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Production and Feeding Strategies for Phosphorus Management on Dairy Farms in New York

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Abstract.

Whole-farm simulation was used to evaluate the long-term effects of changes in feeding, cropping, and other production strategies on phosphorus loading and the economics of two farms in southeastern New York. Reducing the level of dietary P fed and maximizing the use of farm-grown forage provided a long-term P balance along with an increase in farm profitability.

Keywords. Farm system, Nutrient management, Phosphorus, Simulation, Economics

Introduction

On many watersheds across our country, phosphorus (P) loading from agricultural land to water is becoming an important concern. The Catskill and Delaware watersheds in eastern New York are a prime example. These watersheds, which are primarily covered with forests and dairy farms, supply 90% of the 1.3 billion gallons of water required each day for New York City (NRC, 2000). This water supply is processed using chlorine disinfection without filtration so natural purity is important. Regional and national health concerns however, have led state officials to address the need for water filtration. Also, the formation of carcinogens during chlorination of water containing algae (*Cyanobacteria*) has challenged the wisdom of this costly treatment for drinking water (Kotak et al., 1993). New York City would prefer to manage the watersheds for reduced pollutant loading instead of investing in the filtration facility and other potential treatments required to clean up the water.

Because cropland can provide a natural filter that supplies clean water, agriculture is the preferred land use for these watersheds (NRC, 2000). Agriculture must also do its part though, to maintain low pollutant loading. A Watershed Agricultural Program has been established to encourage the voluntary participation of farmers in developing whole-farm management plans that reduce pollutant loading. A portion of the Catskill/Delaware watershed has been designated as P restricted, thus management practices are being implemented or considered to reduce P loads on farms in this area.

Many factors affect P loss from dairy farms. These include the amount of feed and fertilizer brought onto a farm, number of animals per unit of land, feeding strategies, manure application procedures, crops grown, and topographic features of the landscape. Over the last 20 years, intensification and expansion of dairy farms has increased the tendency for farm-scale surpluses of P, where inputs in feed and fertilizer are greater than those exported in produce (Haygarth et al., 1998; Lander et al., 1998; Withers et al., 1999). These surpluses increase the potential for P enrichment of runoff (Kellogg and Lander, 1999; Sims et al., 1998). For example, the land application of dairy manure in the LaPlatte River Basin, Vermont, (8,832 ha) was identified as an important source of P to Lake Champlain (2.2 kg P ha^{-1} yr^{-1}; Meals, 1990). Also, McFarland and Hauck (1999) found streamflow P in subwatersheds of the upper North Bosque River Basin, TX, was most closely correlated with the percentage of land used for dairy manure application (r^2 of 0.82 for dissolved and 0.77 for total P) and cow density (r^2 of 0.70 for dissolved and 0.64 for total P) in each subwatershed.

Target implementation of best management practices (BMPs) on critical areas for P loss can decrease watershed export of P (Sharpley and Rekolainen, 1997). In the LaPlatte River Basin, implementation of BMPs (control of barnyard runoff, milkhouse waste treatment, and use of waste storage facilities) decreased P export to Lake Champlain when more than 90% of the animals were under BMPs (Meals, 1990). Studies have also demonstrated that changes in crop rotations (Sugiharto et al., 1994), manure-handling procedures (VanDyke et al., 1999), and feeding practices (Satter, 2001; Withers et al., 1999) can reduce surplus P on dairy farms, and often this can be done while maintaining or improving farm profit.

An analysis is needed to evaluate and compare the environmental and economic impacts of various management practices on dairy farms in the New York City watersheds. Such long-term effects are best evaluated through the use of computer simulation. By simulating farms over many weather years, the effects of management changes can be predicted quickly with little cost or risk to the farm owner. The Dairy Forage System Model (DAFOSYM) provides a tool for this type of assessment of dairy farms (Rotz et al., 1999b). This model has been widely used to evaluate forage conservation (Rotz et al., 1993), manure handling (Harrigan et al., 1996),

cropping system (Rotz et al., 2001), grazing (Soder et al., 2001) and feeding (Rotz et al., 1999b) options for various sizes and types of dairy farms in various locations.

Objective

To compare the long term environmental and economic impacts of various management changes on dairy farms in the Catskill region of southeastern New York to determine the most effective changes that can be made by dairy producers to reduce the potential loss of P to the watershed while maintaining or improving farm profitability.

Procedure

Model Description

DAFOSYM is a whole-farm model where crop production, feed use, and the return of manure nutrients back to the land are simulated over many years of weather (Rotz et al., 1989; Harrigan et al., 1996; Rotz et al., 2001). Growth and development of alfalfa, grass, corn, soybean, and small grain crops are predicted on a daily time step based on soil and weather conditions. Tillage, planting, harvest, and storage operations are simulated to predict resource use, timeliness of operations, crop losses, and nutritive changes in feeds. Feed allocation and animal response are related to the nutritive value of available feeds and the nutrient requirements of six animal groups making up the dairy herd (Rotz et al., 1999a).

Nutrient flows through the farm are modeled to predict potential nutrient accumulation in soil and loss to the environment (Rotz et al., 1999b). The quantity and nutrient content of the manure produced is a function of the quantity and nutrient content of the feeds consumed. Nitrogen (N) volatilization occurs in the barn, during storage, and between field application and incorporation into the soil (Borton et al., 1995). Denitrification and leaching losses from the soil are controlled by the rate of moisture movement and drainage from the soil profile as influenced by soil properties, rainfall, and the amount and timing of manure and fertilizer applications using functions from the NLEAP model (Schaffer et al., 1991).

A whole-farm balance of N, P and potassium (K) is determined that considers the import of nutrients in feed and fertilizer and the export in milk and animals. Phosphorus and K intakes are set as the greater of the sum of that contained in feeds or the requirement of the animal group. For lactating animals, P supplementation above the quantities contained in feeds is often required; thus, P intake is normally based upon animal requirements. Concentrations of the three nutrients contained in milk and live body weight were set as 0.53% N, 0.09% P, and 0.15% K for milk and 2.75% N, 0.79% P, and 0.20% K for live weight. Nutrient removals in each crop and the nutrients contained in feeds were determined using simulated crop yields and assigned or predicted nutrient concentrations (Table 1). Phosphorus and K losses from the farm were set at 5% of that applied in manure and fertilizer to represent normal losses that would occur through runoff. Excess P and K accumulated in the soil where the accumulation was the difference between the total export and import of each mineral divided by the total farm area. This whole-farm balance assumed that, over the long term, these nutrients were uniformly distributed over all available land.

Simulated performance was used to predict production costs, income, and net return, or profit, of the farm for each weather year. A whole-farm budget was used where investments in equipment and structures were depreciated over their useful life and annual expenditures and incomes were accounted. Possible government subsidies and income tax implications were not considered. By modeling several alternatives, the effects of system changes were compared

Table 1. Nutrient concentrations in feeds produced or purchased for the farms.

	N	P	K
	% DM		
Pasture	3.2-4.2*	0.35	3.00
Alfalfa hay and silage	3.0-3.8	0.30	2.50
Grass hay or silage	1.3-2.1	0.30	2.50
Corn, oat or triticale silage	1.3-1.6	0.26	1.18
High moisture ear corn	1.4	0.34	0.31
Feed mix (smaller farm)	3.2	0.60	1.21
Corn grain or corn meal	1.6	0.30	0.42
Soybean meal	8.8	0.70	2.41
Cotton seed	3.7	0.60	1.13
Distiller's grain	3.7	0.83	1.10

* Protein (N) concentrations in forage crops are related to growing and harvest conditions

including resource use, production efficiency, environmental impact, and net return. The distribution of annual values obtained was used to assess the risk involved in alternative technologies or strategies as weather conditions varied.

Farm Descriptions

Two farms were modeled representing actual farms in southeastern New York. The first was a typical farm for the region, having about 100 cows, while the other modeled one of the larger farms that included 800 cows. Farms were simulated for 25 weather years using historical weather data (1976-2000) from Cooperstown, NY. The typical soil in this region is a Lewbeach (shallow silt loam) soil with an available water holding capacity of about 10 cm.

The smaller farm consisted of 142 ha of owned land and 40 ha of rented land. Crops included 40 ha of alfalfa, 109 ha of grass, and 32 ha of corn. Alfalfa was seeded with an oat cover crop and maintained with a four-year stand life. Sixty-five percent of the manure was applied to the corn crop with the remainder applied to grassland. Additional fertilizer application to the corn land included 105, 22, and 45 kg/ha of N, phosphate (P_2O_5), and potash (K_2O), respectively. Grass received 100 kg/ha of N and 135 kg/ha of potash. Alfalfa received 135 kg/ha of potash, but when established with an oat cover crop, the rates were 34, 17, and 100 kg/ha for N, phosphate, and potash fertilizers. Simulated crop yields are summarized in Table 2. Yields for rotationally grazed pastures and less intensively managed pastures were adjusted to provide pasture consumption representative of that reported for this region (Fox et al., 1992).

Alfalfa was harvested using a three-cutting strategy where the first and third cuttings were harvested as wilted silage, and the second cutting was predominantly harvested as dry hay. About 50 ha of the grassland were grazed, with the remainder harvested as dry hay using a two-cutting strategy. Higher quality grass hay was fed, but some of the lower quality hay was used as bedding. Most of the corn was harvested as silage, but in high-yielding years, a portion was harvested and fed as high moisture ear corn. The oat cover crop was also harvested as silage. All operations were performed with equipment owned by the farmer (Table 3).

Table 2. Mean and range in annual harvested crop and available pasture yields over 25 yr simulations for southeastern New York.

Crop	Mean	Low	High	CV
	----------	t DM/ha	----------	%
Alfalfa (3 cuttings)	6.7	5.1	8.2	14.3
Alfalfa (2 cuttings)	4.4	3.2	5.0	10.0
Corn silage	12.2	6.5	16.9	24.7
High-moisture ear corn*	6.8	4.7	9.0	20.5
Oat or triticale silage	5.1	3.1	6.5	15.0
Grass hay or silage	4.3	3.1	5.3	13.9
Pasture with rotational grazing	6.9	6.5	7.3	3.6
Other pasture	4.5	4.1	4.7	3.8

*For only a few weather years when high-moisture corn is produced.

Storage facilities included a barn for hay storage, a bunker silo and three concrete stave tower silos (Table 3). Most of the corn and oat silage was stored in the bunker silo along with about half of the alfalfa silage. The largest tower silo was used for corn silage, the mid-sized silo was used for alfalfa silage, and the smallest was used for high moisture ear corn or corn silage.

The herd consisted of 105 large-framed Holstein cows, 60 heifers over one year old, and 45 younger heifers. Annual milk production was 9,500 kg/cow with 30% of the herd replaced each year. Remaining heifers were sold as bred, high-producing stock at an average price of $1,650 each. Cows were housed, fed, and milked in a stanchion barn (Table 3). Heifers were in pens during half of the year and on pasture during the growing season. Cows were fed a high-forage diet supplemented with a grain mix. The mix included primarily corn meal, distiller's grain, rolled corn, wheat bran, whole cottonseed, and roasted soybeans along with minerals and vitamins.

Manure was handled in a solid form (20% DM) using gutter cleaners, a front-end loader, and box spreader. There was some short-term storage, but essentially the manure was spread on a daily basis. The manure remained on the field surface following spreading where much of the ammonia N was volatilized and lost to the atmosphere.

The large farm consisted of 2,025 ha of crop and grassland on the Lewbeach soil. Crops included 162 ha each of alfalfa and corn and 1,700 ha of grassland. Alfalfa was established with a cover crop of triticale. The grassland consisted of 373 ha of seeded grasses including 117 ha of Reed Canary grass with the remainder in other pasture grasses. About 150 ha of the pasture grass was rotationally grazed with lactating cows, and the remainder was used less intensively by heifers and dry cows. Pasture management required about 35 h per week of labor.

All alfalfa and about 360 ha of grass were each harvested with a two-cutting strategy. First cutting alfalfa was predominantly harvested as dry hay and the second cutting of alfalfa and both cuttings of grass were predominantly harvested and stored as wilted silage. Following these harvests, this land was available for grazing. All corn and the triticale were harvested as silage. All silage was packed and stored on a large concrete pad where movable walls were used to separate silage types. Silage was packed with two large tractors to a depth of as much as 12 m. Any hay harvested was stored outdoors.

There were 800 cows and 600 replacement heifers on the farm. Milking animals were divided into three herds for housing and milking. Two of these herds were housed in free stall barns and milked in double-eight herringbone parlors. The third group, which was primarily older animals, was milked in a tie stall barn with a pipeline milking system. The average annual milk production level was 8,170 kg/cow. Mixed rations were fed using mobile mixing wagons. Forages produced on the farm were supplemented with purchased corn meal, soybean meal, whole cottonseed, distiller's grain, and minerals.

Table 3. Machines and structures used to represent those of the two New York dairy farms.

	Small (100-cow) farm			Large (800-cow) farm		
Machine or facility	Size/type	No.	Initial Cost ($)	Size/type	No.	Initial Cost ($)
Tractors	35 kW, used	1	10,000	65 kW	2	34,800
	71 kW	1	34,800	74 kW	2	53,000
	112 kW	1	67,900	82 kW	2	58,000
Skid steer loader	25 kW	1	20,700	35 kW	2	31,500
Mower-conditioner	3.7 m, disk	1	23,900	4.0 m, disk	1	23,900
Hay rake	4.0 m	1	9,000	11.0 m	1	12,600
Baler	Med., 8 t DM h^{-1}	1	14,400	---	--	---
Bale wagons	4.5 t	5	3,100	---	--	---
Forage harvester	Med., 12 t DM h^{-1}	1	27,900	SP, 25 t DM h^{-1}	2	197,400
Forage hauling	wagons, 6 t	3	11,900	dump trucks, used	5	30,000
Feed mixer wagon	---	-	---	medium, 8.5 t	2	18,900
Manure spreader	8 t	1	8,100	tank trucks, used	3	30,000
Moldboard plow	2.3 m	1	13,000	2.3 m	2	13,000
Tandem disk harrow	3.7 m	1	7,600	3.7 m	2	7,600
Seedbed conditioner	6.0 m	1	18,500	---	-	---
Corn planter	4 row	1	14,000	6 row	2	18,000
Grain drill	2.4 m	1	7,200	3.7 m	1	15,300
Hay shed	100 t	1	10,000	100 t	1	10,000
Silage bunker	9.8 x 32 x 2.4 m	1	27,700	30.5 x 46 x 9.1 m	1	60,000
Tower silo	6.1 x 18.3 m	1	19,400	---	-	---
Tower silo	4.9 x 15.2 m	1	12,800	---	-	---
Tower silo	4.3 x 10.7 m	1	9,200	---	-	---
Manure storage	---	-	---	lagoon, 88 x 3 m	-	115,000
Machinery shed	---	1	60,000	---	-	150,000
Milking center	pipeline	1	126,000	double eight parlor	-	512,000
Cow housing	stanchion barn	1	158,000	free stall barns	-	800,000
Heifer & calf housing	pens	1	68,000	freestall & hutch	-	375,000
Feed storage	bin	1	3,200	commodity shed	-	56,000
Pasture fence, etc.	---	-	11,000	---	-	27,000

Table 4. Economic parameters and prices assumed for various system inputs and outputs for the analysis of the New York dairy farms. Prices were set to represent long-term relative prices in current value, which were not necessarily current prices.

Parameter	Value	Parameter	Value
Diesel fuel price	$ 0.29 /L	Selling price of feeds/animals	
Electricity price	$ 0.08 /kW-h	- Grass hay	$ 66 /t DM
Mailbox milk price	$ 30 /hL	- Grain crop silage	$ 75 /t DM
Total of livestock expenses	$ 238 /cow/yr	- Calf	$ 20 /animal
Land rental charge	$ 25 /ha	- Cull cow	$ 0.88 /kg
Property tax rate	5.0 %/yr	- Heifer	$1200 /animal*
Fertilizer prices		Buying price of feeds/bedding	
- Nitrogen	$ 0.55 /kg	- Alfalfa hay	$ 135 /t DM
- Phosphorus	$ 0.66 /kg	- Grain mix	$ 187 /t DM
- Potassium	$ 0.29 /kg	- Minerals/vitamins	$ 550 /t DM
Annual cost of seed and chemicals		- Bedding	$ 66 /t DM
- New alfalfa or grass	$ 200 /ha	Economic life	
- Established alfalfa or grass	$ 15 /ha	- Structures	20 yr
- Corn following corn	$ 165 /ha	- Machinery	10 yr
- Corn following other crop	$ 135 /ha	Salvage value	
- Oat or triticale	$ 56 /ha	- Structures	0 %
- Pasture	$ 0 /ha	- Machinery	30 %
		Real interest rate	6.0 %/yr

* Due to higher genetic value, bred heifers were sold for $1,650 from the smaller farm.

Most of the manure was handled as slurry that was stored in lagoons. This manure was normally applied to cropland in the spring and fall where it was incorporated within a week of application. Remaining manure was hauled on a daily basis where rapid incorporation was practiced. Most of the bedding was sawdust imported onto the farm at a rate of about one tonne per day.

The economic analysis of the farm systems assumed new equipment and facilities depreciated over an appropriate amount of time (Tables 3 and 4). Prices on feeds, milk, and other farm inputs and outputs were set to reflect long-term relative values in current dollars. Prices were held constant across simulated years so that economic differences across years were solely due to weather effects on farm performance. A real interest rate (approximately nominal rate minus inflation) of 6%/yr was assumed on investments. For the small farm, a labor cost was not included since the family management unit supplied all labor. On the large farm, the labor cost was fixed at the current actual cost of $600,000 per year.

The costs of production and the net returns predicted by the model were average annual values determined under the price and economic parameters assumed for these farms. These values represented the costs and returns for these production systems, but they were not the costs and profit of the actual farms.

Farm Level Verification

Evaluations were made to assure that the model adequately described the existing farms. The first step was a comparison of predicted yields to yields determined by the New York Agricultural Statistics Service (2000) for the county in which the farms were located. Predicted yields were set to long-term averages that were about 10% above the reported county yields for weather years 1990 to 2000. This level of yield represented above-average management, which was consistent with the producers' reported production.

Next, total feed production and use on the farm was verified by comparing predicted feeds sales to that reported by the producers. On the smaller farm, small amounts of corn or hay were sold in only a few of the high-yielding weather years. On the larger farm, essentially no feed was sold from the farm and no forage was purchased. Simulation results showed small amounts of hay or corn grain sold from the smaller farm in half of the weather years, and no forage purchased or sold from the larger farm.

The final step was to compare predicted and actual purchased feed requirements for an average or typical year. On both farms, essentially all forage was produced on the farm, but most of the grain and all other concentrate feeds were purchased and imported to the farm. The predicted requirement of purchased concentrate was found to be essentially the same as that reported by farm records. The supplemental P fed on each farm was set in the model to equal that determined from the amount of concentrate and minerals imported to the farm and the P concentration in those feeds.

