Buchanan, H.F., Challoner, D.J., Colloff, A.D., Cranwell, M.P., Daniel, R.G., Davies, I.H., Duff, J.P., Hogg, R.A.T., Kirby, F.D., Millar, M.F., Monies, R.J., Nicholls, M.J., Payne, J.H., 2003. Prevalence of faecal excretion of verocytotoxigenic *Escherichia coli* O157 in cattle in England and Wales. Vet. Rec. 153, 347–353.

Palm, C.A., Gachengo, C.N., Delve, R.J., Cadisch, G., Giller, K.E., 2001. Organic inputs for soil fertility management in tropical agroecosystems: application of an organic resource database. Agric. Ecosyst. Environ. 83, 27–42.

Pang, L., 2009. Microbial removal rates in subsurface media estimated from published studies of field experiments and large intact soil cores. J. Environ. Qual. 38, 1531–1559.

Park, G.W., Diez-Gonzalez, F., 2003. Utilization of carbonate and ammonia-based treatments to eliminate *Escherichia coli* O157:H7 and *Salmonella typhimurium* DT104 from cattle manure. J. Appl. Microbiol. 94, 675–685.

Parker, D.B., 2008. Reduction of odor and VOC emissions from a dairy lagoon. Appl. Eng. Agric. 24, 647–655.

Parker, C.F., Sommers, L.E., 1983. Mineralization of nitrogen in sewage sludges. J. Environ. Qual. 12, 150–156.

Parveen, S., Taabodi, M., Schwarz, J.G., Oscar, T.P., Harter-Dennis, J., White, D.G., 2007. Prevalence and antimicrobial resistance of Salmonella recovered from processed poultry. J. Food Prot. 70, 2466–2472.

Patni, N.K., Jui, P.Y., 1987. Changes in solids and carbon content of dairy-cattle slurry in farm tanks. Biol. Wastes 20, 11–34.

Paul, J.W., Beauchamp, E.G., 1994. Short-term nitrogen dynamics in soil amended with fresh and composted cattle manures. Can. J. Soil Sci. 74, 147–155.

Peacock, A.D., Mullen, M.D., Ringelberg, D.B., Tyler, D.D., Hedrick, D.B., Gale, P.M., White, D.C., 2001. Soil microbial community responses to dairy manure or ammonium nitrate applications. Soil Biol. Biochem. 33, 1011–1019.
Pell, A.N., 1997. Manure and microbes: public and animal health problem? J. Dairy Sci. 80, 2673–2681.
Penttilä, O., Hurme, H., Neuvonen, P.J., 1975. Effect of zinc sulphate on the absorption of tetracycline and doxycycline in man. Eur. J. Clin. Pharmacol. 19, 131–134.
Perko-Mäkelä, P., Alter, T., Isohanni, P., Zimmermann, S., Lyhs, U., 2011. Distribution of *Campylobacter jejuni* isolates from turkey farms and different stages at slaughter using pulsed-field gel electrophoresis and flaA-short variable region sequencing. Zoonoses Public Health 58, 388–398.
Peterson, T.C., Ward, R.C., 1989. Development of a bacterial transport model for coarse soils. Water Resour. Bull. 25, 349–357.
Peterson, E.W., Davis, R.K., Orndorff, H.A., 2000. 17 β-Estradiol as an indicator of animal waste contamination in mantled karst aquifers. J. Environ. Qual. 29, 826–834.
Pilon, J., Higgins, R., Quessy, S., 2000. Epidemiological study of *Yersinia enterocolitica* in swine herds in Quebec. Can. Vet. J. 41, 383–387.
Pintar, K.D.M., Fazil, A., Pollari, F., Waltner-Toews, D., Charron, D.F., McEwen, S.A., Walton, T., 2012. Considering the risk of infection by *Cryptosporidium* via consumption of municipally treated drinking water from a surface water source in a Southwestern Ontario community. Risk Anal. 32, 1122–1138.
Pradham, A.K., Van Kessel, J.S., Karns, J.S., Wolfgang, D.R., Hovingh, E., Nelen, K.A., Smith, J.M., Whitlock, R.H., Fyock, T., Ladely, S., Fedorka-Cray, P.J., Schukken, Y.H., 2009. Dynamics of endemic infectious diseases of animal and human importance on three dairy herds in the northeastern United States. J. Dairy Sci. 92, 1811–1825.
Provin, T.L., Joern, B.C., Franzmeier, D.P., Sutton, A.L., 1995. Phosphorus retention in selected Indiana soils using short-term sorption isotherms and long-term aerobic incubations. In: Steele, K. (Ed.), Animal Waste and Land-water Interface, CRC Press, Inc., Boca Rotan, Florida, USA, pp. 35–42.
Prusiner, S.B., 1998. Prions. Proc. Natl. Acad. Sci. U.S.A. 95, 13363–13383.
Quílez, J., Sánchez-Acedo, C., del Cacho, E., Clavel, A., Causapé, A.C., 1996a. Prevalence of *Cryptosporidium* and *Giardia* infections in cattle in Aragón (northeastern Spain). Vet. Parasitol. 66, 139–146.
Quílez, J., Sánchez-Acedo, C., Clavel, A., del Cacho, E., López-Bernad, F., 1996b. Prevalence of *Cryptosporidium* infections in pigs in Aragón (northeastern Spain). Vet. Parasitol. 67, 83–88.
Rabølle, M., Spliid, N.H., 2000. Sorption and mobility of metronidazole, olaquindox, oxytetracycline and tylosin in soil. Chemosphere 40, 715–722.
Raina, P.S., Pollari, F.L., Teare, G.F., Goss, M.J., Barry, D.A.J., Wilson, J.B., 1999. The relationship between *E. coli* indicator bacteria in well-water and gastrointestinal illnesses in rural families. Can. J. Public Health 90, 172–175.
Rajic, A., Keenliside, J., McFall, M.E., Deckert, A.E., Mucklec, A.C., O'Connor, B.P., Manninen, K., Dewey, C.E., McEwen, S.A., 2005. Longitudinal study of *Salmonella* species in 90 Alberta swine finishing farms. Vet. Microbiol. 105, 47–56.
Ralston, B.J., McAllister, T.A., Olson, M.E., 2003. Prevalence and infection pattern of naturally acquired giardiasis and cryptosporidiosis in range beef calves and their dams. Vet. Parasitol. 114, 113–122.
Raman, D.R., Williams, E.L., Layton, A.C., Burns, R.T., Easter, J.P., Daugherty, A.S., Mullen, M.D., Sayler, G.S., 2004. Estrogen content of dairy and swine wastes. Environ. Sci. Technol. 38, 3567–3573.
Ramaswamy, J., Prasher, S.O., Patel, R.M., Hussain, S.A., Barrington, S.F., 2010. The effect of composting on the degradation of a veterinary pharmaceutical. Biores. Technol. 101, 2294–2299.
Ramchandani, M., Manges, A.R., DebRoy, C., Smith, C.P., Johnson, J.R., Riley, L.W., 2005. Possible animal origin of human-associated, multidrug-resistant, uropathogenic *Escherichia coli*. Clin. Infect. Dis. 40, 251–257.