Management Options

After the modeled farms were found to adequately portray the existing farms, alternative management strategies were evaluated to determine their potential impact on the environment and farm profit. These management alternatives were selected in consultation with the farm owners. Thus, strategies studied were those of interest to the farmer that would potentially affect the whole-farm P balance.

Five alternative strategies were evaluated for the smaller farm. The first was an adjustment of the P imported as fertilizer and supplemental feed. On the current farm, small amounts of P were applied as starter fertilizer for the corn and oat crops (about 9 kg P/ha). Since there is excess P on the farm, this added P might be unnecessary. These applications were removed to determine their impact on soil P level. Like most farms, the amount of P fed to the animals was somewhat higher than that recommended by the NRC (1989). New NRC (2001) requirements have recently been released which reduce the recommended minimum level of P fed to dairy animals. The new requirements, based upon absorbable P, were implemented in the model to determine the whole-farm impact of reducing imported feeds and minerals to meet this lower level.

The next strategy evaluated greater use of pasture. Cows were fed during the growing season using intensively managed rotational grazing. This required that all land owned near the milking barn be converted to grass, providing 72 ha of pasture. An additional investment of $23,000 in fence and watering equipment was assumed for a total investment of $34,000. This included a high tensile wire fence around the perimeter of the pasture area and electric fence to maintain 20-30 paddocks. This change allowed better use of existing grassland and converted some existing corn and alfalfa land to grass. More intensive use of pasture increased the available yield by 50% (Table 2). Other changes made to represent a grazing farm included the following: 1) a 5% reduction in milk production, 2) a 15% reduction in the culling rate of cows, 3) a 30% reduction in veterinary expenses, and 4) a 20% increase in the life of equipment.

The third strategy evaluated removal of corn and alfalfa from the farm. This included changes made for the second strategy, plus all remaining corn and alfalfa land was converted to grass harvested as silage. To better utilize high quality grass, a three-cutting harvest was used where first and third cuttings were harvested as silage and second was harvested as hay. Considering the lower energy content of grass silage, milk production was decreased an additional 5%. This change in forage fed also affected the concentrate feeds required to meet animal energy and protein requirements (Rotz et al., 1999a).

The fourth strategy evaluated the conversion of the dairy operation to a heifer raising facility. All mature animals were removed, and heifer numbers were increased to 180 over one year old and 180 under one year old for a total of 360. Animals were purchased as calves ($100/calf) and sold as bred heifers ($1,650/heifer) near the time of calving. Feed production and land use were essentially that of the current farm. Facility changes were needed to accommodate the increased number of heifers and to remove milking equipment, but the overall investment required for animal facilities was the same as that of the current dairy system. Livestock expenses, consisting of veterinary, breeding, utility, supply, and related costs, were set at $106 per heifer produced (Penn State, 2000).

The fifth strategy evaluated expanding the farm business to include other family members. Animal numbers were increased to 250 cows with 200 replacement heifers. A double-ten milking parlor and freestall facility were used where all cows were housed and fed in confinement using a mobile mixing wagon. A manure storage tank was added to provide seven months of storage. Manure was applied as a custom operation in the spring and fall where it was tilled into the soil soon after application. The land area was increased to 304 ha, which was enough land to provide all the forage needed for the herd. The added land was used to produce 80 ha of alfalfa (with 20 ha of oat cover crop) and 40 ha of corn. Two large bunker silos were used to store the annual requirement of silage from the corn, alfalfa, and oat crops. A large forage harvester was used, but all other equipment remained the same as that on the current farm. Animals were fed to meet the latest recommended P requirements (NRC, 2001), and no starter fertilizer was used beyond that needed to maintain a long-term nutrient balance.

Four alternative strategies were evaluated for the 800-cow farm. The first was a change in feeding strategy that reduced the dietary P requirements to meet current recommendations of the NRC (2001). The second strategy included the same adjustment in the P feeding requirement along with an increase in milk production. Production was increased by 11% to an annual level of 9,080 kg/cow. This proposed increase may be achieved through changes in feed and animal management. Although this production level is high for the grazing-based management system used on this farm, it provides an achievable goal.

The third strategy involved an intensification of grass use on the farm. The current farm has a large land base with most of the land in grass production. This provides 2.5 ha/cow in total land with 2.1 ha/cow in grassland. Much of the grassland is under-utilized. Little to no fertilization or pest control is used, so grass yields are relatively low. To simulate greater utilization of the grass, grass crop area was reduced to 560 ha. This provided 884 ha of total farmed land, or a little over 1 ha per milking animal. Other management changes included application of N fertilizer at a rate of 75 kg/ha and an annual charge of $15/ha of grassland for weed and insect control, and perhaps overseeding of improved pasture species. As a result, annual grass yield was increased to 4.7 t DM/ha. Wilted silage harvests were taken from 145 ha of the grass in the spring and again in early summer. For the remainder of the year and for the remaining 415 ha of grassland, the grass was efficiently utilized through rotational grazing. This strategy, including the existing corn, alfalfa and triticale crops, provided all the forage needed to maintain the herd.

A fourth strategy combined all of the previous options. Grass production and use was intensified as just described; in addition, milk production was increased to 9,080 kg/cow and dietary P requirement was decreased to the newly recommended level (NRC, 2001).

Results

Twenty-five year simulation results include average annual predictions of feed production and use, nutrient loss and accumulation, production costs, and the net return or profit of the farm. Important results to consider are the comparisons between the different strategies simulated, not the absolute values generated for a particular farm. Predicted values for a given farm, such as soil P accumulation and net return, vary greatly depending upon model assumptions, and thus should not be used to judge the viability of a specific farm. Relative differences between simulated systems though, provide meaningful evaluation of the effects of system changes.

Small Farm

Simulation results for the 100-cow farm with the current production strategy are listed in the first column of Table 5. Long-term predictions of feed use compared closely to those of the actual farm. A small amount of grass hay was occasionally sold from the farm, and the purchase of concentrate feed mix was about 340 t DM per year. This verified that the model was adequately predicting crop production and feed use for the herd with an annual milk production of 9,500 kg/cow. Based upon the simulated feed production, purchased fertilizer, purchased concentrate feeds, and export in milk, animals and hay sold, the farm had an annual excess or accumulation of P. The annual accumulation of soil P averaged over the farm area was 5.1 kg/ha considering the assumed loss to the watershed of 1.0 kg/ha. An economic analysis of this production system indicated an annual net return to labor and management of $102,500 or $976/cow.

Alterations to reduce the P imported in fertilizer and feed are among the easiest management changes available to reduce the level of excess P. Eliminating the use of P starter fertilizer on the corn and oat crops reduced the accumulation of soil P to 3.1 kg/ha (data not shown). Assuming that this could be done without adversely affecting crop production, this reduction in fertilizer use increased annual farm net return a small amount ($5.50/cow). Dropping the P requirement in animal rations to the newly recommended requirement levels (NRC, 2001) further reduced the soil P accumulation, providing a long-term nutrient balance with a need for a small amount of P fertilizer (0.9 kg P/ha). Decreasing the level of P fed without a change in the current fertilizing practice reduced the whole-farm accumulation of soil P, giving an excess of 1.1 kg P/ha with a loss of 0.8 kg P/ha. The savings in purchased minerals increased the annual farm net return by $21/cow.

Use of intensive rotational grazing reduced the concentrate feed purchased, reduced soil P accumulation, and increased farm profitability (Table 5, column 2). Rotational grazing increased the productivity of grassland, which increased the amount of forage produced on the farm. Use of high-quality pasture, along with the assumed 5% reduction in milk production, reduced the purchase of concentrate feed by 17%. High protein levels in pasture forage led to excess N excretion, which increased N volatilization loss 10%. Reducing the amount of manure applied to fallow corn land in the fall though, reduced N leaching loss 14%. The reduction in the import of fertilizer and concentrate feed reduced the import of P to the farm by 3 kg/ha, which reduced the accumulation of excess P in the soil by about the same amount. A reduction in harvest, feeding, and manure handling operations, along with a reduction in purchased feed costs, increased farm profitability by $10,000 or $96/cow. Use of grazing also reduced the economic risk due to annual weather variations. This occurred because grass yields were more consistent than corn yields across weather years on this shallow soil. Also, there was an excess of pasture forage during most years, so drought years did not have much effect on production costs.

Table 5. A comparison of annual production and economic effects for various management changes simulated on the smaller (100-cow) farm in New York.

Production or economic output	Units	Current*	Intensive grazing†	All grass‡	Heifer raising§	Expansion#
Hay and silage production	t DM	395	220	430	308	828
Corn silage production	t DM	303	264	0	303	707
High moisture corn production	t DM	41	10	0	41	118
Grazed forage consumed	t DM	146	481	481	387	223
Forage purchased (sold)	t DM	(28)	(40)	(29)	47	30
Concentrate purchased	t DM	339	282	290	125	649
Milk production	kg/cow	9,530	9,080	8,620	0	9,530
Nitrogen imported	kg/ha	214.0	212.9	198.7	193.5	225.2
Nitrogen exported	kg/ha	57.5	55.8	47.8	28.9	68.3
Nitrogen volatilization loss	kg/ha	66.4	73.4	70.3	87.3	56.7
Nitrogen leaching loss	kg/ha	39.0	33.5	15.1	39.9	51.1
Phosphorus imported	kg/ha	14.6	11.7	10.7	4.9	10.5
Phosphorus exported	kg/ha	8.5	8.4	8.0	4.6	10.2
Phosphorus loss	kg/ha	1.0	1.0	1.0	0.9	0.9
Phosphorus accumulation	kg/ha	5.1	2.3	1.7	0.0	0.0
Machinery cost	$	48,528	47,147	43,148	48,338	73,423
Fuel and electric cost	$	4,047	3,353	3,124	4,212	10,481
Storage facilities cost	$	14,217	13,002	11,575	14,217	31,722
Seed, fertilizer, and chem. cost	$	26,633	27,366	25,442	26,633	30,733
Land rental and property tax	$	13,614	13,478	13,380	13,614	25,820
Purchased feed & bedding cost	$	73,004	60,891	61,914	34,169	145,520
Animal & milking facilities cost	$	42,746	42,746	42,746	22,173	81,666
Livestock expenses	$	38,108	34,958	34,958	40,050	61,150
Total production cost	$	260,897	242,940	236,285	203,406	460,515
Milk, feed, and animal income	$	363,422	355,499	340,229	294,414	785,060
Net return to labor and mgmnt	$	102,525	112,559	103,944	91,008	324,545
Standard deviation in net return	$	9,068	7,348	3,615	10,768	27,345

* 105 cows and 105 heifers on 182 ha of land with 32 ha of corn, 40 ha of alfalfa and 109 ha of grassland.

† Pasture area is increased to 72 ha and cows and older heifers are rotationally grazed.

‡ All crop land is converted to grassland, pasture area is increased to 72 ha and cows and older heifers are rotationally grazed.

§ Feed production is similar to current practice, but milking animals and facilities are removed and replaced by growing heifers and associated housing. Animals are fed to meet NRC (2001) P requirements.

\# Farm is expanded to 250 cows and 200 replacement heifers on 304 ha (80 ha alfalfa seeded with oat cover crop, 40 ha corn, and 184 ha grassland). Animals are fed in confinement to meet NRC (2001) P requirements.

Replacing the remaining corn and alfalfa with grassland had little effect on forage use and concentrate feed requirements (Table 5, column 3). Excess P dropped to 1.7 kg/ha; including the newly recommended reduction in dietary P allowed a long-term P balance (data not shown). The major change was a large reduction in N leaching loss to ground water. Compared to fall application of manure on fallow corn land, the model predicted that much more of the soil nitrate from manure would be captured and retained by the grass, thus reducing the potential for leaching loss in the spring. Production costs were a little less than those of the previous grazing strategy due to a reduction in annual costs of tillage, planting, seed, fertilizer, and chemicals for corn production. Due to the assumed decrease in milk production, farm income decreased providing an annual net return similar to that of the current farm. As in the previous strategy, the risk or variation in net return across weather years was reduced due to more consistent crop yields and an abundance of pasture forage during most weather years.

Changing the dairy farm to a heifer raising operation may also provide a long-term P balance for the farm, but this is dependant upon the number of animals raised. A production level of 180 bred heifers per year was selected because this provided similar feed utilization as the current dairy strategy (Table 5, columns 1 and 4). More pasture forage and more total forage were used by the heifers, so a small amount of forage was needed during some weather years. The need for concentrate feeds was reduced by over 60%, and the need for supplemental P in heifer diets was low compared to that of lactating animals (NRC, 2001). Both the import and export of P were greatly reduced providing a long-term P balance.

The heifer raising strategy also showed good economic potential. Production costs were similar to those of the current dairy facility except for a large reduction in purchased feed costs and the elimination of milking equipment costs. The annual net return to labor and management was $11,500 less than that of the current operation. Considering the labor saved by eliminating the milking operation (about 1,200 h/yr) though, the return is very good for the amount of time devoted to the operation.

Expanding the farm to 250 cows can also be done in a manner that leads to less accumulation of P on the farm. However, the stocking rate (animals per unit land) and the supplementation level of P in animal rations are important considerations in planning this expansion. In this analysis, land areas in corn and alfalfa were increased (along with current grass area) to provide nearly all of the forage needed and some high-moisture corn during good corn growing years. This feed production plan was similar to that of the current operation. This required a 70% increase in land area with a good soil similar to that currently used for corn and alfalfa production. Phosphorus requirements in animal rations were also reduced to the new NRC requirements, which reduced the import of P in purchased feeds.

Nutrient flows through the farm were generally improved in the expanded farm. With six months of manure storage and more rapid incorporation of that manure, N volatilization loss was reduced 17% (Table 5, columns 1 and 5). However, with less volatile loss and greater N application per unit of land, N leaching loss increased 30% (12 kg/ha). A long-term P balance was maintained under the land area, animal numbers, and other farm characteristics assumed.

The expanded dairy operation also provided a good economic return. Most production costs doubled with the increase in animal numbers and crop area, but the income also more than doubled. The difference between income and production costs (excluding labor), or the annual net return to labor and management, increased by more than three-fold to $324,500 or $1,300/cow. Given that this net income must support three families, the net return per family was similar to that of the current farm. Although the standard deviation in net return increased considerably, the CV, or variation expressed as a percent of the annual mean, was similar to that of the current farm.

Large Farm

Production and economic results of the current production system on the 800-cow farm are listed in column 1 of Table 6. Grazed forage yields were reduced considerably to match simulated and actual feed production and utilization records. With this change, though, predicted forage production, grazed forage consumed and concentrate feeds purchased were all similar to that experienced on the actual farm.

Based upon the simulated production levels, purchased feed and fertilizer, and the large land base, the nutrient balance and losses from the farm appear very reasonable when viewed on a per unit of land basis (Table 6, column 1). Nitrogen volatilization loss is quite low compared to that of the smaller farm (Table 5, column 1). This is partially due to more rapid incorporation of manure, but primarily this number is small because the total loss is spread over a large land base. When the loss is determined on an animal unit basis, N volatile loss is only 35% less on this farm compared to the smaller farm. Nitrogen leaching loss is also less when spread over the entire farm area, but the loss per animal unit is similar between farms. Excess P accumulation in the soil is also less per unit of farmland, but considering the large land base, this accumulation represents a substantial build up of P on the watershed.

Again, an easy and economical method to reduce the P import and accumulation on the farm is to reduce the dietary P of the animals. Implementing the new NRC (2001) recommended feeding level reduced the import of P by 5,900 kg per year, which brought the farm close to a long-term balance (data not shown). Thus, P imported in feed essentially equaled that exported in milk and animals sold. Reducing the mineral P added to feed decreased the annual feed cost and thus improved farm net return by $19,000 ($24/cow).

Increasing the milk production level of the herd increased feed consumption and purchased feed costs. The increased cost was more than offset by the increase in milk income, providing an increase in net return of over $150,000 per year. Using the current P feeding levels, this change created a small increase in the accumulation of excess P on the farm (data not shown). By combining this increase in production with a reduction in the P feeding requirement, a P balance for the farm was approached and net return was increased further (Table 6, column 2).

Farming less grassland area and farming it more intensively showed some economic benefit with mixed effects on nutrient balance and loss (Table 6, column 3). The simulation indicated that grazed forage consumption was increased along with a reduction in harvested grass silage. This provided a reduction in machinery and fuel costs and an increase in seed, fertilizer, and chemical costs. Purchased feed cost also decreased. All together, these changes increased the annual net return by $57,500 or $72/cow. Nitrogen volatilization increased a small amount with increased use of N fertilizer and leaching loss decreased as N was recycled more efficiently in the higher yielding, higher quality forage. The P balance over the whole land base (2,025 ha) was affected very little by this change. Since the excess P would be confined to less land, the annual accumulation on the 884 ha farmed would increase to 6.8 kg/ha.

Combining more intensive use of grass with greater milk production and reduced feeding of mineral P improved nutrient use and increased profitability (Table 6, column 4). With the combined changes, a reduction in N leaching loss was obtained along with a long-term P balance. The predicted profitability of the farm more than doubled with the increased milk sales and reduced production costs. The risk or year-to-year variation in net return also decreased by obtaining a more consistent forage supply on the available land area.

Table 6. A comparison of annual production and economic effects for various management changes simulated on the large (800-cow) farm in New York.

Production or economic output	Units	Current production strategy*	Increased milk production†	Intensified grass use‡ Current production	Intensified grass use‡ Increased production†
Hay and silage production	t DM	1,676	1,678	955	956
Corn silage production	t DM	1,928	1,929	1,921	1,921
Grazed forage consumed	t DM	1,548	1,546	2,270	2,265
Concentrate purchased	t DM	1,981	2,281	2,015	2,312
Milk production	kg/cow	8,172	9,080	8,172	9,080
Nitrogen imported	kg/ha	102.5	105.9	102.2	105.3
Nitrogen exported	kg/ha	28.5	30.7	28.8	30.9
Nitrogen volatilization loss	kg/ha	28.5	28.9	30.6	30.7
Nitrogen leaching loss	kg/ha	27.3	27.5	21.7	21.9
Phosphorus imported	kg/ha	7.9	5.5	7.7	5.2
Phosphorus exported	kg/ha	4.2	4.6	4.2	4.6
Phosphorus loss	kg/ha	0.6	0.4	0.6	0.4
Phosphorus accumulation	kg/ha	3.1	0.5	2.9	0.2
Machinery cost	$	216,792	217,650	195,862	196,673
Fuel and electric cost	$	47,561	48,004	34,841	35,266
Storage facilities cost	$	43,851	43,854	42,598	42,600
Seed, fertilizer, and chem. cost	$	88,083	88,083	91,196	91,196
Land rental and property tax	$	100,305	100,305	100,305	100,305
Purchased feed & bedding cost	$	427,617	455,318	400,405	427,093
Labor	$	600,000	600,000	600,000	600,000
Animal & milking facilities cost	$	193,187	193,187	193,187	193,187
Livestock expenses	$	194,000	194,000	194,000	194,000
Total production cost	$	1,911,398	1,940,402	1,852,395	1,880,320
Milk, feed, and animal income	$	2,102,755	2,311,998	2,101,271	2,310,613
Net return to management	$	191,357	371,596	248,876	430,293
Standard deviation in net return	$	60,374	59,525	53,251	52,944

* 800 cows and 600 replacement heifers on 2,025 ha of land with 162 ha of corn, 162 ha of alfalfa and 1,700 ha of grassland.

† Milk production is increased 5% through changes in animal and feeding management. Animals are fed to meet the new NRC (2001) P requirements.

‡ Grass area is reduced to 560 ha with 415 ha grazed in the spring and all grass grazed in summer and fall. Management changes include application of 75 kg/ha of N fertilizer on all grassland and an annual cost of $15/ha for pest control and overseeding.

Discussion

These farm simulations illustrate that management changes can be made to reduce the P loading on dairy farms in this region. These changes can be made while maintaining and likely improving the profitability of the farms.

One of the easiest ways of reducing P import and the resulting accumulation in soil is to reduce the amount of P fed. The newly released NRC (2001) recommended P requirements for dairy animals are lower than past recommendations, and recent experimental work has shown that these reduced dietary levels can be maintained without adverse effects on animals (Wu et al., 2000; Satter, 2001). In many diets, this lower level of P can be met by removing mineral P that is added to supplemental feed at a cost of about $3.00/kg of P. As found in these simulations, removing this added mineral can reduce the annual feed cost and thus improve farm profitability by more than $20/cow. When byproduct feeds high in P content are fed, these supplemental feeds may need to be replaced with other feeds similar in cost and lower in P content to achieve the same reduction in imported P.

Changes in cropping strategies may also affect P loading, but these effects are normally small. A cropping change of replacing corn and alfalfa with grass showed a small difference in the P balance for the farm. In a previous study, added corn, barley, soybean, or pastureland had essentially the same impact on the P balance of a dairy farm (Rotz et al., 2000). Cropping changes will only affect the whole-farm P balance if the crop change greatly affects the import of supplemental feed or fertilizer or the export in sold feed. On dairy farms where manure provides most of the P requirement for crops and those crops are used on the farm, cropping changes have little effect on the import and export of P.

Better utilization of a crop such as grass may provide some reduction in the level of excess P on a farm. This was illustrated on the smaller farm where the use of rotational grazing of the milking animals provided a substantial reduction in purchased feed. On the large farm where more intensive use of grass had little effect on purchased feed, there was also little difference in the accumulation of excess soil P.

Use of more grass on farms in this region may have added benefit in reducing P runoff flow into streams and ultimately the reservoirs for the New York City water supply. Topography in this region is very hilly, and the soils are erodable. A sod cover of perennial grass will reduce water movement and help hold the soil and nutrients on the landscape (Sharpley and Rekolainen, 1997). As differences in runoff loss of P among crops were not modeled in this study, this should be viewed as an added benefit for grass production. Ideally we want to prevent the accumulation of soil P, but if high levels are obtained, losses should be less if a perennial ground cover is maintained compared to annual tillage for corn planting.

Intensive use of grazing potentially reduces the whole farm accumulation of P, but this strategy may cause a problem for the distribution of P within the farm. When cows fed a high-concentrate diet are on permanent pasture for a major portion of the day, there is a potential for more P to be applied through manure than is removed through grazing. For these farms, rotational grazing of the lactating cows improved grass production and utilization, which led to a reduction in the amount of excess P on the farm. Lactating cow pastures were also near a long-term balance as long as additional manure was not spread over these fields.

A synopsis of all simulation results indicates that dairy farms in this region can maintain a long-term P balance if: 1) animals are fed to meet NRC (2001) P requirements, 2) the cropping strategy and land base used supplies all of the forage needed, 3) all animals are fed a high forage diet, and 4) replacement heifers are produced on the farm. Here, high forage diets are defined as those that seek to feed the maximum amount of forage to each animal group while

meeting their energy and protein requirements with supplemental feeds (Rotz et al., 1999a). The current 100-cow farm, the expanded 250-cow farm, and the current 800-cow farm were all able to maintain a P balance when these conditions were met. Raising replacement heifers helps maintain a nutrient balance because these animals consume more homegrown forage and less purchased concentrate relative to lactating cows, which reduces nutrient imports. Conversion of the smaller dairy farm to a heifer-raising facility also provided a long-term balance when these conditions were met. Maintaining higher milk production levels through greater import of purchased concentrates increases the level of excess P, but as illustrated on the 800-cow farm, excess P is minimal when these conditions on dietary P and forage production and use are followed.

Clearly, a key strategy for P management on dairy farms is to reduce concentrate P inputs from feed and fertilizer and to use farm-grown forage to recycle nutrients. When combined with conservation practices that minimize runoff and erosion, P loss to the watershed will be reduced by making better use of nutrients on the farm. When excess nutrients accumulate on the farm, it is essential that this excess P in manure be applied to parts of the farm that are least vulnerable to loss by runoff and erosion (Gburek et al., 2000). Thus, with consideration of site topography and hydrology, as well as P needs of the crops, carefully managed application of P accumulated in each modeled strategy can reduce losses. If land application of the accumulated P is not carefully managed, the production strategy that accumulates more P will have the greatest long-term impact on losses to the watershed.

Conclusion

Management changes can be made to prevent the long-term accumulation of soil P on dairy farms in southeastern New York. On simulated farms, reducing the level of dietary P fed and maximizing the use of farm-grown forage provided a long-term P balance along with an increase in farm profitability.

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References

Borton, L. R., C. A. Rotz, H. L. Person, T. M. Harrigan, and W. G. Bickert. 1995. Simulation to evaluate dairy manure systems. *Applied Engineering in Agriculture* 11:301-309.

Fox, D.G., R.R. Seaney, and D.L. Emmick. 1992. Increasing the productivity of forages and cattle on hillside land in New York State. In *Search Agriculture*. Ithaca, NY: Cornell Univ. Agr. Exp. Sta. No. 35, 76 pp.

Gburek, W.J., A.N. Sharpley, A.L. Heathwaite, and G.J. Folmar. 2000. Phosphorus management at the watershed scale. J. Environ. Qual. 29:130-144.

Harrigan, T. M.; W. G. Bickert and C. A. Rotz. 1996. Simulation of dairy manure management and cropping systems. *Applied Engineering in Agriculture* 12:563-574.

Haygarth, P.M., P.J. Chapman, S.C. Jarvis, and R.V. Smith. 1998. Phosphorus budgets for two contrasting grassland farming systems in the U.K. *Soil Use and Management* 14:160-167.

Kellogg, R.L. and C.H. Lander. 1999. Trends in the potential for nutrient loadings from confined livestock operations. In *The State of North America's Private Land.* U.S. Govt. Print. Off., Washington DC: USDA-NRCS <www.nhq.nrcs.usda.gov/land/pubs/ntrend.html>.

Kotak, B.G., S.L. Kenefick, D.L. Fritz, C.G. Rousseaux, E.E. Prepas, and S.E. Hrudey. 1993. Occurrence and toxicological evaluation of cyanobacterial toxins in Alberta lakes and farm dugouts. *Water Research* 27:495-506.

Lander, C.H., D. Moffitt, and K. Alt. 1998. Nutrients Available from Livestock Manure Relative to Crop Growth Requirements. Resource Assessment and Strategic Planning Working Paper 98-1. Washington DC: USDA, Natural Resources Conservation Service. <http://www.nhq.nrcs.usda.gov/land/pubs/nlweb.html>.

McFarland, A.M.S. and L.M. Hauck. 1999. Relating agricultural land uses to in-stream stormwater quality. J. Environ. Qual. 28:836-844.

Meals, D.W. 1990. LaPlatte River watershed water quality monitoring and analysis program: comprehensive final report. Program Report No. 12. Res. Center, Univ. Vermont, Burlington, VT: Vermont Water Resource.

National Research Council. 1989. *Nutrient Requirements of Dairy Cattle.* 6th rev. ed. National Academy Press. Washington, DC.

National Research Council. 2000. *Watershed Management for Potable Water Supply: Assessing the New York City Strategy*. National Academy Press. Washington D.C.

National Research Council. 2001. *Nutrient Requirements of Dairy Cattle.* 7th rev. ed. National Academy Press. Washington, DC.

New York Agricultural Statistics Service. 2000. County Estimates. Dept. of Agriculture and Markets, Albany, New York. <http://www.nass.usda.gov/ny>.

Penn State. 2000. Agricultural alternatives – dairy heifer production. Cooperative Extension Service, The Pennsylvania State University, 112 Agricultural Administration Building, University Park, PA.

Rotz, C. A., D. R. Buckmaster, D. R. Mertens, and J. R. Black. 1989. DAFOSYM: a dairy forage system model for evaluating alternatives in forage conservation. *J. Dairy Science* 72:3050-3063.

Rotz, C.A., N.P. Martin, and J. Grabber. 2000. Crop options for an expanding dairy farm in Wisconsin. p. 100-104. Proc. American Forage and Grassl. Conf. Madison, WI.

Rotz, C. A., D. R. Mertens, D. R. Buckmaster, M. S. Allen, and J. H. Harrison. 1999a. A dairy herd model for use in whole farm simulations. *J. Dairy Science* 82:2826-2840.

Rotz, C.A., R.E. Pitt, R.E. Muck, M.S. Allen, and D.R. Buckmaster. 1993. Direct-cut harvest and storage of alfalfa on the dairy farm. *Transactions of the ASAE* 36:621-628.

Rotz, C. A., G. W. Roth, K. J. Soder, and R. R. Schnabel. 2001. Economic and environmental implications of soybean production and use on Pennsylvania dairy farms. *Agronomy J.* 93:418-428.

Rotz, C.A., L. D. Satter, D.R. Mertens, and R. E. Muck. 1999b. Feeding strategy, nitrogen cycling, and profitability of dairy farms. *J. Dairy Science* 82:2841-2855.

Satter, L..D. 2001. Phosphorus in, phosphorus out. *Resource.* pp 13-14, May 2001.

Shaffer, M. J., A. D. Halvorson, and F. J. Pierce. 1991. Nitrate leaching and economic analysis package (NLEAP): model description and application. In *Managing Nitrogen for Groundwater*

Quality and Farm Profitability, 285-298, eds. R.F. Follett, D.R. Keeney, and R.M. Cruse. Madison, WI: Soil Science Society of America, Inc.

Sharpley, A.N. and S. Rekolainen. 1997. Phosphorus in agriculture and its environmental implications. In *Phosphorus Loss from Soil to Water.* 1-54, eds. H.Tunney, O.T. Carton, P.C. Brookes, and A.E. Johnston, Cambridge, England: CAB International Press.

Sims, J.T., R.R. Simard, B.C. Joern. 1998. P losses in agricultural drainage: Historical perspective and current research. *J. Environmental Quality* 27:277-293.

Soder, K.J. and C.A. Rotz. 2001. Economic and environmental impact of four levels of concentrate supplementation in grazing dairy herds. *J. Dairy Sci.* in press.

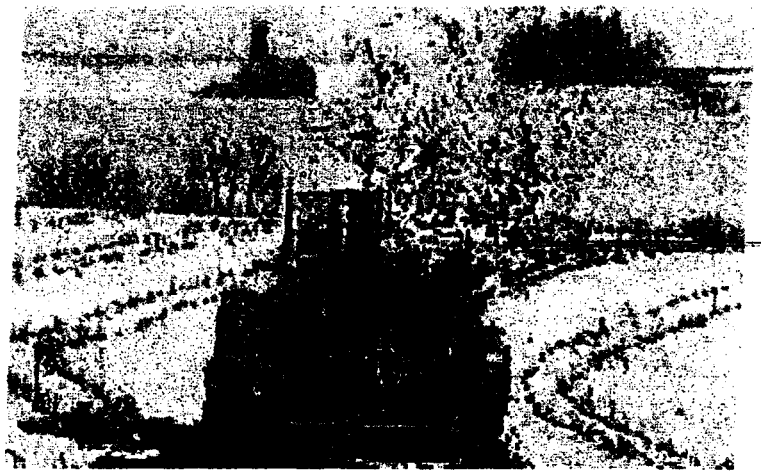
Sugiharto, T., T.H. McIntosh, R.C. Uhrig, and J.J. Lardimois. 1994. Modeling alternatives to reduce dairy farm and watershed nonpoint source pollution. *J. Environmental Quality* 23:18-24.

VanDyke, L.S., J.W. Pease, D.J. Bosch, and J.C. Baker. 1999. Nutrient management planning on four Virginia livestock farms: impacts on net income and nutrient losses. J. Soil and Water Conservation, 54:499-505.

Withers, P.J.A., S. Peel, R.M. Mansbridge, A.C. Chalmers, and S.J. Lane. 1999. Transfers of phosphorus within three dairy farming systems receiving varying inputs in feeds and fertilizers. *Nutrient Cycling in Agroecosystems* 55:63-75.

Wu, Z., L.D. Satter, and R. Sojo. 2000. Milk production, reproductive performance, and fecal excretion of Phosphorus by dairy cows fed three amounts of phosphorus. J. Dairy Sci. 83:1028-1041.

Impacts of Winter Spreading of Manure on Water Quality - Literature Review



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Impacts of Winter Spreading of Manure on Water Quality - Literature Review

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Background

Spreading livestock manure in the winter has been a common practice in Ontario for many years. For the farmer, there are several advantages, including: a) ability to build smaller manure storages; b) ability to spread the manure at a time when there is less pressure to get to the crop in the ground; c) spreading manure on frozen ground may help to reduce soil compaction. Despite these practical reasons for spreading manure in the winter, the concern about impacts on water quality has lead to a general acceptance that spreading manure in the winter is no longer environmentally acceptable.

Depending on the condition of the soil, runoff can potentially carry manure nutrients and bacteria to nearby surface waters. It is widely believed that frozen or snow-covered soils allow less infiltration than non-frozen bare soils. This is the main reason why policies within Canada recommend avoiding the winter spreading of manure.

The objectives of this literature review are:

1. to provide a brief overview of some of the policies in Canada concerning the winter spreading of manure; and
2. to review North American research that examines the implications of spreading manure in the winter.

Canadian Recommendations and Policies

Most provinces in Canada recommend avoiding the application of manure on frozen or snow-covered ground. The policies for selected provinces are summarized in Table 1. Provinces not listed generally have recommendations similar to those outlined below. The guidelines tend to be fairly general and rely mainly on the common sense of the manure applicator.

Table 1 - Summary of selected provincial policies dealing with winter-spreading of manure

Province	Type	Summary
Prince Edward Island	guidelines	If it is necessary to spread manure in the winter, i) it should only be applied when the potential for surface runoff is minimal; ii) it be applied to stubble fields with good trash cover; and iii) the distance from watercourses and wells be increased (PEI government 2000).
Ontario	guidelines for manure; regulations for sewage biosolids	Manure should "not be spread on frozen or ice covered soil". It is acceptable to spread manure when there is snow on the ground only when the ground is not frozen. The government acknowledges that slope influences nutrient movement over a surface: they indicate that manure may be spread on frozen ground providing the field has a sustained slope of 3% or less. These recommendations also govern the use of sewage biosolids on agricultural land (Ministry of the Environment and Energy, and Ministry of Agriculture, Food and Rural Affairs 1996). Manure should only be applied in emergency situations (during those months when the land is frozen, bare or snow-covered) onto grass winter cover crops or onto fields with high crop residue where there is no danger of run-off or floods (OMAFRA and AAFC 1992).
Manitoba	regulations	Large-scale livestock operations are prohibited from spreading manure between November 10 and April 15. A large operation is defined as any livestock operation having more than 400 animal units of a given livestock type. Smaller-scale operations do not have to comply with this regulation, although they are still responsible for meeting minimum setback distance requirements from sensitive areas such as watercourses, wells, sinkholes and springs (Manitoba Agriculture and Food 2000).
Quebec	regulations	Manure spreading is prohibited between October 1 and March 31, and any other time when the ground is either frozen or snow covered. The October 1 date may be delayed if conditions permit (Quebec Ministry of the Environment 2000).

Research on Winter-Spreading of Manure

A - Winter-Spreading Studies

A number of studies have been conducted over the past several years to examine this issue. Table 2 contains a summary of the various projects. It distinguishes whether each study was small- or large-scale, the location of the study, duration of the study, manure type, and other pertinent information. Following is a brief description of each study, listed in chronological order:

1. Midgley and Dunklee (1945) carried out an extensive study (begun in 1935) in Vermont using fresh dairy manure. The field investigations were carried out at three different sites, having fairly steep slopes (i.e. 8, 10, and 20%). Manure was spread onto frozen, snow-covered ground in late December or early January. They found that all frozen soils were impervious to water, and considerable runoff therefore occurred when the soil was frozen. Steepness of slope had relatively little impact on runoff losses. Spreading manure on frozen ground resulted in large losses of N (nitrogen) in the runoff. In addition to the runoff losses, volatilization of ammonia was also considered to contribute to large N losses from the manure.

2. Hensler et al. (1970) spread fresh dairy manure onto field-scale plots. They found that runoff losses from manure applied to frozen ground were variable. During the first year of observation, they noted significant losses of N and P (phosphorus) in the runoff. They attributed these losses to a 2 cm rain that fell within 24 hours after manure application. During the subsequent year, there was very little precipitation throughout the winter months. This resulted in minimal nutrient losses in runoff. Over the two-year study, average runoff losses of N, P, and K were 10%, 6%, and 8%, respectively.

3. Converse et al. (1976) compared the nutrient run-off from fall, winter and spring treatments of solid dairy manure on ten different plots. They observed no significant difference in nutrient losses between seasonal treatments over the three year study period. However, they did find that the amount of nutrients lost varied directly with the volume of runoff. The following observations were made regarding runoff volume: 1) winter- and spring-manured plots had more runoff volume than fall-manured plots; and 2) the check plots had more runoff volume than all manured plots. These differences were attributed to variations in infiltration rates. The infiltration rates appeared to be influenced by number of earthworms present and by grass and mulch cover.

4. Klausner et al (1976) found that application rate and weather conditions played a large role in determining the amount of nutrients lost in runoff from winter-applied manure. Dairy manure was applied on frozen ground at three spreading rates for three consecutive winters. Excessive nutrient losses were seen when manure spreading occurred during active thaw periods. Minimal nutrient losses were seen when manure was applied (application rate: 35 tonnes/ha) and covered with snow, which melted at a later date. These minimal losses were comparable to the nutrient losses of plots receiving no manure.

5. In a Minnesota study, Young and Mutchler (1976) examined the effects of different

manure application times. Solid dairy manure was either applied: a) in the fall and plowed under, b) in the fall on frozen ground or c) in the spring on top of snow. Fields were plowed "up and down" the slope to represent the most severe erosion and runoff averages. The slope of the plots averaged 9%. Plots were covered with either fall-plowed corn, new alfalfa, or old alfalfa crops. The combination that created the most serious pollution potential was: manure applied onto frozen ground having alfalfa cover - up to 20% of the manure-N was lost in the spring runoff. They attributed this to two factors: 1) the plots with alfalfa provided a less rugged surface with which to slow the movement of water; and 2) the alfalfa plots remained frozen longer in the spring, thus allowing less infiltration time and more runoff for a prolonged period of time. This study pointed out the importance of measuring concentrations, as well as total volumes of runoff. The concentration of nutrients in the runoff from manured plots was much higher than that from the check plots. However, the total losses of nutrients was not much greater, because the total runoff volume was less. The researchers noted that spreading manure on top of snow, rather than before a snowfall, resulted in less soil, water and nutrient losses.