Ramos, M.C., Martínez-Casasnovas, J.A., 2006. Erosion rates and nutrient losses affected by composted cattle manure application in vineyard soils of NE Spain. Catena 68, 177–185.

Ramos, M.C., Quinton, J.N., Tyrrel, S.F., 2006. Effects of cattle manure on erosion rates and runoff water pollution by faecal coliforms. J. Environ. Manage. 78, 97–101.

Raven, K.P., Hossner, L.R., 1993. Phosphorus desorption quantity-intensity relationships in soil. Soil Sci. Soc. Am. J. 57, 1501–1508.

Ravva, S., Korn, A., 2007. Extractable organic components and nutrients in wastewater from dairy lagoons influence the growth and survival of *Escherichia coli* O157:H7. Appl. Environ. Microbiol. 73, 2191–2198.

Reeves, P.G., Chaney, R.L., 2008. Bioavailability as an issue in risk assessment and management of food cadmium: a review. Sci. Total Environ. 398, 13–19.

Ridley, A.M., Morris, V.K., Cawthraw, S.A., Ellis-Iversen, J., Harris, J.A., Kennedy, E.M., Newell, D.G., Allen, V.M., 2011. Longitudinal molecular epidemiological study of thermophilic campylobacters on one conventional broiler chicken farm. Appl. Environ. Microbiol. 77, 98–107.

Ritter, W.F., Chirnside, A.E.M., Scarborough, R.W., 1990. Soil nitrate profiles under irrigation on Coastal Plain soils. J. Irrig. Drain. Eng. 116, 738–751.

Roberts, T.L., 2006. Improving nutrient use efficiency. Optimizing Resource Use Efficiency for Sustainable Intensification of Agriculture, Proceedings of the IFA Agriculture Conference 27 February – 2 March 2006, Kunming, China, International Fertilizer Industry Association (IFA), Paris, France, pp. 1–8.

Robertson, L.J., Campbell, A.T., Smith, H.V., 1992. Survival of *Cryptosporidium parvum* ocysts under various environmental pressures. Appl. Environ. Microbiol. 58, 3494–3500.

Robinson, J.S., Sharpley, A.N., Smith, S.J., 1995. The effect of animal manure applications on the forms of soil phosphorus. In: Steele, K. (Ed.), Animal Waste and Land-water Interface, CRC Press, Inc., Boca Rotan, Fl, USA, pp. 43–48.

Rogers, S.W., Donnelly, M., Peed, L., Kelty, C.A., Mondal, S., Zhong, Z., Shanks, O.C., 2011. Decay of bacterial pathogens, fecal indicators and real-time quantitative PCR genetic markers in manure-amended soils. Appl. Environ. Microbiol. 77, 4839–4848.

Ros, M., Klammer, S., Knapp, B., Aichberger, K., Insam, H., 2006. Long-term effects of compost amendment of soil on functional and structural diversity and microbial activity. Soil Use Manage. 22, 209–218.

Roszak, D.B., Colwell, R.R., 1987. Survival strategies of bacteria in the natural environment. FEMS Microbiol. Rev. 51, 365–379.

Rothrock, M.J., Frantz, J.M., Burnett, S., 2012. Effect of volumetric water content and clover (*Trifolium incarnatum*) on the survival of *Escherichia coli* O157:H7 in a soil matrix. Curr. Microbiol. 65, 272–283.

Rovira, A.D., 1995. Sustainable farming systems in the livestock areas of the Mediterranean region of Australia. In: Cook, H.F., Lee, H.C. (Eds.), Soil Management in Sustainable Agriculture. Proceedings of the Third International Conference on Sustainable Agriculture Wye College, University of London, 31 August to 4 September 1993, Wye College Press, Wye, Ashford, Kent, UK, pp. 12–30.

Roy, P., Dhillon, A.S., Lauerman, L.H., Schaberg, D.M., Bandli, D., Johnson, S., 2002. Results of *Salmonella* isolation from poultry products, poultry, poultry environment, and other characteristics. Avian Dis. 46, 17–24.

Rudolph, D., Goss, M.J., Graham, A., Kachanoski, G., Scafe, M., Aspinal, D., Vandenbroek, B., Clegg, S., Barry, D., Stimson, J., 1992. Ontario Farm Groundwater Quality Survey – Winter 1991/92. A Report Prepared for Agriculture Canada Agri-Food Development Branch, Guelph, Ontario, Canada.