6. Young and Holt (1977) conducted an experiment based on the design of Young and Mutchler (1976). Using the same plots, solid dairy manure was again applied. They confirmed that winter-applied manure can appreciably reduce soil loss and reduce runoff and nutrient loss on plowed ground when compared to un-manured plots. Soil loss decreased because the manure acted as a mulch on the soil surface – absorbing the impact of raindrops and reducing the volume of the surface runoff. They also found that total nutrient and runoff losses were consistently less in the manured corn compared to the un-manured corn.

7. Phillips et al. (1981) conducted a six-year study aimed at finding the effects of rate and timing of manure application on nutrient loading of surface and subsurface water and on crop yields. They spread liquid dairy manure in the spring, fall and winter on a series of field plots. Winter-spreading resulted in considerably higher concentrations of N, P, and K in runoff, compared to spring and fall applications. The higher the rate of winter application, the higher the concentration of nutrients in the runoff. They concluded that manure application to areas that contribute snow-melt directly to surface water should be avoided.

8. Steenhuis et al. (1981) determined through laboratory and field experiments in Wisconsin that solid dairy manure spread on frozen ground (no snow cover) did not necessarily lead to a loss of N because not all frozen soils are impermeable. They found permeability varied with the temperature of the soil as well as the extent that pores were blocked by ice. Therefore, under some conditions, applying manure onto frozen ground may pose no more threat of contamination than fall-applied manure. They also found that the first meltwater had the highest concentration of N. It is this first meltwater after spreading that largely determines the fate of manure nitrogen. If the water infiltrates, there will be very little loss of N. However, if the water runs off, the losses will be high.

Table 2 – Summary of the main details of winter-spreading studies

Authors	Duration of Study	Location	Type of Study	# of Plots	Plot Size (m)	Soil Type	Manure Type	Slope (%)	Cover	Tillage
Midgley and Dunklee (1945)	3 to 6 yrs	Vermont	field and lab	N/A	92 m^2		fresh dairy manure	8, 10, 20		
				6			fresh dairy manure	slight		
Hensler et al. (1970)	2 yrs	Wisconsin	field	4	N/A	silt loam	fresh dairy manure	11	none	plowed on the contour
Converse et al. (1976)	3 yrs	Wisconsin	field	10	3 x 13.2	silt loam	solid dairy manure	10 – 12	alfalfa-grass mixture	N/A
Klausner et al. (1976)	3 yrs	New York	field	8	61 x 53.3	silt loam	dairy manure	2	corn trash	N/A
Young and Mutchler (1976)	3 yrs	Minnesota	field	8	4.06 x 23.35	N/A	solid dairy manure	9	4-corn, 2-new alfalfa with oat cover crop, 2- 6yr old alfalfa	corn-fall plowed
Young and Holt (1977)	3 yrs	Minnesota	field						tilled corn	up and down slope
Philips et al. (1981)	6 yrs	Ontario	field	14	75.6 x 11.6	sandy clay loam	liquid dairy	0.8	corn stubble	none
Steenhuis et al. (1981)	1 yr	Wisconsin	field	8	13 x 3	silt loam	solid dairy manure	10 – 12	none	plowed
			lab	4	N/A	2.5 cm sheet of polystyrene	solid dairy manure.	2 or 12	none	
Lorimor and Melvin (1996)	2 yrs	Iowa	field	24	3.8 x 22	silt loam	liquid swine manure	2.9	12-short bean, 12-long corn stubble	
Qu et al. (1996)	—	Alberta	lab	16 (trials)	N/A	N/A	dairy manure, compost	0.4	none	
Blais and Weil (1999)	2 yrs	Ontario	field	12	N/A	clay	liquid	level	N/A	N/A

9. Lorimor and Melvin (1996) investigated N losses in snowmelt runoff from winter-applied liquid swine manure. Manure was applied on bean stubble (12 plots) and corn stalks (12 plots) over a two year period. They examined runoff from fall-incorporated manure, early winter broadcast manure on frozen soil, late-winter broadcast manure on top of snow, and spring broadcast manure. Runoff N losses were measured and expressed as a percentage of the manure-N applied. They found that, generally, there was no significant difference between treatments - with the exception of one "catastrophic" event. Average runoff losses of N (% of manure-N applied) were: fall-incorporated - 1.5; early winter broadcast - 1.4; late-winter broadcast - 10.3; and spring broadcast - 0.6. A "catastrophic" event occurred when there was a snow-melt two days after a winter application of manure. In this case, the runoff loss was 17.4% of manure-N applied. Because of the risk of high nutrient losses, they advised against applying manure in the winter altogether.

Lorimor and Melvin (1996) also found that the type of winter cover crop affected the amount of nutrients lost in runoff. Because there was a higher accumulation of snow in the taller corn stubble compared to the shorter bean stubble, the resulting volume of water lost from the corn was larger. As a result, more nutrients were carried away in the runoff from the corn stubble than from the bean stubble. Lorimor and Melvin advised that if manure must be applied in the winter, it should be applied early so as to minimize the risk of snow-melt occurring. For late-winter application, they recommended waiting until after snowmelt, when most runoff had already occurred.

10. In a lab-scale experiment, Qu et al. (1996) found that the pollution potential of snow-melt runoff from composted manure applied on top of snow was significantly lower than the pollution potential from fresh manure.

11. A recently-completed study at Alfred College – University of Guelph measured the water quality implications of liquid manure applications on a level clay soil. Manure was applied in the late fall on frozen ground, and in the spring on unfrozen ground. Preliminary results indicated that, for these soil and weather conditions, neither late fall application nor spring application caused significant N contamination of surface runoff or subsurface drainage water (Blais and Weil, 1999).

B - Pathogens

Manure contains bacteria and protozoa such as fecal coliforms, like *Escherichia coli* (*E.coli*), and *Cryptosporidium parvum*, that can cause severe gastrointestinal illness in humans. The maximum allowable concentration of *E.coli* colonies in drinking water is zero. Water is deemed unsafe for swimming if the *E.coli* levels exceeds 100 colonies per 100 mL of water (Ontario Ministry of the Environment 1984).

The survival rate of *E. coli* in manure was studied by Tamasi (1981). The results indicated that survival is greater in cooler conditions (8°C) compared to warmer conditions (20°C). However, freezing conditions were not considered.

Freezing conditions were considered in studies by both Stoddard et al. (1998) and Kibbey

et al. (1978). Both studies found that manure applied in freezing conditions had a higher mortality rate of fecal coliform than spring-applied manure; and that freezing conditions are usually lethal to fecal bacteria.

Cryptosporidium parvum is a protozoan parasite that is transported through the fecal-oral route in the form of oocycts. Though infective doses vary, as few as 10 oocysts can establish an infection. An infection can be lethal if the host is immunocompromised, such as an AIDS victim or a chemotherapy patient (Carrington 1995). Olson (1999) found that the most favourable conditions for *Cryptosporidium* oocyst survival were at temperatures between -4° and 4°C in feces and water, whereas the least favourable conditions were at 25° C. In another study, Carrington and Ransome (1994) found that winter and spring stream water conditions were favourable for oocyst survival. Both of these studies illustrate that winter-spreading of manure does not guarantee oocyst die-off.

Overall, very little literature focussed on how temperature affects the survivability of pathogens following land application of manure.

C - Models

Mathematical modelling of manure application to snow-covered fields, confirmed by both lab and field studies, determined that particulate losses were minimal in snow melt, but the loss of organic nitrogen, ammonium and potassium were related to the melt rate (Steenhuis et al 1980)

D - Air Quality

While it was not the focus of this study, ammonia losses to the atmosphere are also an environmental concern. Steenhuis et al. (1979) determined that the rate of volatilization of ammonia from manure was diminished if the manure was spread in the winter. This was because of decreased wind speeds and temperature during the winter months. Lauer et al. (1976) found that when liquid dairy manure was spread onto snow and subsequently covered by a blanket of snow, the potential for ammonia volatilization was reduced to zero. Midgley and Dunklee (1945) found that even though N runoff losses were high for winter-spread manure, volatilization of N accounted for a higher proportion of the total N lost from manure (mainly due to the fact that volatilization starts as soon as the manure is produced).

E - Climate

All of the studies cited in this report have been carried out in areas where a typical winter involves a significant amount of snow cover. Several of the studies have pointed out the importance of weather conditions and snow cover on the potential for manure runoff following winter spreading. There are differences, however, from year to year for any given region, and from area to area within a state or province. Identifying areas of highest risk of runoff could involve using accurate climate data. An example of the type of mapping that is available to assist with this is included as Figure 1. It shows the average annual number of days with more than 5 cm of snow cover. Even within Southern Ontario, there is a considerable range of values. For example, Essex has typically less than 30 days, while Pembroke has greater than 120 days.

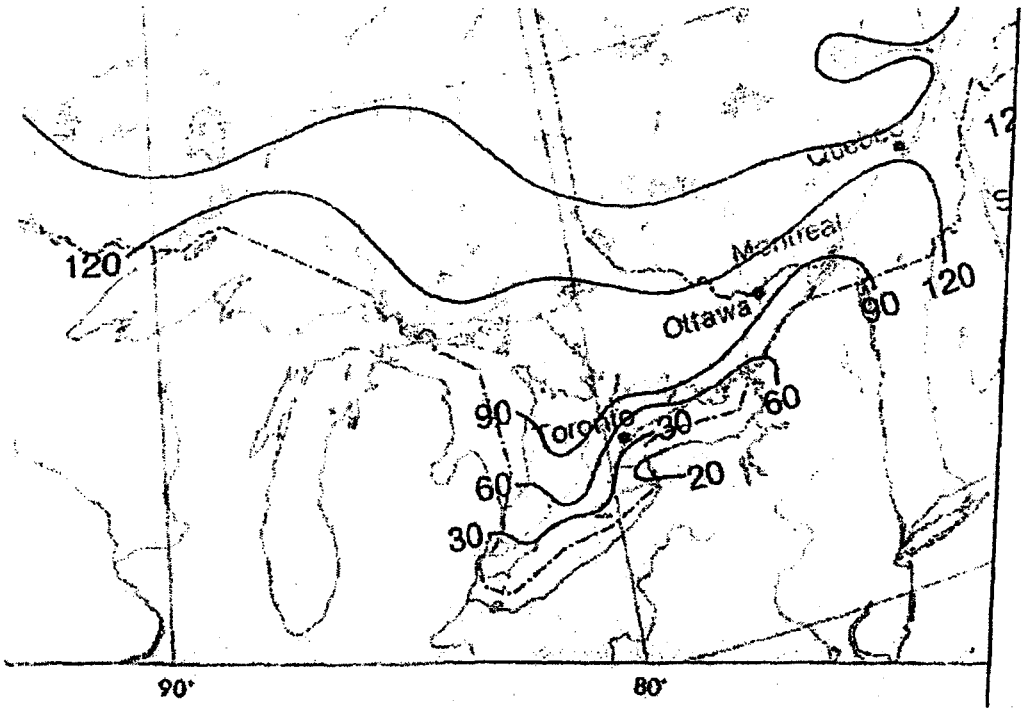

Figure 1 Mean annual number of days with more than 5 cm of snow on the ground - showing Southern Ontario (1951 to 1980) excerpted from: Climatic Atlas of Canada (Environment Canada 1987)

Summary

Of the several studies summarized in this report, it appears that there are similarities in the findings, and there are some conflicts. The following points appear to be generally true:

- Nitrogen lost in runoff following winter manure spreading can vary from negligible levels to upwards of 20% of the manure-N applied.
- The amounts of nutrients lost in runoff following winter application of manure are usually greater than from manure spread in other seasons, though this is not always the case.
- Many (not all) frozen soils are virtually impervious - there is a high likelihood that snowmelt and rainfall on manure-covered frozen ground will result in the runoff of manure constituents.
- The fate of the first meltwater or rainfall following winter manure spreading will usually determine the amount of manure runoff - if it soaks into the ground, the runoff amount will be relatively small - if the ground is frozen, the runoff amount can be relatively high.
- The risk of manure runoff appears to be similar, whether the manure is spread on frozen bare ground or on snow-covered ground.
- Spreading manure onto a cover crop in the winter does not necessarily reduce the risk of runoff.

- Spreading solid manure in the winter can actually reduce the amount of runoff and of soil erosion. It forms a mulch on the soil surface that slows down the flow of water.
- For the single study that looked at the influence of slope on manure runoff, it appeared that there was little difference for slopes of 10% and 20%. No information is available on the impact of lower slopes.
- Spreading manure in the winter provides no guarantee of pathogen die-off, though freezing conditions are usually lethal to fecal bacteria.
- The rate of volatilization of ammonia from manure is diminished for winter-spread manure, especially if the manure is covered by snow.

Recommendations

One of the goals of this literature review was to outline the various risk factors associated with winter-spreading of manure. However, the single greatest impact is "weather". Since this is a factor that is out of the control of the farmer and cannot be accurately predicted, the risk of runoff from winter-spread manure will be low some years and high in other years. Climate records may help to identify those areas where the risks are highest, though there are not likely any areas of the province where the risks are acceptable. The current Canadian standards and Best Management Practices appear to be quite reasonable and should be followed.

References

Blais, P.A., and Weil, C. 1999. Nitrogen losses in surface and drainage waters from spring and late fall liquid manure applications on a flat clay soil. Abstract of Technical Paper in Canadian Journal of Soil Science. 79(4):656.

Carrington, E.G. and Smith, H.V. 1995. The occurrence of *Cryptosporidium* spp. oocysts in surface waters and factors influencing the levels, with particular reference to the UK. In Protozoan Parasites and Water, ed. W.B. Betts et al., 57-62. Cambridge: the Royal Society of Chemistry.

Carrington, E.G. and Ransome, M.E. 1994. Factors influencing the survival of *Cryptosproidium* oocysts in the environment. Report no FR 0456, Foundations for Water Research, Marlow, Bucks.

Converse, J.C., Bubenzer, G.D., and Paulson, W.H.. 1976. Nutrient losses in surface runoff from winter spread manure. Trans.of ASAE. 517-519.

Environment Canada. 1987. Mean number of days with more than 5 cm of snow on the ground 1951-1980. in Climatic Atlas of Canada

Government of Prince Edward Island. 2000. Guidelines for manure management for Prince Edward Island: land application of manure. www2.gov.pe.ca/af/agweb/library/documents/manureguide/land.asp. Date accessed May 3, 2000.

Hensler, R.F., Olsen, R.J., Witzel, S.A., Attoe, O.J., Paulson, W.H, and Johannes, R.F. 1970. Effect of method of manure handling on crop yields, nutrient recovery and runoff losses. Trans.of ASAE. 726-731.

Klausner, S.D., Zwerman, P.J., and Ellis, D.F. 1976. Nitrogen and phosphorus losses from winter disposal of manure. Journal of Environmental Qaulity. 5(1):47-49.

Lauer, D.A., Bouldin, D.R., and Klausner, S.D. 1976. Ammonia volatilization from dairy manure spread on the soil surface. Journal of Environmental Quality. 5(2):134-141

Lorimor, J.C. amd Melvin, J.C. 1996. Nitrogen losses in surface runoff from winter-applied manure. University of Iowa. Final Report.

Kibbey, H.J., Hagedorn, C., and McCoy, E.L. 1978. Use of fecal streptococci as indicators of pollution in soil. Applied Environmental Microbiology. 35:711-717.

Manitoba Agriculture and Food. 2000. The Environment Act. http://www.gov.mb.ca/agriculture/livestock/pork/swine/bah02s02.html. Date accessed June 8, 2000.

Midgley, A.R., and Dunklee, D.E. 1945. Fertility runoff losses from manure spread during the winter. Univ. of Vermont, Agric. Exp. Station. Bulletin 523. pg 1-19

OMAFRA and AAFC. 1992. Best Management Practices: Livestock and Poultry Waste Management. Agriculture and Agri-Food Canada and the Ontario Ministry of Agriculture and Food. p 39.

Ontario Ministry of Environment and Energy, and Ontario Ministry of Agriculture, Food and Rural Affairs. March 1996. Criteria relating to spreading site: snow covered and frozen ground. In Guidelines for the Utilization of Biosolids and other Wastes on Agricultural Land. p.13.

Ontario Ministry of the Environment. 1984. Water management: goals, policies, objectives and implementation procedures of the Ministry of the Environment.

Olson, M.E., Goh, J., Phillips, M., Gusell, N., and McAllister, T. 1999. Survival of *Giardia* cysts and *Cryptoporidium* oocysts in water, soil, and cattle feces. Final Report.

Philips, P.A., Culley, J.L.B., Hore, F.R., and Patni, N.K. Pollution potential and corn yields from selected rates and timing of liquid manure applications. Trans. of ASAE. 1981: 139-144

Qu, G., Leonard, J, Feddes, J., and McGill, W. 1996. Effects of fresh and composted manure on runoff water quality when applied on snow. Canadian Society of Agricultural Engineering. Paper No. 96-205.

Quebec Ministry of the Environment. Reduction of pollution from agricultural sources: manure spreading conditions and limits. www.menu.gouv.qc.ca/sol/agricole-en/manure.html. Date accessed May 3, 2000.

Steenhuis, T.S., Bubenzer, G.D., Converse, J.C.. 1979. Ammonia volatilization of winter spread manure. Trans.of ASAE. p 152-157, 161.

Steenhuis, T.S., Bubenzer, G.D., Converse, J.C., and Watler, M.F. 1981. Winter-spread manure nitrogen loss. Trans.of ASAE. p. 436-449.

Steenhuis, T.S., Muck, R.E., Bubenzer, G.D., and Converse, J.C.. 1980. Modelling nutrient runoff losses from winter spread manure. In Livestock waste: a renewable source, proceedings, 4th International Symposium on Livestock Wastes- April 15-17. p 281-285.

Stoddard, C.S., Coyne, M.S. and Grove, J.H. 1998. Fecal bacteria survival and infiltration through a shallow agricultural soil: timing and tillage effects. Journal of Environmental Quality. 27:1516-1523.

Tamasi, G. 1981. Factors influencing the survival of pathogenic bacteria in soils. Acta Veterinaria Academiae Scientiarum Hungaricae. 29(2):119-126.

Young, R.A. and Holt, R.F. 1977. Winter-applied manure: effects on annual runoff, erosion, and nutrient movement. Journal of Soil and Water Conservation. 35(5): 219-222.

Young, R.A. and Mutcher, C.K. 1976. Pollution potential of manure spread on frozen ground. Journal of Environmental Quality. 5(2): 174-179.