Rüprich, A., 1994. "Field Studies on th Infiltration Capacity and the Viability of Fecal Bateria in the Soil after Slurry Application (in German)", Ph.D. Thesis. University of Hohenheim, Hohenheim, Germany.

Russell, E.J., 1946. A Student's Book on Soils and Manures. Cambridge University Press, Cambridge, UK.
Ryder, S., Dexter, G., Bellworthy, S., Tongue, S., 2004. Demonstration of lateral transmission of scrapie between sheep kept under natural conditions using lymphoid tissue biopsy. Res. Vet. Sci. 76, 211–217.
Sabourin, L., Beck, A., Duenk, P.W., Kleywegt, S., Lapen, D.R., Li, H., Metcalfe, C.D., Payne, M., Topp, E., 2009. Runoff of pharmaceuticals and personal care products following application of dewatered municipal biosolids to an agricultural field. Sci. Total Environ. 407, 4596–4604.
Safley, L.M., Baker, J.C., Westerman, P.W., 1992. Loss of nitrogen during sprinkler irrigation of swine lagoon liquid. Biores. Technol. 40, 7–15.
Samuelsen, O., 1989. Degradation of oxytetracycline in seawater at two different temperatures and light intensities, and the persistence of oxytetracycline in the sediment from a fish farm. Aquaculture 83, 7–16.
Satchell, J.E., Martin, K., 1984. Phosphatase activity in earthworm faeces. Soil Biol. Biochem. 16, 191–194.
Sato, K., Bartlett, P.C., Kaneene, J.B., Downes, F.P., 2004. Comparison of prevalence and antimicrobial susceptibilities of *Campylobacter* spp. isolates from organic and conventional dairy herds in Wisconsin. Appl. Environ. Microbiol. 70, 1442–1447.
Scarsbrook, C.E., 1965. Nitrogen availability. In: Bartholomew, W.V. (Ed.), Soil Nitrogen, In: Clark, F.E. (Ed.), Agronomy, vol. 10. American Society of Agronomy, Madison, Wisconsin, USA, pp. 481–502.
Schiffman, S.S., Williams, C.M., 2005. Science of odor as a potential health issue. J. Environ. Qual. 34, 129–138.
Schiffman, S.S., Sattely-Miller, E.A.S., Suggs, M.S., Graham, B.G., 1995. The effect of environmental odors emanating from commercial swine operations on the mood of nearby residents. Brain Res. Bull. 37, 369–375.
Schiffman, S.S., Walker, J.M., Dalton, P., Lorig, T.S., Raymer, J.H., Shusterman, D., Williams, C.M., 2000. Potential health effects of odor from animal operations, wastewater treatment, and recycling of byproducts. J. Agromedicine 7, 7–81.
Schlüsener, M.P., Bester, K., Spiteller, M., 2003. Determination of antibiotics such as macrolides, ionophores and tiamulin in liquid manure by HPLC-MS/MS. Anal. Bioanal. Chem. 374, 942–947.
Schröder, J., 2005. Revisiting the agronomic benefits of manure: a correct assessment and exploitation of its fertilizer value spares the environment. Biores. Technol. 96, 253–261.
Schuppers, M.E., Stephan, R., Ledergerber, U., Danuser, J., Bissig-Choisat, B., Stärk, K.D.C., Regula, G., 2005. Clinical herd health, farm management and antimicrobial resistance in *Campylobacter coli* on finishing pig farms in Switzerland. Prev. Vet. Med. 69, 189–202.
Scott, C.A., Smith, H.V., Gibbs, H.A., 1994. Excretion of *Cryptosporium parvum* oocysts by herd of beef suckler cows. Vet. Rec. 134, 172.
Scott, C.A., Smith, H.V., Mtambo, M.M.A., Gibbs, H.A., 1995. An epidemiological study of *Cryptosporidium parvurn* in two herds of adult beef cattle. Vet. Parasitol. 57, 227–288.
Scott, L., Menzies, P., Reid-Smith, R.J., Avery, B.P., McEwen, S.A., Moon, C.S., Berke, O., 2012. Antimicrobial resistance in *Campylobacter* spp. isolated from Ontario sheep flocks and associations between antimicrobial use and antimicrobial resistance. Zoonoses Public Health 59, 294–301.
Seaton, A., MacNee, W., Donaldson, K., Godden, D., 1995. Particulate air pollution and acute health effects. Lancet 345, 176–178.
Semenov, A.V., van Bruggen, A.H.C., van Overbeek, L., Termorshuizen, A.J., Semenov, A.M., 2007. Influence of temperature fluctuations on *Escherichia coli* 0157:H7 and *Salmonella enterica* serovar typhimurium in cow manure. FEMS Microbiol. Ecol. 60, 419–428.

Semenov, A.V., van Overbeek, L., van Bruggen, A.H.C., 2009. Percolation and survival of *Escherichia coli* 0157:H7 and *Salmonella enterica* serovar Typhimurium in soil amended with contaminated dairy manure or slurry. Appl. Environ. Microbiol. 75, 3206–3215.
Servos, M., Delorme, P., Fox, G., Sutcliffe, R., Wade, M., 2001. Endocrine disrupting substances. (Chapter 5) Threats to Sources of Drinking Water and Aquatic Ecosystem Health in Canada, Environment Canada, National Water Research Institute, Burlington, Ontario, Canada, pp. 18–21.
Sharpley, A.N., 1997. Rainfall frequency and nitrogen and phosphorus runoff from soil amended with poultry litter. J. Environ. Qual. 26, 1127–1132.
Sharpley, A.N., Smith, S.J., 1989. Mineralization and leaching of phosphorus from soil incubated with surface-applied and incorporated crop residue. J. Environ. Qual. 18, 101–105.
Sharpley, A.N., Withers, P.J.A., 1994. The environmentally-sound management of agricultural phosphorus. Fert. Res. 39, 133–146.
Shen, Y., Ren, L., Li, G., Chen, T., Guo, R., 2011. Influence of aeration on CH_4, N_2O and NH_3 emissions during aerobic composting of a chicken manure and high C/N waste mixture. Waste Manage. 31, 33–38.
Sheppard, S.C., Grant, C.A., Sheppard, M.I., de Jong, R., Long, J., 2009. Risk indicator for agricultural inputs of trace elements to Canadian soils. J. Environ. Qual. 38, 919–932.
Shere, J.A., Bartlett, K.J., Kaspar, C.W., 1998. Longitudinal study of *Escherichia coli* O157:H7 dissemination on four dairy farms in Wisconsin. Appl. Environ. Microbiol. 64, 1390–1399.
Shore, L.S., Shemesh, M., Cohen, R., 1988. The role of oestrodiol and oestrone in chicken manure silage in hyperoestrogenism in cattle. Aust. Vet. J. 65, 68.
Shuman, L.M., 1988. Effect of organic matter on the distribution of manganese, copper, iron, and zinc in soil fractions. Soil Sci. 146, 192–198.
Sidjabat-Tambunan, H., Bensink, J.C., 1997. Verotoxin-producing *Escherichia coli* from the faeces of sheep, calves and pigs. Aust. Vet. J. l75, 292–293.
Simonetti, G., Francioso, O., Nardi, S., Berti, A., Brugnolid, E., Morari, E.L.F., 2011. Characterization of humic carbon in soil aggregates in a long-term experiment with manure and mineral fertilization. Soil Sci. Soc. Am. J. 76, 880–890.
Sims, J.T., Simard, R.R., Joern, B.C., 1998. Phosphorus loss in agricultural drainage: historical perspective and current research. J. Environ. Qual. 27, 277–293.
Singh, R.P., Agrawal, M., 2008. Potential benefits and risks of land application of sewage sludge. Waste Manage. 28, 347–358.
Sinton, L.W., Finlay, R.K., Lynch, A.P., 1999. Sunlight inactivation of fecal bacteriophages and bacteria in sewage-polluted seawater. Appl. Environ. Microbiol. 65, 3605–3613.
Sistani, K.R., Torbert, H.A., Way, T.R., Bolster, C.H., Pote, D.H., Warren, J.G., 2009. Broiler litter application method and runoff timing effects on nutrient and *Escherichia coli* losses from tall fescue pasture. J. Environ. Qual. 38, 1216–1223.
Skovgaard, N., Nørrung, B., 1989. The incidence of Listeria spp. in faeces of Danish pigs and in minced pork meat. Int. J. Food Microbiol. 8, 59–63.
Smith, S.R., Woods, V., Evans, T.D., 1998a. Nitrate dynamics in biosolids-treated soils. III. Significance of the organic nitrogen, a twin-pool exponential model for nitrogen management and comparison with the nitrate production from animal wastes. Biores. Technol. 66, 161–174.
Smith, S.R., Woods, V., Evans, T.D., 1998b. Nitrate dynamics in biosolids-treated soils. I. Influence of biosolids type and soil type. Biores. Technol. 66, 139–149.
Smith, K.A., Jackson, D.R., Pepper, T.J., 2001a. Nutrient losses by surface run-off following the application of organic manures to arable land. 1. Nitrogen. Environ. Pollut. 112, 41–51.
Smith, K.A., Jackson, D.R., Withers, P.J.A., 2001b. Nutrient losses by surface run-off following the application of organic manures to arable land. 2. Phosphorus. Environ. Pollut. 112, 53–60.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M., Smith, J., 2008. Greenhouse gas mitigation in agriculture. Philos. Trans. R. Soc. B 363, 789–813.

Smolders, E., Oorts, K., Lombi, E., Schoeters, I., Ma, Y., Zrna, Z., McLaughlin, M.J., 2012. The availability of copper in soils historically amended with sewage sludge, manure and compost. J. Environ. Qual. 41, 506–514.

Sobsey, M.D., 1983. Transport and fate of viruses in soils. (Publication no. 600/9-83-017) In: Canter, L.W. (Ed.), Microbial Health Considerations of Soil Disposal of Domestic Wastewaters, U.S. Environmental Protection Agency, Cincinnati, Ohio, USA, pp. 175–197.

Solomon, E.B., Yaron, S., Matthews, K.R., 2002. Transmission of *Escherichia coli* O157:H7 from contaminated manure and irrigation water to lettuce plant tissue and its subsequent internalization. Appl. Environ. Microbiol. 68, 397–400.

Sommer, C., Steffansen, B., 1993. Changes with time after treatment in the concentrations of ivermectin in fresh cow dung and cow pats aged in the field. Vet. Parasitol. 48, 67–73.

Spiehs, M.J., Woodbury, B.L., Doran, B.E., Eigenberg, R.A., Kohl, K.D., 2010. Ammonia emissions from feedlot surface of beef deep-bedded monoslope facilities. International Symposium on Air Quality and Manure Management for Agriculture Conference Proceedings, 13–16 September 2010, Dallas, Texas, American Society of Agricultural and Biological Engineers (ASABE), St. Joseph, Michigan, USA.

St. Jean, R., 1997. On-farm Manure Composting Techniques: Understanding Nitrogen and Carbon Conservation. Research Report 1.3. COESA Report No.: RES/MAN-003/97. Prepared for Agriculture Canada, London Research Centre, London, Ontario Prepared by Ecologistics Limited, Waterloo, Ontario Available at: http://gis.lrs.uoguelph.ca/AgriEnvArchives/gp/gpres/download/rep1_3.pdf (Cited 27th March. 2010; verified 1st August 2012).