Generic Environmental Impact Statement for Animal Agriculture in Minnesota

Topics I & J Soils and Manure Issues

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Executive Summary

Animal production is growing in Minnesota and animal densities are increasing. There are two main driving forces at work when manure utilization is considered. One is "on farm" profitability and the other is the negative "down stream" impacts. At times economic and environmental quality goals are competing. This Manure and Soils report addresses the value of manure as well the possible negative environmental effects of the use of manure on cropland.

Manure value, costs, economic forces, and water quality

Manure can be a valuable resource in a crop production system. Manure contains the macronutrients nitrogen (N), phosphorus (P), and potassium (K) and also contains numerous micronutrients. The nutrient value of manure for crop production depends on the site-specific reserve of plant available soil nutrients, the nutrient concentrations in manure, and the nutrient demands of the crop. In many areas of Minnesota soil levels of P and most micronutrients are already at adequate levels. In some soils K is also adequate for crop production. Under conditions of adequate soil P, K, or micronutrients there is little or no economic value associated with these elements in applied manure. Nitrogen is always needed for production of nonleguminous crops (e.g. corn, small grains, and grass pasture) and when correctly managed the N in manure is valuable. For legumes (e.g. alfalfa and soybeans) the N in manure is of little economic value because these crops are able to convert N in the air to plant available N. However, when, manure is applied to legumes these plants can utilize the plant available N instead of producing their own N and application to legumes can be an environmentally benign method for utilization of manure N.

The value of the nutrients in manure also depends on the value of alternative sources, mostly commercial fertilizers. Most N fertilizers are produced by conversion of N from the air using natural gas. Due to the recent rise in energy prices, the cost of N fertilizer has doubled from about $.20 to $.40 per pound. It took the previous decade for the cost to double from $.10 to $.20. The increased cost of N fertilizer has a direct impact on the value of N in manure. The cost of both fertilizer P and K has also doubled in the last decade. Thus, the value of these nutrients in manure has increased considerably and the interest in manure utilization is increasing.
In addition to supplying nutrients, manure can also improve biological and physical properties of the soil, making it more productive and less erosive. Manure, used as a part of good soil management, improves soil quality. It is difficult to put a dollar figure on this benefit.

The direct economic cost of manure is mainly for storage and application. Environmental costs are incurred if manure is over-applied, applied at the wrong time in the growth cycle, applied unevenly, or managed in such a way as to allow nutrient losses in storage, handling, and application. Water and air quality are often degraded in these situations. The risks of water and air quality degradation can be reduced with investments in high quality storage facilities. Good storage facilities reduce the environmental risk of poor containment during storage and provide for more flexibility in timing of a land application; allowing for land application to better match the time of plant uptake of nutrients. This reduces the likelihood of runoff and leaching losses of

N and P. The cost for application can be a major cost that offsets much of the value of manure if the site of application is not near the site of animal production. The cost of manure hauling increases with distance from the source. The cost of hauling is greater for manure with high water content compared to dryer manure.

The N challenge

For farmers, N is the most difficult manure element to efficiently utilize for crop production. Availability of N to the crop largely depends on the weather (temperature and precipitation) between application and plant uptake. Corn takes most of its N up between "knee high" (early June in Minnesota) and tassel emergence (early July to mid August). When manure is applied before planting there is a risk that some N can be lost by leaching before the plants are ready to take up the N. If temperatures are warm after application manure N can be converted to nitrate N, which can be lost by leaching during wet weather. This nitrate can eventually be leached into groundwater, degrading water quality. If temperatures are cool during the crop nutrient uptake period, crop uptake will be reduced; both because of low crop N demand and e the slow conversion of organic N to plant available forms.. The uncertainty associated with potential N losses and plant availability, and the recognized difficulty in applying manure evenly at the desired rate are reasons why farmers take low N credit from manure. This results in over application of manure or fertilizer N.

The P challenge

Phosphorus from manure is relatively stable in soil and can reliably be measured with soil tests. This is why farmer surveys often reveal that they take P credits from manure application rather than N credits when assessing crop nutrient needs. Because of the stability of P in soil, repeated manure applications can result in an accumulation of soil plant available P. Although this usually does not have a detrimental effect on crop production, it does present possible environmental concerns. Excess P when delivered to surface waters in runoff water can greatly increase the risk of algal blooms resulting in the degradation of water quality. The risk of phosphorus loss to surface waters is associated with runoff volume and soil erosion.

Recommended manure management practices should consider the risk of off-site movement of phosphorus to surface waters. Any manure application near surface water bodies, as well as manure applied on the soil surface without incorporation, applied at excessive rates, or applied on frozen or snow covered ground pose a high level of risk. However, the risk is also site dependent, with the erosion potential and the soil P level being important considerations. The P index being developed as a part of the project that produced this report will be an important tool for identifying sites where the risk of P loss is high.

N vs. P based manure applications

The quantity of manure application is one of the most important considerations when developing a manure plan. Should the rate be based on the manure P content and soil test P levels? Or should it be based on the manure N content and crop N requirements? Manure varies in relative content of N and P, and both are present as soluble inorganic forms and relatively insoluble organic forms. The composition of manure influences both crop uptake and risk of N and P losses to ground and surface waters. Most animal species have N and K contents in their manure that is greater than the P content. Poultry manure is an exception, with more P than N and K.

Some methods of manure management can increase the P to N ratio in manures, increasing the possibility of P risk to surface waters. Ammonium N can be lost as ammonia gas during storage, handling, and after application, which increases the P to N ratio. Crops require less P than N and the result of N losses is that fertilization to maximize the production of nonlegumues will require either commercial fertilizer N or an increase in manure application rate. Increasing the manure application rate increases the risk P contamination of surface waters.

Soils are both a source and sink for P. Continued manure application to meet N needs of crops for maximal crop production can increase soil test P to values where no additional P is required.. At high soil test P the risk that runoff water will carry excessive P to surface waters increases. The risks posed by high soil test P vary depending on many landscape and management factors.

Currently, best management practices recommendations for manure application are based on N. This approach reduces the potential for nitrate pollution of waters but in the long term this approach results in a buildup of soil P levels. The increase in soil test P to high values depends on the crop sequence, frequency of manure application, and soil type. The best environmental strategy would be to use P-based rates with supplemental fertilizer N where there is high risk of environmental impact from P (high P index), and use N based rates where the P risk is low.

One approach to limit the risk posed by high soil test P is to define a critical soil test P value that sets an upper limit for soil test P. However, some soils with high soil test P pose little risk to surface waters. These are soils that are distant from surface water bodies or surface tile inlets, low in erosion potential, in depressions without drainage outlets, or are separated from a water body by a wide buffer strip of native vegetation. A better approach than setting a critical limit is to use a P index. The P index approach takes into account the likelihood of P actually reaching receiving waters. It also provides more management options for farmers to reduce the environmental risk compared to a yes or no approach based on a critical soil test P value.

Application costs

Due to the cost of hauling and applying manure, the use of commercial fertilizers is more economical than manure as the hauling distance increases. In other words, the economic pressure encourages farmers to apply manure closer to the barn and at higher rates. Water quality concerns result in a pressure to apply manure further from the barn and at lower rates to better recover nutrients and reduce losses. As the animal concentrations increases the distances for application become greater. With liquid or semiliquid sources (especially if nutrient

concentrations are low) the transportation costs become prohibitive at distances greater than about one mile. With dry sources (the best example is poultry manure due to its high nutrient concentration and low water content) the distances with favorable economics are greater (~25 miles).

Environmental vulnerability due to landscape and climate

The most sensitive environmental regions in Minnesota for nitrate groundwater contamination are the deep glacial outwash sands in the central part of the state with shallow aquifers and the karst area in southeastern Minnesota where fractured limestone bedrock provides for entry of mobile contaminants directly into the aquifer. The relatively impermeable glacial till and glacial lakebed sediments in other major agricultural areas also pose a risk of nitrate loss to surface waters through the tile drainage systems that have been installed to remove excess water from the soil.

The most sensitive areas of Minnesota to P contamination of surface waters are where slopes are steep and erosion potential is high. This includes glacial moraines such as the one just south of the Twin Cities and near Alexandria as well as highly dissected landscapes such as in the southeastern part of Minnesota. Soil and water conservation techniques are important components to environmentally sound farming systems in these areas.

Risk of disease

Animal pathogens that can infect humans (zoonotic organisms) are a possible risk from application of manure to fresh market fruits and vegetables. Although transmission of pathogens to produce at concentrations that can infect consumers has occurred, the existing evidence suggests that this occurs rarely. Storage of manure decreases the concentration of disease organisms, especially under aerobic conditions, and generally reduces risk. The heat generated during composting essentially eliminates the risk. Risks can also be reduced by restricting fresh market produce production on recently manured land.

Recommendations

Nitrogen

- We must do a better job of characterizing the N in manure by utilizing chemical test to partition N into ammonium and organic forms. This allows more precision in prediction of plant available N.

- We must do a better job of placing manure N in the crop sequence to capture N. Manure N provides the most economic return if application precedes an N dependent crop. Application preceding legumes is environmentally acceptable if N does not exceed the ability of the

plants to take up the available N but it does not provide any economic benefit. Deeply-rooted, long-season crops have the best chance of recovering manure N.

- Manure applications are better if they are made closer to the time of crop N need.

- We need to develop combinations of soil and plant tests and climate monitoring that will allow synchronizing N availability with crop needs to minimize risk of crop yield reduction and environmental losses.

Phosphorus

- The use of a P risk index that considers P source characteristics and transport probabilities will provide a science-based risk assessment and help assess management options that can greatly reduce the entry of this pollutant into surface waters.

- The use of conservation tillage reduces erosion and total P losses in runoff. Increased P concentration near the soil surface and contributions from plant residues with these systems needs further evaluation.

- Set backs, buffer strips, and sand filters at surface tile inlets also need further research.

Nitrogen and Phosphorus

- The best environmental strategy is to use P-based rates with supplemental N where there is high risk of environmental impact from P (high P index) and use N based rates where the P risk is low.

Government Assistance

A combination of "carrot and stick" measures must be adopted to reduce the potential degradation of Minnesota's water resources by manure N and P.

Incentives

- The financial burden of upgrading manure systems should be shared with farmers. This includes storage, handling, and application costs. Governmental agencies should have programs to promote manure trading, adoption of soil and water conservation practices, and efficient nutrient management techniques to reduce risks.

- In high risk zones easements should be purchased for establishment of buffer strips in in close proximity to environmentally sensitive surface waters. Well designed buffers can provide multiple benefits, including wildlife corridors, habitat, and feed, aesthetic surroundings for water recreation, improved water quality, etc.

Education

- Educational programs must be implemented to increase the awareness and understanding of the complexity of the issue and options available to farmers to maximize profits and minimize environmental risk. This would lead to development of specific farm and field manure management plans (utilizing tools like the "P risk index" for assessment and development of plans).

- Research support is necessary to develop improved management techniques. This includes incorporating "real time" weather considerations; soil, plant, and manure testing; and improved soil, water, crop, animal ration, and manure management techniques. This research should include traditional "plot" scale research and large-scale evaluations of the economic costs and environmental benefits.

Regulation

- Eventually, manure management plans should be required on most farms that handle livestock and poultry manure.
- There must be penalties for egregious disregard of the environmental consequences of improper manure handling.

Update of Literature Review for Soils and Manure

Introduction

The value of manure for maintaining and improving the productivity of the soil has been recognized from antiquity, and fertilizing crops with livestock manure nutrients began millennia ago. With careful management, manure can be a good source of N (nitrogen), P (phosphorus) and K (potassium) in a crop production system. Manure can also improve soil quality. On the other hand, improper management of land application of manure can result in negative environmental effects, including soil build-up of toxic trace elements, and pollution of ground and surface waters with nitrate-N and phosphorus. In addition human and animal pathogens can be transmitted by contamination of water and the food chain. The risk of contamination in the human food chain is discussed in a separate section.

This section of the technical working paper (TWP) is an update of the soils and manure literature reviews (sections I and J) completed in 1999 by the University of Minnesota. This update focuses mainly on literature that has been published since the original literature review, and also includes interviews with researchers who have active studies related to the published literature we reviewed. It includes sections on soil quality, air quality, toxic trace elements, nutrient management, and risk of nitrates and phosphorus transport to ground and surface water. This literature review does not update any of the material on manure storage or the phosphorus index. Updates on storage and the phosphorus index are included in separate sections in this TWP.

Soil Quality

Management practices that minimize tillage operations, provide surface residues, use cover crops, or utilize organic residues like manure maintain soil organic matter and improve soil quality. Indicators of good soil quality include improved structure, water infiltration, pH, and soil respiration, decreased bulk density, and increased available-water holding capacity.

Soil quality on organic versus conventional farms

Increasing interest in food produced using organic sources of nutrients rather than agrochemicals has led to a growth in the sales of organic foods of 20% on average for the last 8 years in the U.S. (Looker, 1997). Manure management is a key component of most organically managed farms, and generally, organic management improves soil quality.

This was verified by the results of a comparative study of five organic and five conventional farms in Nebraska and North Dakota matched by soil type (Liebig and Doran, 1999). Four of the five organic farms used manure. The organic farms had 22% more organic matter and 20% more total N in the surface 30.5 cm of the soil. Microbial biomass C and N and soil respiration were found to be higher in the soil of the organic farms as compared with that of the conventional farms, at four of five locations. The organic farms had soil pH closer to neutral, lower bulk

density and higher available-water holding capacity as compared with conventional farms. The organic farms generally had more potentially mineralizable organic N (anaerobic incubation). The effects of organic management on soil nitrate, however, were inconsistent. Soil NO_3-N was greater on conventional farms, at four locations. However, soil NO_3-N on one organic farm was 10 times higher than on the comparative conventional farm. This high level of N suggests that organic production practices may not consistently be better at minimizing potential negative off-site impacts due to N loss to the environment.

Salts in manure

Very high manure applications can lead to salt accumulation in soils. Increases in soil salt content may have adverse effects on crops, particularly in dry climates. Eghball and Gilley (1999) determined the effects of simulated rainfall on runoff losses of salt following application of manure and compost to a silty clay loam soil in Nebraska. Manure, compost, and fertilizer were applied to the soil surface of no-till fields following sorghum and winter wheat, at rates required to meet the N or P requirements for corn production, and were either left on the soil surface or disked to 8 cm. No significant differences in runoff salt concentrations were found between the treatments.

Weed seeds in manure

Animal feed is often a source of weed seeds, which can consequently appear in manure. Composted and noncomposted beef cattle feedlot manures were applied to no-till fields on a silty clay loam soil in Nebraska (Eghball and Power, 1999). The effects of composted and noncomposted manure application on corn yield and weed biomass were determined. The authors concluded that weed biomass was more influenced by nutrient availability than by any weed seeds introduced by the composted or noncomposted manure application.

Air quality effects of manure in pastures and field soils

Methane (CH_4) is a greenhouse gas that can be produced by manure under anaerobic conditions. It is also produced in manure patches in pastures. This was shown in a simulated field study by Yamulki et al. (1999) in the UK. Dung and urine samples were deposited on six separate experimental plots at different times of the year to study the effects of environmental factors on CH_4 emission. The total cumulative flux of CH_4 for the dung and urine patches had mean values of 42.8 and 0.2 mg CH_4-C/patch, respectively. However, the CH_4 emissions from dung were insignificant compared to the methane produced in the cattle rumen.

Applying cattle slurry to soil may induce emissions of both CH_4 and nitrous oxide (N_2O). Nitrous oxide is also a greenhouse gas. The effects of slurry application (43.6m^3/ha) were studied for 9 weeks under controlled laboratory conditions using a soil microcosm system with automated monitoring of the CO_2, N_2O, and CH_4 fluxes (Flessa and Beese, 2000). A silty loam soil with a constant water-filled pore space of 67% was used. Emissions of N_2O and CH_4 from the injected slurry were significantly higher than from the surface-applied slurry, probably due to restricted aeration. Total N_2O-N emissions were 0.2% (surface application) and 3.3% (slit injection) of the slurry N added. Methane emission occurred only during the first few days following application. Losses of N_2O from cattle slurry were of minor importance compared with the nitrate leaching and other losses from the slurry treated soil but they may significantly contribute to the N_2O emissions from agricultural ecosystems. Slurry injection, which is recommended to reduce NH_3 volatilization, appears to increase emissions of the greenhouse gases N_2O and CH_4 from the fertilized fields.

Potentially Toxic Trace Elements

Under intensive livestock management, manure can contain significant concentrations of trace elements such as arsenic (As), cobalt (Co), copper (Cu), iron (Fe), manganese (Mn), selenium (Se) and zinc (Zn), due to feed additives. These trace elements are mostly retained in the solid phase after solid-liquid separation from slurry (Giusquiani et al., 1998). Excessive applications of poultry or pig manure may led to the accumulation of harmful levels of some of these elements in soils, and has the potential to result in toxicity to plants (phytotoxicity).

The most recent studies have concentrated on extractability, plant uptake, and leachability of trace elements, ways to reduce trace element concentrations in manure, and their potential for environmental contamination.

Extractability and plant uptake

Poultry litter can contain elements such as arsenic, cobalt, copper, iron, manganese, selenium, and zinc, all of which are added to poultry feed (Tufft and Nockels, 1991). Poultry litter from northern Georgia containing 1196, 944, and 631 mg/kg Cu, Mn, and Zn respectively, was used to study bioavailability for sorghum (Vanderwatt et al., 1994). Plant uptake of these metals from

poultry litter applications equivalent to 0, 15, 30, and 60 Mg/ha (Mg = metric ton) were compared on pure quartz sand and two Georgia soils. These levels of manure application are much higher than required to meet the needs of the crop. The quantity of Cu, Mn, and Zn added in the litter treatments were in the ranges 5 to 15, 62 to 1933, and 19 to 55 mg/kg, respectively. During a 21-day growth period, the plant concentrations of Cu and Zn were in the normal range, while concentrations of Mn were found to be toxic (>400 mg/kg), but only in the clay soil. Soil pH was important in determining extractable soil Cu, Mn, and Zn. More Cu, Mn, and Zn were found in sorghum on the soils with lower soil pH. Analyses of field soils revealed a build-up of possible phytotoxic levels of Cu, Mn, and Zn in one soil that had received 6 Mg/ha/yr of poultry litter for 16 years.

Amendments with organic matter may change soil pH and make metals more available to plants. One study found that eight years of sewage sludge or pig manure applied at rates of 5 tons/ha/year of dry organic waste on a sandy loam soil decreased soil pH by half a unit and increased soil Zn and Cd concentrations, as well as Cd concentrations in field pea (Krebs et al., 1998). Ten years of liming raised the soil pH by approximately one unit and resulted in Cu, Zn and Cd concentration decreases in seeds and crop residues of field pea. The study also found that plant uptake and solubility of Zn, Cu, and Cd continued to be higher in pig manure-treated soils than in control plots after the applications ceased.

Solubility and leachability

Solubility and leachability of trace metals after application of manure is of concern for water quality. The chemical forms rather than the total concentrations are important for potential leachability and environmental pollution (He et al., 1992; Hsu and Lo, 2000). Metal leachability may be modified due to changes in the quantity of dissolved organic matter and pH.