Stanford, K., Bach, S.J., Marx, T.H., Jones, S., Hansen, J.R., Wallins, G.L., Zahiroddini, H., McAllister, T.A., 2005. Monitoring *Escherichia coli* O157:H7 in inoculated and naturally colonized feedlot cattle and their environment. J. Food Prot. 68, 26–33.

Stanley, K.N., Wallace, J.S., Currie, J.E., Diggle, P.J., Jones, K., 1998a. The seasonal variation of thermophilic campylobacters in beef cattle, dairy cattle and calves. J. Appl. Microbiol. 85, 472–480.

Stanley, K.N., Wallace, J.S., Jones, K., 1998b. Thermophilic campylobacters in dairy slurries on Lancashire farms: seasonal effects of storage and land application. J. Appl. Microbiol. 85, 405–409.

Steenhof, M., Gosens, I., Strak, M., Godri, K.J., Hoek, G., Cassee, F.R., Mudway, I.S., Kelly, F.J., Harrison, R.M., Lebret, E., Brunekreef, B., Janssen, N.A.H., Pieters, R.H.H., 2011. Part. Fibre Toxicol. 8, 26 Available at: http://www.particleandfibretoxicology.com/content/8/1/26.

Steinbach, G., Blaha, T., Methner, U., 2002. Estimating the prevalence of Salmonella spp. in swine herds – influence of sensitivity and specificity of Salmonella detection. J. Vet. Med. Ser. B 49, 438–444.

Stelma, G.N., McCabe, L.J., 1992. Nonpoint pollution from animal sources and shellfish sanitation. J. Food Prot. 55, 649–656.

Stephenson, R.E., Chapman, H.D., 1931. Phosphate penetration in field soils. Agron. J. 23, 759–770.

Stevenson, F.J., 1982. Origin and distribution of nitrogen in soil. In: Stevenson, F.J. (Ed.), Nitrogen in Agricultural Soils, Agronomy, vol. 22. ASA-CSSA-SSSA, Madison, Wisconsin, USA, pp. 1–42.

Stevenson, F.J., 1991. Organic matter-micronutrient reactions in soil. In: Mortvedt, J.J. (Ed.), Micronutrients in Agriculture, second ed. Soil Science Society of America, Inc., Madison, Wisconsin, USA, pp. 145–186.

Stewart, I., Schluter, P.J., Shaw, G.R., 2006. Cyanobacterial lipopolysaccharides and human health – a review. Environ. Health 5, 7.
Stomberg, A.L., Hemphill, D.D., Volk, V.V., 1984. Elemental concentration of dweet corn grown on tannery waste-amended soil. J. Environ. Qual. 13, 162–166.
Strauch, D., 1987. Hygiene of animal waste management. In: Strauch, D. (Ed.), Animal Production and Environmental Health, Elsevier Science Publisher B.V., Amsterdam, Netherlands, pp. 155–202.
Strauch, D., 1988. Pathogenic disease organisms in feces and their epidemiological significance. Tierarztl. Prax. Suppl. 3, 21–27 (in German).
Stumpe, B., Marschner, B., 2007. Long-term sewage sludge application and wastewater irrigation on the mineralization and sorption of 17β-estradiol and testosterone in soils. Sci. Total Environ. 374, 282–291.
Stumpe, B., Marschner, B., 2010a. Organic waste effects on the behavior of 17β-estradiol, estrone, and 17α-ethinylestradiol in agricultural soils in long- and short-term setups. J. Environ. Qual. 39, 907–916.
Stumpe, B., Marschner, B., 2010b. Dissolved organic carbon from sewage sludge and manure can affect estrogen sorption and mineralization in soils. Environ. Pollut. 158, 148–154.
Stutter, M.I., Langan, S.J., Lumsdon, D.G., 2009. Vegetated buffer strips can lead to increased release of phosphorus to waters: a biogeochemical assessment of the mechanisms. Environ. Sci. Technol. 43, 1858–1863.
Sukul, P., Lamshöft, M., Kusari, S., Zühlke, S., Spiteller, M., 2009. Metabolism and excretion kinetics of ^{14}C-labeled and non-labeled difloxacin in pigs after oral administration, and antimicrobial activity of manure containing difloxacin and its metabolites. Environ. Res. 109, 225–231.
Sultani, M.I., Gill, M.A., Anwar, M.M., Athar, M., 2007. Evaluation of soil physical properties as influenced by various green manuring legumes and phosphorus fertilization under rain fed conditions. Int. J. Environ. Sci. Technol. 4, 109–118.
Suslow, T.V., 2003. Key Points of Control & Management for Microbial Food Safety: Growers & Packers. University of California Division of Agriculture and Natural Resources (UCDANR), Publication #8102. Available at: http://anrcatalog.ucdavis.edu/pdf/8102.pdf (Cited 23rd July 2012; verified 1st August, 2012).
Suslow, T., Oria, M., Beuchat, L., Garret, E., Parish, M., Harris, L., Farber, J., Busta, F., 2001. Analysis and Evaluation of Preventive Control Measures for the Control and Reduction/elimination of Microbial Hazards on Fresh and Fresh-cut Produce. Available at: http://www.fda.gov/Food/ScienceResearch/ResearchAreas/SafePracticesforFoodProcesses/ucm090977.htm (Cited 25th April 2011; verified 1st August, 2012).
Sutton, M.D., 1983. Phytopathogens and weed seeds in manure. Farm Animal Manures in the Canadian Environment, National Research Council of Canada Associate Committee on Scientific Criteria for Environmental Quality, Ottawa, Canada, pp. 109–114.
Søgaard, H.T., Sommer, S.G., Hutchings, N.J., Huijsmans, J.F.M., Bussink, D.W., Nicholson, F., 2002. Ammonia volatilization from field-applied animal slurry-the ALFAM model. Atmos. Environ. 36, 3309–3319.
Taiganides, E.P., 1987. Animal waste management and wastewater treatment. In: Strauch, D. (Ed.), Animal Production and Environmental Health, Elsevier Science Publisher, Amsterdam, Netherlands, pp. 91–154.
Tamgüney, G., Miller, M.W., Wolfe, L.L., Sirochman, T.M., Glidden, D.V., Palmer, C., Lemus, A., DeArmond, S.J., Prusiner, S.B., 2009. Asymptomatic deer excrete infectious prions in faeces. Nature 461, 529–533.
Tanner, B.D., Brooks, J.P., Gerba, C.P., Haas, C.N., Josephson, K.L., Pepper, I.L., 2008. Estimated occupational risk from bioaerosols generated during land application of Class B biosolids. J. Environ. Qual. 37, 2311–2321.