A study was conducted to assess the leachability and environmental hazard of Cu, Mn, and Zn in swine manure composts (Hsu and Lo, 2000). The composts were enriched with Cu (154-1380 mg/kg), Mn (239-976 mg/kg), and Zn (372-2840 mg/kg). A series of extraction schemes were used to determine salt extractable metals and their distribution at various pH levels. The results showed that the Cu, Mn, and Zn contents of composted manure collected from different pig farms varied substantially. The chemical form and extractability of Cu, Mn, and Zn were independent of total content in the composted manure. The distribution of Cu, Mn, and Zn in the various soil extractions revealed that the potential leachability of these elements is generally in the order Zn > Mn > Cu. Dissolution of organic C resulting from pH changes substantially modified Cu leachability, but had little effect on the Mn and Zn leachability. Dissolution of organic matter generally increases with increase in pH. The authors concluded that most of the Cu in these composts is bonded to the soil organic matter. Furthermore, the potential leachability of Cu, Mn, and Zn are likely low although the composts may contain high amounts of these elements.

In another study the effect of swine manure were reported on the solubility of Cd and Zn in soils Almås et al. (2000) suggested that treatment increased the solubility of both Cd and Zn by metal-

organic complex formation. Soluble organic acids from manure can increase the solubility of Cd and Zn and lead to increases in leaching or increased runoff losses.

Fluidized bed combustion (FBC) materials, an ash byproduct of clean coal burning, can be used to stabilize dairy feedlot surfaces, thus decreasing the mobility of trace elements. Elrashidi et al. (1999) investigated the use of FBC residue to stabilize dairy feedlot surfaces, and reduce leaching of trace elements in a laboratory column experiment with dairy manure. The columns were subjected to 10 weekly leaching with distilled water. They FBC decreased leachate concentrations by 5.6 to 100%, for most elements (e.g., P, N, K, calcium, aluminum, silicon, iron, manganese, copper, zinc, lead, cadmium, cobalt, chromium, nickel, arsenic, and selenium. Several mechanisms for this observation were proposed: (i) formation of insoluble metal-organic complexes; (ii) sorption of insoluble organic and inorganic species on mineral surfaces; and (iii) precipitation of insoluble inorganic species. However, the FBC increased concentrations of B (235%), S (47.3%), and Mg (36.5%) due to the high concentration of these elements in mobile forms in the FBC.

Reduction of trace elements in manure

Mohanna and Nys (1999) conducted a study on chicken nutrition to determine the effect of decreasing dietary Zn content on growth, plasma, tibia and whole body Zn concentrations, immune function, enzyme activity, Zn body retention and Zn concentration in excreta. Results indicated that lowering dietary Zn supplementation did not affect enzyme activity or immune response of the chicks. Furthermore, a reduction in dietary Zn content from 190 to 60 mg/kg . decreased Zn concentration in chicken manure by 75%. The authors concluded that lower dietary Zn supplementation could reduce excessive Zn concentration in manure and the risks of phytotoxicity in the soil.

Surface Water Impacts of Manure P

The initial impact of manure on the concentration of P in runoff is greatly influenced by the solubility of the P in the manure. Sharpley, and Moyer (2000) investigated P solubility in samples of dairy manure, swine manure, poultry litter, and composted manure. Release of P was measured during simulated rainfall (70 mm/h for 30 min) in the laboratory (10 Mg/ha manure application rate). During a 30-min rainfall, the dissolved inorganic P concentration in leachate from manure and compost ranged from 34 mg/L (poultry compost) to 75 mg/L (poultry manure). During five simulated rainfall events, the total quantity of P leached from dairy manure, poultry manure, and poultry compost and litter, and dairy compost and swine slurry were 58%, 21%, 20%, and 15%, of the P in the manure. The amount of dissolved inorganic or organic P leached from of each material was significantly correlated to the respective water extractable inorganic ($r^2 = 0.98$) or organic P ($r^2 = 0.99$). The authors suggested that water extractable P may be used to estimate the potential for land-applied manures or composts to enrich leachate and surface runoff P.

Soil test P and other forms of extractable soil P are being used to evaluate the potential of soil P to deliver runoff P to surface waters. However, it is not clear which soil tests better correlate

with the potential P pollution. Hooda et al. (2000) determined whether soil release of soil P to runoff could be predicted either by soil test values, sorption-desorption indices, or the degree of soil saturation with phosphorus. Degree of soil saturation with phosphorus is the ratio extractable P to the total quantity of P a soils can adsorb. Five methods were compared for predicting potential P release to runoff. The results of this study clearly showed that the amount of P that can potentially be released to runoff water had little relationship with either total soil P content or P sorption capacity. The most important property relating to water soluble P was the degree of soil saturation with phosphorus.

A series of experiments were conducted over 10 years to evaluate soil test methods using 163 Vermont and New York soils (Magdoff, et al., 1999). Phosphorus availability to plants, the equilibrium soluble P concentration, and $CaCl_2$-extractable P were all more closely related to ammonium acetate extractable soil test P (NH_4OAc-extractable) than soil test P extracted by solutions containing fluoride (such as Mehlich 3, Bray 1). The authors concluded NH_4OAc-extractable P could be a good parameter to be included as part of an index that ranks soils according to their potential for contribution of P to runoff.

A UK study compared the effects of high rates of liquid cattle manure with inorganic fertilizer and showed that both can increase runoff P concentrations when surface applied (Withers et al., 2001). Liquid cattle manure (186 kg P/ha) or triplesuperphoshate (330 kg P/ha) were applied to a cereal crop on a silty clay loam soil, over a 2 years period. In the first runoff events after surface application in the spring dissolved inorganic P concentrations in the control plots were only 0.1 mg/L compared to 6.5 mg/L for the fertilizer and 3.8 mg/L for the manure. Runoff dissolved P concentrations were typically <0.5 mg/L across all plots, and particulate P was the dominant P form in the runoff, for fertilizer and manure incorporated in the fall. These results show that for P in runoff the method of P application is more important than the source of P.

In soils subjected to flooding or with a fluctuating shallow water table the reducing conditions induced by moisture saturation can increase the availability of P in runoff. In a laboratory study topsoil and 2 subsoils were amended with 4g/kg poultry litter and flooded for 28 days then drained for 14 days (Vadas and Sims, 1999). The P adsorption capacity decreased in all soil horizons under these conditions. For the poultry litter amended topsoils soluble P concentrations were consistently higher than in unamended topsoils.

Poultry litter applications to pasture land result in high P concentrations in runoff. This was shown in a simulated rainfall study (Sauer et al., 2000). In a runoff event one month after application of 4.5 Mg/ha of poultry litter the dissolved inorganic P concentrations in the runoff averaged 2.20 mg/L.

The risk of transport of both P and N from pastured land into streams can be reduced by exclusion of animals from the land near streams. This was demonstrated in a study that involved excluding dairy cows from the riparian corridor along a small North Carolina stream (Line et al., 2000). The data following exclusion fencing indicated 33, 78, 76, and 82% reductions in weekly nitrate plus nitrite N, total nitrogen (TKN), total phosphorus (TP), and sediment loads,

respectively. The reductions in mean weekly loads were significant ($P<0.05$) for all pollutants except nitrate plus nitrite.

Chemical additives can reduce the bioavailablity and leaching of manure P. A study was conducted to determine the effects of alum, caliche (ground arid region soil material), and class C fly ash on extractable P concentrations in stockpiled and composted cattle manure at rates of 0, 0.10, 0.25, and 0.50 kg/kg manure (Dao, 1999). The caliche, alum, and fly ash reduced water-extractable P in stockpiled manure by 21, 60, and 85% and by 50, 83, and 93% in composted manure at the 0.1 kg/kg rate. Alum and fly ash also significantly reduced soil test P (Bray-P) concentrations by 75 and 90% in stockpiled and composted manure, respectively, and >90% at higher rates. Alum contains aluminum that can become a problem in acid soils and fly ash contains a large number of trace elements. The possible negative effects of these materials must be considered before they are recommended for treatment of manure. Caliche is mostly limestone and should not pose any environmental risk.

Ground Water Impacts of Manure P

Although the risk of P to subsurface water quality is generally considered to be low, some investigators have measured manure P movement to ground water. Two to three annual applications of cattle slurry to grass and grass-clover pastures at about 25 kg P/ha/y resulted in some soluble in P in lysimeter drainage water (Hooda et al. 1999). After 9 years, an average increase of 1.0 mg Olsen-P/kg/y (soil test P) was observed in the topsoil (0-10 cm). The total soluble phosphorus concentrations in the drainage water ranged from 0.45 to 0.79 mg P/L. These P concentrations in drainage water were much higher than previous estimates. Subsoils have a much higher P-adsorption capacity than topsoil, which should keep soluble P concentrations low. The high measured soluble P indicates that significant P transport in soils can occur through preferential hydrological flow pathways in the soil (cracks and crevasses).

Manure impacts on soluble P in shallow ground water were observed after 10 years of intensive swine manure application to a Coastal Plain spray field at disposal rates (Novak et al., 2000). After 10 years the topsoil had very high soil test P (377-435 mg P/kg Mehlich 3) while in the control soil the soil test P was <10 mg P/kg. Groundwater dissolved P concentrations were initially very low (<0.040 mg P/L) in the wells installed around the spray field. After 10 years the measured values increased substantially to 0.04-0.48 mg P/L.

A laboratory study from Denmark suggests that the localized high concentrations of manure under a manure patches in a pasture can result in localized leaching of P (Magid, et al, 1999). Manure was placed on top of, or incorporated in, the soil of 40 cm intact columns. Incorporation or surface application of manure resulted in a 10- to 20-fold and 100- to 200-fold increases in soluble inorganic P, respectively. The authors suggested that special attention should be given to drained pastures, where the important P affecting drainage water may not be the soil P, but the P in manure patches on the surface. This P could be transported directly through preferential hydrological pathways in the soil.

Ground Water Impacts of Manure N

Nitrate leaching to ground water is a problem when excess plant available N is added regardless of whether the source is organic or inorganic fertilizer. One of the areas where there is a concern for nitrate levels in shallow ground water is on a sandy out-wash soils in West-Central Minnesota. To better understand NO_3^- contamination of ground water Puckett et al. (1999) did a mass-balance budget of N cycling study for this intensive agricultural area. The budget was developed using ground water data collected throughout the 212 km^2 study area. They looked at all forms of soil applied N including manure. On a regional scale the N sources were fertilizer, biological fixation, atmospheric deposition, and animal feed. The N sinks in the model were crop harvests, animal product exports, volatilization from fertilizer and manure, and denitrification. Denitrification, estimated by adjusting its value so the predicted and measured concentrations of NO_3^- in ground water agreed, appeared to remove 50% of the nitrate leached below the soil profile. The authors did not find a general threat to ground water. Animal agriculture seems to be of no threat to ground water, because of the small size and dispersed nature of animal production in the area. The regional approach, however, does not account for local problems that can occur from over application of N.

Denitrification process that can convert nitrate-N into gaseous products, and removing nitrate from soil and water. Denitrification can be a major pathway of N loss from soil. Marshall et al. (1999) quantified N loss via denitrification from tall fescue pastures following chicken litter application (70 kg of available N/ha) at Alabama, Georgia, and Tennessee. Total losses of N gases were all <6 kg/ha during 150 days after application, representing a loss of <5% of total N applied. The authors concluded that gaseous N losses from soil in the southeastern US are not significantly increased by the addition of chicken litter.

Nutrient Management

Recently, manure nutrient management has become a major focus in efforts to sustain or improve environmental quality while sustaining agriculture production. Today, the agronomic and economic factors of nutrient planning remain central, but nutrient planning also requires environmental impact consideration (Beegle et al., 2000). Traditionally, farmers were concerned with nutrient management to optimize economic return from crop production by applying inorganic fertilizer without giving credit for nutrients applied in manure (Schmitt et al., 1999). This practice has resulted in nutrient accumulation in the soil that exceeds agronomic requirements for crop production. Consequently, nutrients leach to groundwater or runoff to surface water leading to contamination, eutrophication and hypoxia of the water bodies (Burkart and James, 1999; Hession and Storm, 2000).

Animal agriculture in the United States is continuing to increase as illustrated in Figure 1 by Aschmann et al. (1999). Also, the size of animal operations has increased. For example, over 97% of the poultry produced in the US comes from operations of 100,000 animals or more (Aschmann et al., 1999). Deterioration of soil, water, and air quality in many localized areas has

been linked to the intensification of animal production (USDA-EPA, 1999). Specialization and intensification of agricultural systems has led to P accumulation in excess of crop needs in some areas (Haygarth and Sharpley, 2000; Sims et al., 2000).

Nutrient losses from agricultural nonpoint sources are a key component of surface water impairment in the United States. Nitrogen is the primary pollutant problem in many agricultural areas; however, development of management practices that reduce phosphorus loading is becoming more important in many watersheds because phosphorus is often the limiting nutrient for fresh water eutrophication. The very recent literature has concentrated on nutrient management planning and implementation, phosphorus-based versus nitrogen-based manure management, and nutrient dynamics modeling.

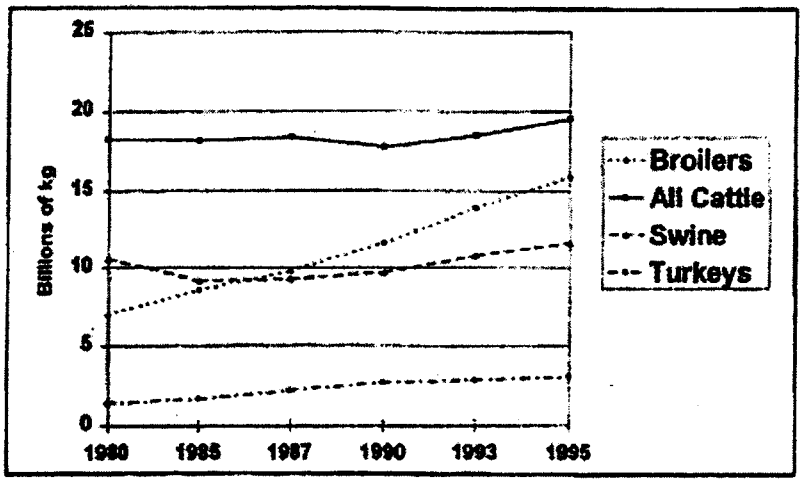

Figure 1. U.S. meat production 1980-1995. Source: USDA-NASS 1997.

Timing nitrogen availability

The goal of attaining high yields while protecting soil and water quality requires matching soil inorganic N supplies with crop N requirement over the cropping season. This was investigated using ENVIRON-GRO, a computer model, which accounts for N and irrigation management effects on crop yield and N leaching (Pang and Letey, 2000). Published data were used with the model to evaluate the multi year dynamics of N mineralization and N-uptake for corn and wheat when manure is added using N based recommendations. The results show that the N-uptake of the corn can exceed the cumulative mineralized N during part of the season. This deficit is caused by the very rapid uptake of N from knee high to tasseling. The deficit can reduce yield if no fertilizer topdress N is added. However, wheat has a low and flat N-uptake peak and manure N could better meet the N demand for wheat. The authors concluded that a crop with a very high maximum N-uptake rate, such as corn, would require inorganic N to meet peak demands if excessive N application is to be avoided.

Nutrient management planning

Nutrient management planning can reduce the N and P loss from farmland. This is illustrated in a case study conducted by VanDyke et al. (1999). Nutrient management planning was evaluated on four Virginia livestock farms using the Erosion Productivity Index Calculator (EPIC) model to calculate the effects of management changes. Changes in management practices with implementation of nutrient management are illustrated in Table 1.

Table 1. Changes in management practices with implementation of nutrient management plans (adapted from VanDyke et al., 1999)

Farm	Changes in management practices with plan
Southwest Dairy	Credit manure nutrients Reduce nitrogen fertilizer Split nitrogen applications Nitrate quick test
Shenandoah Valley Dairy	Install manure pit Apply manure to more land Credit manure nutrients Reduce commercial fertilizer applications
Southeast Swine	Construct manure storage Apply manure to all cropland Inject manure applied to corn Credit manure nutrients Reduce commercial fertilizer applications
Piedmont Poultry	Construct 2 litter sheds and mortality composter Reduce litter applications Compost poultry mortalities Sell excess litter

The results indicate that after the farms implementation of a nutrient management plan, average annual nitrogen and phosphorus losses were reduced by 23 to 45%, and 0 to 66%, respectively (Table 2). Study results suggest that the magnitude of nutrient loss reductions on livestock farms is contingent on unique farm characteristics, fertilizer management practices, and weather. Generally, fields with poor quality soils or steep slopes have much greater nutrient losses, particularly when manure provides some of the crop nutrient requirements. Nutrient management results in greater reductions of nutrient losses on these soils compared to soils with less risk of runoff and leaching.

Calculated annual farm income for these farms increased by $395 to $4,593, primarily due to reduced fertilizer expenses associated with crediting of manure nutrient content and increased sales of poultry litter to nearby farms (Table 3). However, it cannot be concluded that nutrient management planning increases income on all farms, due to the small number of farms analyzed for this study.

Table 2. Average annual per hectare nitrogen and phosphorus losses before and after nutrient management plan (adapted from VanDyke et al., 1999)

	Southwest Dairy	Shenandoah Valley Dairy	Southeast Swine	Piedmont Poultry
		kg/ha		
Total N loss before plan[1]	53	69	46	24
Total N loss after plan[1]	39	46	25	19
Total N loss reduction[2]	14 (27%)	23 (33%)	21 (45%)	6 (23%)
Range of N loss reduction by field	10 to 15	3 to 105	-3 to 58	1 to 12
Total P loss before plan[3]	8	18	3	9
Total P loss after plan[3]	8	14	1	6
Total P loss reduction[3]	0 (0%)	4 (23%)	2 (66%)	2 (32%)
Range in P loss reduction by field	0	-2 to 27	-1 to 10	0 to 7

[1/] Average per hectare losses are weighted averages based on the acreage of each soil and crop rotation on farm.
[2/] Totals may be affected by rounding.
[3/] P losses are nearly all attached to eroded sediment.

Table 3. Annual economic impact of nutrient management[1] (adapted from VanDyke et al., 1999)

Farm	Additional costs ($)	Reduced income ($)	Additional income ($)	Reduced costs ($)	Net income change[2] ($)
Southwest Dairy	2,270	0	0	2,665	395
Shenandoah Valley Dairy	7,643	0	0	12,236	4,593
Southeast Swine	15,041	2,195	0	20,251	3,015
Piedmont Poultry	3,020	562	2,240	3,639	2,297

[1/] Costs including annualized cost of investments and operator labor.
[2/] Net income change equals additional income plus reduced costs minus additional costs minus reduced income.

Fertilizer management practices, farm characteristics, and weather influenced nutrient losses within and across farms. Manure storage, manure nutrient crediting, and proper timing of manure applications are critical in reducing nutrient losses and increasing cost savings. The construction of storage allows flexibility to apply manure when and where it will be most beneficial to crops, thus reducing fertilizer applications, costs and nutrient losses.