Tauxe, R.V., Vandepitte, J., Wauters, G., Martin, S.M., Goossens, V., De Mol, P., Van Noyen, R., Thiers, G., 1987. *Yersinia enterocolitica* infections and pork: the missing link. Lancet 330, 1129–1132.

Tejada, M., Dobao, M.M., Benitez, C., Gonzales, J.L., 2001. Study of composting of cotton residues. Biores. Technol. 79, 199–202.

Tejada, M., Hernandez, M.T., Garcia, C., 2006. Application of two organic amendments on soil restoration: effects on the soil biological properties. J. Environ. Qual. 35, 1010–1017.

Terry, R.E., Nelson, D.W., Sommers, L.E., 1981. Nitrogen transformations in sewage sludge-amended soils as affected by soil environmental factors. Soil Sci. Soc. Am. J. 45, 506–513.

Terry, L.A., Howells, L., Bishop, K., Baker, C.A., Everest, S., Thorne, L., Maddison, B.C., Gough, K.C., 2011. Detection of prions in the faeces of sheep naturally infected with classical scrapie. Vet. Res. 42, 65. http://dx.doi.org/10.1186/1297-9716-42-65.

Tester, C.F., Sikora, L.J., Taylor, J.M., Parr, J.F., 1977. Decomposition of sewage sludge compost in soil: 1. Carbon and nitrogen transformations. J. Environ. Qual. 6, 459–463.

Thacker, B., 2007. Management of byproduct solids generated in the pulp and paper industry. In: Presentation to EPA OSW Staff, Washington, DC, January 23, 2007. USAEPA, Washington DC, USA. pp. 18. Available at: http://www.epa.gov/wastes/conserve/imr/irc-meet/03-paper.pdf (Cited 12th Feb. 2010; verified 1st August, 2012).

Thelin, R., Gifford, G.F., 1983. Fecal coliform release patterns from fecal material of cattle. J. Environ. Qual. 12, 57–63.

Thiele-Bruhn, S., 2003. Pharmaceutical antibiotic compounds in soils – a review. J. Plant Nutr. Soil Sci. 166, 145–167.

Thomas, D., Heckrath, G., Brookes, P.C., 1997. Evidence of phosphorus movement from Broadbalk soils by preferential flow. In: Tunney, H. (Ed.), Phosphorus Loss from Soil to Water, CAB International, Wallingford, UK, pp. 369–370.

Thomas, R.J., El-Dessougi, H., Tubeileh, A., 2006. Soil system management under arid and semi-arid conditions. In: Uphoff, N. (Ed.), Biological Approaches to Sustainable Soil Systems, Taylor and Francis, Boca Raton, Florida, USA, pp. 41–55.

Thompson, R.B., Ryden, J.C., Lockyer, D.R., 1987. Fate of nitrogen in cattle slurry following surface application or injection to grassland. J. Soil Sci. 38, 689–700.

Thomsen, I.K., 2005. Crop N utilization and leaching losses as affected by time and method of application of farmyard manure. Eur. J. Agron. 22, 1–9.

Thu, K., Donham, K., Ziegenhorn, R., Reynolds, S., Thorne, P.S., Subramanian, P., Whitten, P., Stookesberry, J., 1997. A control study of the physical and mental health of residents living near a large-scale swine operation. J. Agric. Saf. Health 3, 13–26.

Toth, J.D., Fox, R.H., 1998. Nitrate losses from a core-alfalfa rotation: lysimeter measurement of nitrate leaching. J. Environ. Qual. 27, 1027–1033.

Turco, R.F., Brouder, S., 2004. Fate and transport of *E. coli* in Soil and Subsurface Settings: Swine Waste Management with the Context of TMDL's. Research Report Public Health/Worker Safety. National Pork Board, Des Moines, Iowa, USA.

Turner, B.L., Leytem, A.B., 2004. Phosphorus compounds in sequential extracts of animal manures: chemical speciation and a novel fractionation procedure. Environ. Sci. Technol. 38, 6101–6108.

UC-GAP, 2012. University of California (UC) Good Agricultural Practices (GAP). Website: http://ucfoodsafety.ucdavis.edu/UC_Publications/UC_Good_Agricultural_Practices_GAP/ (Cited July 23rd 2012; verified 1st August, 2012).

Ullrey, D.E., 1992. Basis for regulation of selenium supplements in animal diets. J. Anim. Sci. 70, 3922–3927.

Unc, A., Goss, M.J., 2006a. Culturable *Escherichia coli* in soil mixed with two types of manure. Soil Sci. Soc. Am. J. 70, 763–769.

Unc, A., Goss, M.J., 2006b. Impact of land application of organic wastes on soil hydraulic properties and on water partitioning. J. Environ. Eng. Sci. 5, 243–251.

Unc, A., Goss, M.J., Springthorpe, S., Sattar, S., 2004. Role of the surface charge properties for bacterial retention in soils; significance of soluble and colloidal organic matter in suspension. Proceedings of the 9th European Biosolids and Organic Residuals Conference, AquaEnviro, Wakefield, U.K, pp. 1–5(Paper 50).

Unc, A., Gardner, J., Springthorpe, A., 2006. Recovery of *Escherichia coli* from soil after addition of sterile organic wastes. Appl. Environ. Microbiol. 72, 2287–2289.

Unc, A., Goss, M.J., Cook, S., Li, X., Atwill, E.R., Harter, T., 2012. Analysis of matrix effects critical to microbial transport in organic waste-affected soils across laboratory and field scales. Water Resour. Res. 48, W00L12. http://dx.doi.org/10.1029/2011WR010775.

Unwin, R.J., 1998. Report of an enquiry of Network members into the status of metals in animal manures. In: Martinez, J., Maudet, M. (Eds.), RAMIRAN 98. Proc. 8th Int. Conference on the FAO ESCORENA Network on Recycling of Agricultural, Municipal, and Industrial Residues in Agriculture. Rennes, France, 26–29 May 1998, FAO and Cemagref, France, pp. 401–413.

Vaccari, F.P., Baronti, S., Lugato, E., Genesio, L., Castaldi, S., Fornasier, F., Miglietta, F., 2011. Biochar as a strategy to sequester carbon and increase yield in durum wheat. Eur. J. Agron. 34, 231–238.

Van Faassen, H.G., Van Dijk, H., 1987. Manure as a source of nitrogen and phosphorus in soils. In: Van der Meer, H.G., Unwin, R.J., van Dijk, T.A., Ennik, G.C. (Eds.), Animal Manure on Grassland and Fodder Crops. Fertilizer or Waste? Martinus Nijhoff Publishers, Dordrecht, Netherlands, pp. 27–45.

Van Hoorebeke, S., Van Immerseel, F., De Vylder, J., Ducatelle, R., Haesebrouck, F., Pasmans, F., de Kruif, A., Dewulf, J., 2009. Faecal sampling underestimates the actual prevalence of Salmonella in laying hen flocks. Zoonoses Public Health 56, 471–476.

van Keulen, L.J.M., Bossers, A., van Zijderveld, F., 2008. TSE pathogenesis in cattle and sheep. Vet. Res. 39, 24. http://dx.doi.org/10.1051/vetres:2007061.

van Veen, J.A., Van Overbeek, L.S., Van Elsas, J.D., 1997. Fate and activity of microorganisms introduced into soil. Microbiol. Mol. Biol. Rev. 61, 121–135.

Vanselow, B.A., Hum, S., Hornitzky, M.A., Eamens, G.J., Quinn, K., 2007. *Salmonella typhimurium* persistence in a Hunter Valley dairy herd. Aust. Vet. J. 85, 446–450.

Varela, N.P., Friendship, R.M., Dewey, C.E., 2007. Prevalence of *Campylobacter* spp. isolated from grower-finisher pigs in Ontario. Can. Vet. J. 48, 515–517.

Vinten, A.J.A., 1999. Predicting nitrate leaching from drained arable soils derived from glacial till. J. Environ. Qual. 28, 988–996.

Vogtmann, H., Besson, J.M., 1978. European composting methods: treatment and use of farmyard manure and slurry. Compost Sci./Land Util. 19, 1–23.

Wade, S.E., Mohammed, H.O., Schaaf, S.L., 2000. Prevalence of *Giardia* sp., *Cryptosporidium parvum* and *Cryptosporidium muris* (*C. andersonii*) in 109 dairy herds in five countries of southeastern New York. Vet. Parasitol. 93, 1–11.

Walker, J.T., 1996. Crambe and rapeseed meal as soil amendments: nematicidal potential and phytotoxic effects. Crop Prot. 15 (5), 433–437.

Walter, D.A., Rea, B.A., Stollenwerk, K.G., Savoie, J., 1996. Geochemical and Hydrologic Controls of Phosphorus Transport in a Sewage-Contaminated Sand and Gravel Aquifer Near Ashumet Pond, Cape Cod, Massachusetts. United States Geological Survey. Water-supply paper 2463 United States Government Printing Office, Washington, DC, USA.

Walter, W.D., Walsh, D.P., Farnsworth, M.L., Winkelman, D.L., Miller, M.W., 2011. Soil clay content underlies prion infection odds. Nat. Commun. 2, 200. http://dx.doi.org/10.1038/ncomms12003.

Wang, G., Zhao, T., Doyle, M.P., 1996. Fate of enterohemorrhagic *Escherichia coli* O157:H7 in bovine feces. Appl. Environ. Microbiol. 62, 2567–2570.

Ward, M.H., deKok, T.M., Levallois, P., Brender, J., Gulis, G., Nolan, B.T., VanDerslice, J., 2005. Workgroup report: drinking-water nitrate and health – recent findings and research needs. Environ. Health Perspect. 113, 1607–1614.

Warman, P.R., 1991. Effect of incorporated green manure crops on subsequent oat production in an acid, infertile silt loam. Plant Soil 134, 115–119.

Warnick, L.D., Kaneene, J.B., Ruegg, P.L., Wells, S.J., Fossler, C., Halbert, L., Campbell, A., 2003. Evaluation of herd sampling for Salmonella isolation on midwest and northeast US dairy farms. Prev. Vet. Med. 60, 195–206.

Warriner, K., Ibrahim, F., Dickinson, M., Wright, C., Waites, W.M., 2003. Interaction of *Escherichia coli* with growing salad spinach plants. J. Food Prot. 66, 1790–1797.