Phosphorus-based manure management

Generally, manure is applied to agricultural land based on nitrogen (N) recommendations, to meet N requirements of the crop. This can result in over-application of phosphorus (P) and its accumulation in soil and consequent runoff to surface waters or possible leaching to shallow ground water.

Eghball and Power (1999) conducted a four-year study to evaluate effects of P- and N-based manure and compost application on corn yield, N and P uptake, soil P level, and weed biomass on a silty clay loam soil under rainfed conditions in Nebraska. Composted and noncomposted

beef cattle feedlot manures were applied to supply the N or P needs of corn for either a one- or two-year period. Phosphorus-based manure or compost treatments also received additional N fertilizer as required.

In all four years, manure or compost application increased corn grain yield as compared with the unfertilized control. Manure or compost application resulted in similar grain yields to those of the fertilizer treatment, and yields for biennial and annual applications were similar when applied for an expected grain yield of 9.4 Mg/ha. Annual phosphorus-based manure or compost application resulted in similar grain yields to those for N-based treatments, and significantly lower soil P levels after four years of application. Biennial phosphorus-based manure or compost application also resulted in similar grain yields, but had greater soil P buildup than annual treatments because of greater amounts of P applications.

Estimated N availability was 40% for manure and 15% for compost in the first year and was 18% for manure and 8% for compost in the second year after application. Nitrogen use efficiency (plant N uptake divided by added N) was greater for manure than compost application. Phosphorus use efficiency was 2.5 to four times greater for annual P-based manure or compost application than N-based application.

Eghball and Gilley (1999) also determined the effects of simulated rainfall on runoff losses of P and N, and pH following the application of manure and compost. Manure, compost, and fertilizer were applied to the soil surface of no-till fields or disked to 8 cm following sorghum and winter wheat, at rates required to meet the N or P requirements for corn production. Generally, total and particulate P concentrations in runoff were less after wheat than sorghum, and were less for the no-till than the disked treatment. Application of manure to meet the N requirement of the crop resulted in more P in the runoff than application to meet the P need of the crop. Fertilization resulted in similar P concentration in the runoff as compost or manure application. Dissolved P constituted 91% of the bioavailable P in runoff. These results from Nebraska suggest that the method of application of manure is important in determining P in runoff.

In another study, Whalen and Chang (2001) investigated the P balance of cultivated soils under barley production with long-term annual manure amendments. Nonirrigated soils at the study site in Alberta, Canada, received 0, 30, 60, or 90 Mg/ha manure (wet weight basis) while irrigated plots received 0, 60, 120, and 180 Mg/ha annually for 16 years. All of these are disposal rates, much in excess of crop needs. The amount of P removed in barley grain and straw during the 16-year period was between 5 and 18% of the cumulative manure P applied. Over 16 years, as much as 1.4 Mg/ha of added P (180 Mg/ha/yr treatment) was not recovered in the top 150 cm of soil in the irrigated plots and was probably lost lower depths. In nonirrigated plots, there was a balance between P applied in manure and P recovered in crops and soils (to the 150-cm depth).

The total P concentrations were 1.9 to 5.2 Mg/ha greater in the amended than in the control plot, in irrigated soil to the 150 cm depth. The manure application rates of 30 Mg/ha and 60 Mg/ha provided five to six times more P than was recommended for barley production on nonirrigated and irrigated soils. These results show there is a risk of P movement to ground water with greatly excessive manure applications in irrigated plots.

Watershed ecosystem nutrient dynamics models

Nutrient dynamic models have been used to describe how nutrients are cycled and stored, and to assess the effects of management practices on nutrients transported into and out of a watershed. Osei et al. (2000) performed computer simulations to assess the impact of various management practices on phosphorus losses from dairy farms in a watershed in north central Texas. The results indicated that manure management based on crop P needs, in livestock intensive watersheds, offers sound management for reducing nonpoint source P loading. In some watersheds, where excessive P losses or soil P buildup from previous land uses, greater P reduction to less than crop removal is required. Composting all solid manure can reduce P loads by removal from the watershed. The cost to dairy producers was estimated to be about 27-30% of their baseline net returns, which could be debilitating for smaller dairies. Various options for financing composting alternatives for the dairies could be implemented to help alleviate the financial burden on farmers. The choice for each watershed depends on such key factors as available land area and the load reduction sought.

The USDA Natural Resources Conservation Service (NRCS) Water Science Institute (WSSI), in cooperation with a number of partners, has initiated a project to promote nutrient cycling within watersheds. The project has developed a watershed-scale model to examine dynamic phosphorus (P) flows into, out of, and within watersheds using a mass balance approach. The model is called Watershed Ecosystem Nutrient Dynamics (WEND). The WEND model tracks watershed P balances over time within each of the several land-use sectors. Figure 2 is a diagram of the P compartment-flux for storing and cycling P in agriculture watersheds. The model was used by Aschmann et al. (1999) to study long-term impacts of various strategic policy decisions (status quo, increased rate of development, and increased conservation polices) on P cycling in the Winooski River Watershed in Vermont. The agriculture in the watershed is primarily dairy. The model showed P losses into the surface water were significantly reduced under the conservation scenario, while water quality was impaired under the status quo, and development scenarios projected over 80 years. It was concluded that the WEND model could help in providing the framework needed to create sustainable nutrient strategies for watersheds.

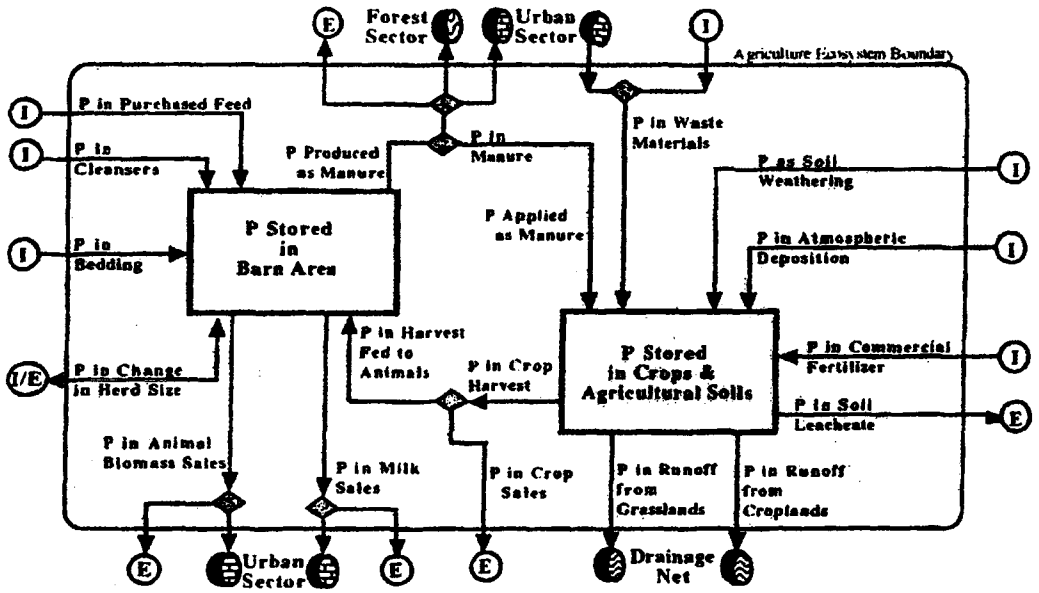

Figure 2. Phosphorus nutrient compartment of the Watershed Ecosystem Nutrient Dynamics model.

Manure nutrient guidelines, regulations and implementation

Recently concerned citizens, environmentalists and regulatory agencies have focused on manure nutrient management and deposition from concentrated feedlots. Development and use of nutrient management planning has the potential to improve the utilization of nutrients and minimize contamination of the environment. The establishment and enforcement of manure nutrient regulations will alter the future of livestock production (Meyer and Mullinax, 1999). Proposed legislation may impose monitoring and record keeping on the livestock operators. Livestock operations need to comply with regulations to minimize environmental liability and to operate.

Proposed legislation and strategies (a simplistic approach to nutrient management) may provide a false sense of security regarding environmental preservation or restoration. The challenge is to create a policy that will allow flexibility for regions and states and still provide a reasonable guideline to minimize contamination. Also proposed strategies should provide a component that accommodates specific conditions of particular farming operations or nature (Beegle et al., 2000).

Successful implementation of nutrient management policy must involve full participation of a broad range of stakeholders (Beegle et al., 2000). Major stakeholders are the farmers, allied agri-industry, public agencies, regulators, policymakers, environmental groups, and the consumer. Stakeholders are critical for developing sound objectives for the nutrient management effort. Also research, education and financial and technical assistance are critical for the success of

nutrient management programs (Jackson et al., 2000). Cost-share funds or tax incentives may be critical for adoption of nutrient management plans (VanDyke et al., 1999).

Summary

- Organic farms, which generally utilize manure rather than commercial fertilizer, have higher soil quality compared to conventional farms as indicated by greater organic C, total N, microbial biomass C and N, and soil respiration, and by pH values closer to neutral, lower bulk density, and higher available-water holding capacity.

- Manure from animals receiving feed additives to improve animal health can contain high concentrations of trace elements. The quantity of these elements in manure depends on the type of manure or source (poultry vs. swine). The potential exists for some of the trace elements to eventually accumulate to phytotoxic levels in soils.

- In some cases reduction of trace metal dietary supplements, with a consequent reduction of concentrations in the manure, is possible without reducing animal health.

- The method of P application is more important that the source of P in predicting the loss of P in runoff. Manure and inorganic fertilizer P are both problematic when surface applied.

- Adding fly ash, alum or ground limestone can reduce the leachability and bioavailability of manure P. Fly ash must be used with care because of potential adverse environmental effects from trace elements. Alum is a possible problem on acid soils.

- Transport of manure P to ground water may be more of a problem than previously thought. Some transport can occur though macropores in soils without interaction with subsoil particles that can adsorb P.

- The fraction of P saturation and soil test P are good parameter for prediction of soil P solubility.

- The adoption of management practices outlined in nutrient management plans could result in significant reductions in N and P losses. Management planning can increase net farm income but the income increases may be insufficient to cause voluntary adoption of nutrient management planning.

- Involving local experts and watershed residents in the development process of nutrient management policy allows stakeholders to understand the complexity and interactions among land-use sectors within their own watersheds.

References

Almås, Å. R. M. B. McBride, and B. R. Singh. 2000. Solubility and lability of cadmium and zinc in two soils treated with organic matter. Soil Sci. 165:250-259.

Aschmann, S. G., D. P. Anderson, R.J. Croft, and E. A. Cassell. 1999. Using a watershed nutrient dynamics model, WEND, to address watershed-scale nutrient management challenges. J. Soil Water Conserv. 54:630-635.

Beegle, D.B., O.T. Carton, and J.S. Bailey. 2000. Nutrient management planning: Justification, theory, practice. J. Environ. Qual. 29:72-79.

Burkart, M.R. and D. E. James. 1999. Agricultural-nitrogen contributions to Hypoxia in the Gulf of Mexico. J. Environ. Qual. 28:850-859.

Dao, T. H. 1999. Coamendments to modify phosphorus extractability and nitrogen/phosphorus ratio in feedlot manure and composted manure. J. Environ. Qual. 28:1114-1121.

Eghball, B. and J.F. Power. 1999. Phosphorus- and nitrogen-based manure and compost applications: Corn production and soil phosphorus. Soil Sci. Soc. Am. J. 63:895-901.

Eghball, B., and J.E. Gilley. 1999. Phosphorus and nitrogen in runoff following beef cattle manure or compost application. J. Environ. Qual. 28:1201-1210.

Elrashidi, M.A., V.C. Baligar, R.F. Korcak, N.Persaud and K.D. Ritchey. 1999. Chemical composition of leachate of dairy manure mixed with fluidized bed combustion residue. J. Environ. Qual. 28:1243-1251.

Flessa, H. and F. Beese. 2000. Laboratory estimates of trace gas emissions following surface application and injection of cattle slurry. J. Environ. Qual. 29:262-268.

Giusquiani, P.L., L. Cocezzi, M. Businelli, and A. Macchioni. 1998. Fate of pig sludge liquid fraction in calcareous soil: Agricultural and environmental implications. J. Environ. Qual. 27:364-371.

Haygarth, P. M., and A. N. Sharpley. 2000. Terminology of phosphorous transfer. J. Environ. Qual. 29:10-15.

He, X.T., S.J. Traina, and T.J. Logan. 1992. Chemical properties of municipal solid waste compost. J. Environ. Qual. 21:318-329.

Hession, W.C. and D.E. Storm. 2000. Watershed-level uncertainties: implications for phosphorus management and eutrophication. J. Environ. Qual.29:1172-1179.

Hooda, P.S. M. Moynagli, I.F. Svoboda, A.C. Edwards, H.A. Anderson, and G. Sym. 1999. Phosphorus loss in drainflow from intensively managed grassland soils. J. Environ. Qual. 28:1235-1242.

Hooda, P.S., A.R. Rendell, A.C. Edwards, P.J.A. Withers, M.N. Aitken, and V.W. Truesdale. 2000. Relating soil phosphorus indices to potential phosphorus release to water. J. Environ. Qual. 29:1166-1171.

Hsu, J., and S. Lo . 2000. Characterization and extractability of copper, manganese, and zinc in swine manure composts. J. Environ. Qual.. 29:447-453.

Hue, N.V. and D.L. Licudine. 1999. Amelioration of subsoil acidity through surface application of organic manures. J. Environ. Qual. 28:623-632.

Jackson, L.L., D.R. Keeney, and E.M. Gilbert. 2000. Swine manure management plans in north-central Iowa: nutrient loading and policy implications. J. Soil and Water Conserv. 55:205-212.

Krebs, R., S. K. Gupta, G. Furrer, and R. Schulin. 1998. Solubility and plant uptake of metals with and without liming of sludge-amended soils. J. Environ. Qual. 27:18-23.

Liebig, M. A., and J. W. Doran.1999. Impact of organic production practices on soil quality indicators; J. Environ. Qual. 28:1601-1609.

Line D. E., W.A. Harman, G.D. Jennings, E.J. Thompson and D.L. Osmond. 2000. Nonpoint-source pollutant load reductions associated with livestock exclusion. J. Environ. Qual. 29:1882-1890.

Looker, D. 1997. Organic goes mainstream. Successful Farming. 95(7):45-46.

Magdoff, F.R., C. Hryshko, W.E. Jokela, R.P. Durieux and Y. Bu. 1999. Comparison of phosphorus soil test extractants for plant availability and environmental assessment. Soil Sci, Soc. Am. J. 63:999-1006.

Magid, J., M.B. Jensen, T. Mueller and H.C.B.Hansen. 1999. Phosphate leaching responses from unperturbed, anaerobic, or cattle manured Mesotrophic sandy loam soils. J. Environ. Qual. 28:1796-1803.

Marshall, S. B., M. L. Cabrera, L. C. Braun, C. W. Wood, M.D. Mullen, and E. A. Guertal. 1999. Denitrification from fescue pastures in the southeastern USA fertilized with broiler litter. J. Environ. Qual. 28:1978-1983.

Meyer D. and D.D. Mullinax.1999. Livestock nutrient management concerns: regulatory and legislative overview. J. Anim. Sci. 77, Suppl. 2/J. Dairy Sci. Vol. 82, Suppl. 2:51-62.

Mohanna, C. and Y. Nys. 1999. Effect of dietary zinc content and sources on the growth, body zinc deposition and retention, zinc excretion and immune response in chickens. British Poultry Sci. 40:109-114.

Nash D., M. Hannah, D. Halliwell and C. Murdoch. 2000. Factors affecting phosphorus export from a pasture-based grazing system. J. Environ. Qual. 29:1160-1166.

Novak, J.M., D.W. Watts, P.G Hunt,.and K.C Stone. 2000. Phosphorus movement through a coastal plain soil after a decade of intensive swine manure application. J. Environ. Qual. 29:1310-1315.

Osei, E., P.W. Gassman, R.D. Jones, S.J. Pratt, L.M. Hauck, L.J. Beran, W.D. Rosenthal, J.R. Williams. 2000. Economic and environmental impacts of alternative practices on dairy farms in an agricultural watershed. J. Soil Water Conserv. 55(4):466-472.

Pang, X. P., and J. Letey. 2000. Organic farming: challenge of timing nitrogen availability to crop nitrogen requirements. Soil Sci. Soc. Am. J. 64:247-253.

Puckett, L. J., T. K. Cowdery, D.L. Lorenz, and J.D. Stoner. 1999. Estimation of nitrate contamination of an agro-ecosystem outwash aquifer using a nitrogen mass-balance budget; J. Environ. Qual. 28:2015-2025.

Sauer, T. J., T.C. Daniel, D.J. Nichols, C.P.West P.A. Moore, K.P. Coffey and G.L Wheeler. 2000. Runoff water quality from poultry litter-treated pasture and forest sites. J. Environ. Qual. 29:515-521.

Schmitt, M.A., M.P. Russelle, G.W. Randall, and J.A. Lory. 1999. Manure nitrogen crediting and management in the USA: survey of university faculty. J. Prod. Agric. 12:419-422.

Sharpley, A., and B.Moyer. 2000. Phosphorus forms in manure and compost and their release during simulated rainfall. J. Environ. Qual.29:1462-1469.

Sims, J. T., A. C. Edwards, O. F. Schoumans, and R. R. Simard. 2000. Integrating soil phosphorus testing into environmentally based agricultural management practices. J. Environ. Qual. 29:60-71.

Tufft, L.S. and C.F. Nockels. 1991. The effects of stress, Escherichia coli, dietary EDTA, and their interaction on tissue trace elements in chicks. Poultry Sci. 70:2439-2449.

USDA and US EPA. 1999. Unified national strategy for animal feeding operations. March 1999. (http://www.nhq.nrcs.usda.gov/cleanwater/afo/index.html).

Vadas, P.A., and J.T. Sims. 1999. Phosphorus sorption in manured atlantic coastal plain soils under flooded and drained conditions. J. Environ. Qual. 28:1870-1877.

Vanderwatt, H.V.H, M.E. Sumner and M.L.Cabrera. 1994. Bioavailability of copper, manganese, and zinc in poultry litter. J. Environ. Qual. 23: 43-49.

VanDyke, L.S, J.W. Pease, D.J. Bosch and J.C. Baker. 1999. Nutrient management planning on four Virginia livestock farms: Impacts on net income and nutrient losses. J. Soil and Water Conserv. 499-504.

Whalen, J.K and C. Chang. 2001. Phosphorus accumulation in cultivated soils from long-term annual applications of cattle feedlot manure. J. Environ. Qual. 30:229-237.

Withers, P.J.A., S.D. Clay and V.G. Breeze. 2001. Phosphorus transfer in runoff following application of fertilizer, manure, and sewage sludge. J. Environ. Qual.30:180-188.

Wood, B.H, C.W.Wood, K.H.Yoo, K.S.Yoon, and D.P. Delaney. 1999. Seasonal surface runoff losses of nutrients and metals from soils fertilized with broiler litter and commercial fertilizer. J. Environ. Qual. 28:1210-1218.