Wellenius, G.A., Burger, M.R., Coull, B.A., Schwartz, J., Suh, H.H., Koutrakis, P., Schlaug, G., Gold, D.R., Mittleman, M.A., 2012. Ambient air pollution and the risk of acute ischemic stroke. Arch. Intern. Med. 172, 229–234.

Wells, J.E., Varel, V.H., 2008. Viability of zoonotic pathogens *Escherichia coli* and *Salmonella* in swine manure slurries with and without a urease inhibitor and thymol. Lett. Appl. Microbiol. 46, 477–482.

Wells, J.G., Shipman, L.D., Greene, K.D., Sowers, E.G., Green, J.H., Cameron, D.N., Downes, F.P., Martin, M.L., Griffin, P.M., Ostroff, S.M., Potter, M.E., Tauxe, R.V., Wachsmuth, I.K., 1991. Isolation of *Escherichia coli* serotype O157:H7 and other shiga-like-toxin-producing *E. coli* from dairy cattle. J. Clin. Microbiol. 29, 985–989.

Wesseling, J.G., Stoof, C.R., Ritsema, C.J., Oostindie, K., Dekker, L.W., 2009. The effect of soil texture and organic amendment on the hydrological behaviour of coarse-textured soils. Soil Use Manage. 25, 274–283.

Weuve, J., Puett, R.C., Schwartz, J., Yanosky, J.D., Laden, F., Grodstein, F., 2012. Exposure to particulate air pollution and cognitive decline in older women. Arch. Intern. Med. 172, 219–227.

WHO (World Health Organization), 1984. Guidelines for Drinking-water Quality. Health Criteria and Other Supporting Information. vol. 2. World Health Organization, Geneva, Switzerland, pp. 128–134.

WHO, 2006. Air Quality Guidelines for Particulate Matter, Ozone, Nitrogen Dioxide and Sulfur Dioxide. Global Update 2005. Summary of Risk Assessment. Available at: http://www.euro.who.int/__data/assets/pdf_file/0005/78638/E90038.pdf (Cited 25th February 2012; verified 1st August, 2012).

WHO, 2011. Air Quality and Health. Fact Sheet 313. Available at: http://www.who.int/mediacentre/factsheets/fs313/en/index.html (Cited 25th February 2012; verified 1st August, 2012).

Wild, A., 1988. Historical. In: Wild, A. (Ed.), Russell's Soil Conditions and Plant Growth, eleventh ed. Longman Scientific & Technical, Harlow, Essex, UK, pp. 1–30.

Wilkes, G., Edge, T., Gannon, V., Jokinen, C., Lyautey, E., Medeiros, D., Neumann, N., Rueckerg, N., Topp, E., Lapen, D.R., 2009. Seasonal relationships among indicator bacteria, pathogenic bacteria, *Cryptosporidium* oocysts, *Giardia* cysts, and hydrological indices for surface waters within an agricultural landscape. Water Res. 43, 2209–2223.

Winckler, C., Grafe, A., 2001. Use of veterinary drugs in intensive animal production. Evidence for persistence of tetracycline in pig slurry. J. Soils Sediments 1, 66–70.

Winkworth, C.L., Matthaei, C.D., Townsend, C.R., 2008. Prevalence of *Giardia* and *Cryptosporidium* spp in calves from a region in New Zealand experiencing intensification of dairying. N. Z. Vet. J. 56, 15–20.

Wirsenius, S., Azar, C., Berndes, G., 2010. How much land is needed for global food production under scenarios of dietary changes and livestock productivity increases in 2030? Agr. Sys. 103, 621–638.

Xiao, L., Herd, R.P., Bowman, G.L., 1994. Prevalence of *Cryptosporidium* and *Giardia* infections on two Ohio pig farms with different management systems. Vet. Parasitol. 52, 331–336.

Xuan, R., Blassengale, A.A., Wang, Q., 2008. Degradation of estrogenic hormones in a silt loam soil. J. Agric. Food Chem. 56, 9152–9158.

Yates, M.V., Yates, S.R., 1988. Modeling microbial fate in the subsurface environment. CRC Crit. Rev. Environ. Control 17, 307–344.
Yazaki, Y., Mizuo, H., Takahashi, M., Nishizawa, T., Sasaki, N., Gotanda, Y., Okamoto, H., 2003. Sporadic acute or fulminant hepatitis E in Hokkaido, Japan, may be food-borne, as suggested by the presence of hepatitis E virus in pig liver as food. J. Gen. Virol. 84, 2351–2357.
Yeager, J.G., O'Brien, R.T., 1979. Enterovirus inactivation in soil. Appl. Environ. Microbiol. 38, 694–701.
Younie, M.F., Burton, D.L., Kachanoski, R.G., Beauchamp, E.G., Gilham, R.W., 1996. "Impact of Livestock Manure and Fertilizer Application on Nitrate Contamination of Groundwater." Final report for the Ontario Ministry of Environment and Energy. RAC No. 488G.
Zhang, G., Ma, L., Beuchat, L.R., Erickson, M.C., Phelan, V.H., Doyle, M.P., 2009. Lack of internalization of *Escherichia coli* O157:H7 in lettuce (*Lactuca sativa* L.) after leaf surface and soil inoculation. J. Food Prot. 72, 2028–2037.
Zhao, T., Doyle, M.P., Shere, J., Garber, L., 1995. Prevalence of enterohemorrhagic *Escherichia coli* O157:H7 in a survey of dairy herds. Appl. Environ. Microbiol. 61, 1290–1293.
Zheng, W., Yates, S.R., Bradford, S.A., 2008. Analysis of steroid hormones in a typical dairy waste disposal system. Environ. Sci. Technol. 42, 530–535.