Yamulki, S., S.C. Jarvis, and P. Owen. 1999. Methane emission and uptake from soils as influenced by excreta deposition from grazing animals. J. Environ. Qual. 28:676-682.

What are the pathways for exposure to pathogens?

The use of untreated manure and manure-contaminated water on edible crops has been identified as a potential source of pathogen contamination. However, it is often difficult to separate contamination due to manure from improper personal hygiene, unsanitized packinghouses, contamination by handlers, poorly or unsanitized transportation vehicles, and inadequate refrigeration during transport (Brackett, 1999). All of these are pathways for exposure to pathogens.

Contaminated soil or manure on the surface of fruits and vegetables
There is evidence of pathogen transmission to humans through ingestion of soil-contaminated fruits and vegetables and drinking water (Burnett et al., 2000; Dingman, 2000; Jones, 1999; Kudva et al., 1998).

Contaminated water (irrigation, hydroponic, rain, dew) on fruits and vegetables
Contamination of radish sprouts after exposure to *E. coli* O157:H7 inoculated water was carried out by Hara-Kudo et al. (1997) in the laboratory. They found that the edible parts became heavily contaminated when they were grown from seeds soaked in *E. coli* O157:H7 inoculated water. It was concluded that contamination of the edible parts of radish sprouts could pose a serious illness if the seeds or hydroponic water are contaminated with the bacterium (Hara-Kudo et al., 1997).

Burnett et al. (2000) suggested that the tissues of fruits and vegetables could be infiltrated by pathogens when produce surfaces come in contact with cells suspended in the contaminated water. In the field this can occur when rain, irrigation or dew collects on the surface of produce, or the fruit falls on contaminated ground (Burnett et al., 2000). Dingman (2000) reported that apples obtained from the ground (wind fallen apples) are highly susceptible to *E. coli* O157:H7 although no direct evidence linking dropped apples to fecal contamination of cider has been presented.

What factors contribute to pathogen transfer from manure to crops?

There are several factors that contribute to the transfer of pathogens from manure to edible crops. These include the presence of pathogens in animals (and consequently in manure), contamination of water by manure, the survivability of pathogens in manure and manure-contaminated water, and the survivability of pathogens on edible crops.

Presence of pathogens in animals
Pathogen proliferation on the farm depends on the livestock health and how the cattle are managed.

Age of the animal

Young calves are more likely to become infected with pathogens than are cows. Very few mature cows shed *E. coli* and in one study calves older than 6 months rarely tested positive (Pell, 1997).

Herd size

The same study found that calves were more likely to have *Cryptosporidium parvum* in herds with more than 100 cows than calves in smaller herds (Pell, 1997).

Animal housing

The study also found that calves born in individual pens were less likely to become infected than those in groups at calving. Those calves housed in pens in which only the bedding was removed were twice as likely to be become infected as those in pens that were washed (Pell, 1997).

Type of animal feed

Use of more natural animal feeds (such as pasture) and well-fermented silage can reduce pathogen infections (Pell, 1997). Well-fed cattle are less conducive to growth of *E. coli* O157:H7 in their gastrointestinal tracts than are cattle deprived of feed. Weaned calves (less than 24 months old) are more likely to shed *E. coli* O157:H7 than milk-fed calves (Pell, 1997). Dairy cattle fed with poultry litter without drying are more likely than cattle fed on composted litter to be infected with pathogens (Jeffrey et al., 1998).

Season

Fecal pathogen excretion in cattle appears to be seasonal, with the highest rate occurring in spring and late summer (Chapman et al., 1997). The seasonal patterns may be related to milk flushes and changes in cattle reproductive hormones, and stresses or changes in diet and water source (Jones, 1999). Increases in fecal coliform concentrations, an indicator of fecal pathogens, in waters of an upland area of northern England coincided with lambing and increased stocking density during summer (Hunter et al., 2000).

Contamination of water by manure

Surface or ground water contamination is possible from poor manure storage and application. Runoff from manure piles can carry pathogens to surface or underground water, especially in areas with karst geology. Peterson et al. (2000) reported that during a winter recharge event in mantled karst aquifers in northwest Arkansas, fecal coliform, *Escherichia coli* were present in five springs. Furthermore, they suggested that the fecal coliform, and *Escherichia coli* were moving through the mantled karst aquifer in a similar manner and provided evidence that applied manure is associated with the indicator bacteria.

Significant numbers of *Cryptosporidium. parvum* oocysts have been identified in Northern American surface waters (LeChevallier et al., 1991; Rose et al., 1991). Rose et al. (1991) also reported *C. parvum* oocysts in well water. The feces of infected farm animals is a suspected source of ground water contamination, either by subsurface or overland transport through highly permeable soils (Mawdsley et al., 1995; 1996a, b) or through drainage tiles (Kemp et al., 1995). Brush et al. (1999) studied transport of *C. parvum* from calf feces through saturated columns packed either with glass beads, coarse sand or shale aggregates. They suggested that oocysts could travel significant distances in both subsurface and overland flow.

In Southeast Minnesota, manure piles near wells or on karst topography could result in water contamination; in the Red River Valley, manure piles or manure-applied fields that are on a floodplain could result in water contamination.

Survivability of pathogens

Survival during storage, handling and treatment

Pathogen survival is affected by the storage temperature, exposure to oxygen, pH, dry matter content, age, source, and chemical composition of the manure, as well as by microbial characteristics. In general, pathogens cannot survive high temperatures and low moisture levels. It is possible that residual viable populations of pathogens are supported by the slow release of nutrients from the breakdown of organic matter and the utilization of substrates released from dying cells within the storage unit. Spore-forming bacteria (such as *Cl. perfringens* and *B. anthratics*) can survive for a long time in a harsh environment by producing endospores (Pell, 1997).

Stehman et al. (1996) outlined the following factors that limit microbial survival (increase death rate) in storage or when spread on the land:

- sunlight (UV radiation),
- freeze/thaw cycles,
- freezing,
- high temperatures,
- high or low pH,
- antibacterial and antiviral compounds,
- oxygen levels, and
- dryness.

Unfortunately, current manure storage systems contain all of the favorable environmental characteristics for pathogen survival and pathogen decrease is primarily slow for some organisms.

<u>Manuals used for storage design ignore pathogens</u>

Design manuals generally do not discuss pathogen survival in detail. For example, in MidWest Plan Service's (MWPS, 2001) new 91-page booklet on *Manure Storages*, there is no mention of pathogens. The Natural Resource, Agriculture, and Engineering Service (NRAES, 1999) publication *Earthen Manure Storage Design Considerations* contains a single paragraph on pathogens out of 90 pages. This single paragraph indicates that manure can be a health concern and that stored manure may still contain some pathogens and should be handled with due caution.

Natural Resource and Conservation Service *Agricultural waste management field handbook* (NRCS, 1992) indicates the presence of manure pathogens as shown by the existence of the indicator organism fecal coliform The only concern here is the water quality criteria by EPA under the 1986 Safe Drinking Water Act.

Survival of bacteria

Effect of Temperature: Decline in viable numbers of bacteria is temperature dependent as indicated by Kearney et al. (1993). They determined the T_{90} values (time taken for viable counts to decrease by one logarithmic unit, equivalent to 90% reduction) for the species reported in Table 5.

Table 5. The effect of temperature on the time (days) for decline of 90% of four bacteria species.

	4°C	17°C
Escherichia coli	>29.0	>29.0
Salmonella typhimurium	21.3	17.5
Yersinia. Enterocolitica	20.8	12.8
Listeria monocytogenes	>84.0	29.2
Campylobacter jejuni	>112.0	>112.0

Jeffrey et al. (1998) found no *Salmonella* in poultry litter samples where the internal temperature in the piles exceeded 40.2° C. This is because heat and ammonia produced from the degradation of uric acid in boiler litter are bacteriocidal for *Salmonella*. The mesophilic temperature range (20-45°C) is a more effective method of reducing pathogen numbers than the psychrophilic temperature range (<20°C). Few bacteria can withstand the heat generated during composting. Pell (1997) recommends that all parts of the compost pile reach and maintain a temperature of 60°C.

Effect of Aeration: Munch et al. (1987) compared bacteria at two temperature ranges (18-20°C and 6-9°C) for aerated and non-aerated slurry for both cattle and swine. For each species, T_{90} was always shorter in aerated than in non-aerated slurry (Table 6).

Table 6. The effect of aeration on the time (days) for decline of 90% of 4 species of bacteria.

	Aerated Slurry		Non-aerated Slurry	
	18-20°C	6-9°C	18-20°C	6-9°C
Salmonella typhimurium	2 – 7	7 – 18	7 – 21	21– 54
Yersinia. Enterocolitica	2	4 – 7	4 – 7	7 – 18
Staphylococcus. Aureus	4 – 10	4 – 10	5 – 21	10 – 119
Escherichia coli	3 – 21	10 – 25	10 – 32	12 – 126
Faecal streptococci	18 – 74	48 – 94	28 – 49	94 – 280

E. coli O157:H7 and many other pathogenic bacteria are facultative anaerobes that can survive and grow in environments with oxygen (aerobic) or without (anaerobic). However, as shown in Table 6, *E. coli* O157:H7 and many other bacteria generally survive longer under anaerobic conditions. Kudva et al. (1998) studied the survival and growth of *E. coli* O157:H7 in inoculated sheep and cattle manure, under various experimental and environmental conditions. They found that a sheep (ovine) manure pile which was periodically aerated by mixing remained culture

positive for 4 months, while a cattle (bovine) manure pile remained culture positive for 47 days. The pathogen survived for more than 12 months in a non-aerated cattle manure pile and for 2 months in an aerated manure pile. *E. coli* O157:H7 viability was reduced to less than 10 days in slurry. In the laboratory, *E. coli* O157:H7 survived best under anaerobic conditions at temperatures below 23° C, but it survived for shorter times than in manure piles in the field. The average times that feces from cattle and sheep remained culture positive were 30 and 50 days, respectively (Kudva et al, 1998).

Effect of management of anaerobic biogas digestion: Anaerobic digestion for production of biogas (methane) is one possible treatment method for reducing pathogens. Kearney et al. (1993) reported the T_{90} (day) means along with lower and upper limits for batch and semi-continuous systems (Table 7). A batch treatment system is more effective at reducing pathogens than a semi-continuous system because new pathogens are not being introduced into the system.

Table 7. Comparison of batch and semi-continuous biogas digestion on the time (days) for decline of 90% of five bacteria species.				
	Batch		Semi-continuous	
	Mean	Limits	Mean	Limits
Escherichia coli	0.8	0.6 – 1.4	1.5	1.0 – 4.0
Salmonella typhimurium	0.9	0.8 – 0.9	1.1	0.7 – 2.6
Yersinia. Enterocolitica	0.7	0.6 – 0.8	2.5	2.3 – 3.0
Listeria monocytogenes	12.3	8.3 – 25.6	35.7	14.2 – 71.4
Campylobacter jejuni	>71.0		>71.0	

Urine alkali treatment to reduce bacterial counts: Diez-Gonzalez et al. (2000) researched the concept that urine has antibacterial activity at a pH of 8.5. If the pH was adjusted down, then urine lost its ability to control bacteria. Under normal conditions, the feces-to-urine ratio is 2.2:1 and *E. coli* counts remain high at >10,000 cells/g. However, if this ratio was adjusted to 1:1 by adding additional urine, then after 10 days, the viable count reduces to < 10 cells/g. If this ratio was further adjusted to 0.4:1 or less, *E. coli* was not killed. Unfortunately, this process of adjusting feces-to-urine ratio is not practical at the producer level.

Animal urine contains large quantities of urea, which break down to ammonia and carbon dioxide by the enzyme urease. Diez-Gonzalez et al. (2000) concluded that the ammonia was not the antibacterial agent against *E. coli*, but the carbon dioxide formed combines with water to form bicarbonate, which has antibacterial activity at pH around 8.5. This leads to the possible supplementation of manure with carbonate for *E. coli* elimination, which Diez-Gonzalez et al. (2000) recommends at 4 g of sodium carbonate or 2 g of sodium hydroxide /kg of manure. The treatment costs would be less than $10 per cow per year.

Survival of viruses

Studies have shown that a variety of conditions can influence the survival of viruses and livestock infection (Ajariyakhajorn et al., 1997; Deng and Cliver, 1995; Pesaro et al., 1995). Factors affecting viral survival are temperature, pH, and the presence of bacteria that can inactivate viruses (Deng and Cliver, 1995). Deng and Cliver (1992 and 1995) found evidence that some bacteria isolated from manure could inactivate viruses. They showed that both hepatitis A and polio type 1 viruses were inactivated more rapidly in dairy and swine slurries than in contaminated septic tank effluent. Kelley et al. (1994) reported that viruses initially found in poultry litter were not found after five months of storage.

Survival of protozoa

When an oocyst (*Cryptosporidium parvum*) or cyst (*Giardia* spp.) is ingested by an animal, these structures can remain viable for long time periods. *Cryptosporidium* persists in calves for about two weeks and calves may shed *Giardia* for several months (O'Handley et al., 1999; Olson et al., 1999). *Cryptosporidium* oocysts are much more resistant to degradation than *Giardia* cysts. With freezing at – 4° C or storage at 4° C *Cryptosporidium* was infective at greater than 12 weeks. One week of freezing rendered *Giardia* infective. At 4° C the Giardia was only infective for one week. At higher temperatures *Cryptosporidium* in manure does degrade (Olson et al., 1999). At 25° C it was not infective after five weeks. The author suggested that manure application should be carried out after 12 weeks of storage and during warm weather to reduce the potential water contamination by *Cryptosporidium* from runoff.

Advanced animal wastewater treatment

Utilization of advanced waste water treatment techniques (techniques similar to municipal sewage treatment) for animal waste will not solve the pathogen problem. All data available indicate that pathogenic microorganisms, particularly viruses, pass through the sewage treatment process in large numbers (Gaudy and Gaudy, 1980).

Survivability in soil and water

Survivability of *E. coli*

Water content and temperature are important factors for *E. coli* survival in soils. A study of *E. coli* in two Kentucky soils showed longer survival with more available water (Mubiru et al., 2000). Studies indicate that heat stress reduces growth and survival of the pathogen under aerobic conditions. One study showed that non-O157 strains survived in soil for more than 60 days at 25° C and 100 days at 4° C (Bogosian et al., 1996).

E. coli can survive for long time periods in water, Survival of O157:H7 strains in river water has been shown for up to 90 days (Wang and Doyle, 1998).

Survivability of *Cryptosporidium* and *Giardia*

Cryptosporidium oocysts can survive in water or soil for more than 6 weeks at 25° C and more than 12 weeks at 25° C (Olson et al., 1999). Freezing for one week destroys *Giardia* but not *Cryptosporidium*. *Giardia* remained infective for more that 6 weeks at 4° C and for 2 to 4 at 35° C.

Survivability of pathogens on edible crops

E. coli

There is significant evidence of *E. coli* O157:H7 survival on fruits and vegetables for periods of more than 3 weeks. *E. coli* O157:H7 is extremely acid tolerant and can survive in fruit drinks even under highly acid conditions. It can survive at pH 3.7 in apple cider stored at 8° C for 31 days and at pH 2.0 in a laboratory medium for 24 hours (Miller and Kaspar, 1994; Zhao et al., 1993). Several studies have shown that *E. coli* O157:H7 can develop resistance to low pH levels (Lin et al., 1995; Lin et al., 1996). Folsom and Frank (2000) found that *E. coli* exposed to

chlorine has more resistance to heat than unexposed cells. Preadaptation to a stress encountered by *E. coli* O157:H7 such as acid can lead to enhanced resistance to a different stress such as heat (Riordan et al., 2000). The chlorinated cells required twice as much heating time.

Listeria monocytogenesy
This bacterium grows at a wide range of pH (5.5-9.0), temperature (3 - 42° C), and in high salt concentrations (up to 12%). It is well adapted to the wet environment of food processing facilities, and is difficult to control due to the range of environmental conditions in which it can survive.

Does manure refeeding pose a health risk?

Refeeding is a of method of manure utilization that reduces the quantity of manure applied to land. CAST (1978) states that before refeeding manure needs to be processed by ensiling, dehydration, composting, chemical treatment, and/or aeration to effectively destroy certain pathogens before mixing in an animal diet.

Georgia beef producers recentely raised the safety question about feeding large quantities of poultry litter. Martin et al. (1998) tested 86 litter samples throughout Georgia for *E. coli* O157:H7 and *Salmonella*. There were 64 samples from composting piles, 18 samples were not composted, and four with unknown treatments. The composting ranged from less than one month to greater than four months. While bacteria were isolated from all litter samples, no *E. coli* O157:H7 or *Salmonella* were detected in any sample. The researchers did find extremely low mold contamination in most samples. These results suggest that poultry litter is not a source of harmful pathogenic bacteria when fed to cattle.

Jeffrey et al. (1998) tested 52 dried poultry litter samples from 13 dairies in California. No *Salmonella*, *E. coli* O157, or *Campylobacter* were identified even though other strains of *E. coli* were found. Based on their study, they determined that dried poultry litter can be used as a feed and that dried poultry litter is probably not a significant source of bacteria associated with food borne disease in humans or clinical illness in cattle.

Pugh et al. (1994) surveyed 77 veterinarians in Georgia who serves cattle growers that use broiler litters. Four of the veterinarians had diagnosed Salmonellasis in cattle, usually calves and young cattle. Eight veterinarians also observed enterotoxemia. The authors stated that salmonellosis is associated with improper processing of litters for pathogen control.

How can pathogen transfer to edible crops be minimized?

Although complete eradication of pathogens on edible crops is highly unlikely, there are management practices that can help reduce pathogen transfer. Management of food-borne pathogens must start with management of animals and their wastes on the farm. However, careful management must also extend to the harvesting, transport, storage, and processing of

produce if contamination is to be avoided. A systems approach that includes all aspects of food production is required for food safety.

Reduce pathogen levels in animals
Pathogen excretion can be reduced on the farm with adequate housing and sanitation that reduces animal stress levels. Possible methods for reducing pathogen levels in animals include:

- smaller herd sizes,
- using individual birthing pens,
- delayed weaning,
- washing birthing pens between uses,
- good animal nutrition,
- feeding natural (pasture) animal feeds and fermented silage, and
- feeding poultry litter only if it is composted.

Reduce pathogen levels in manure
Pathogens in manure can be reduced if the manure is managed well on the farm. There are processes that can reduce pathogens in the manure. Drying, aerobic digestion, chemical treatment, and composting of the animal waste can substantially reduce pathogens. Mixing manure slurry with dry matter or bedding can reduce pathogens, since aerobic fermentation is more likely to occur in manure mixed with bedding than in slurry.

Storing manure can help reduce pathogen levels. In order to reduce or eliminate pathogens present, Jones (1976) recommended that manure should be stored for a month, then after manure spreading on a pasture there should be a month wait during which the pasture should not be grazed. The Commission of the European Communities stated that manure should be stored for a minimum of 60 days before spraying on land (Kelly, 1978). It is not clear what concentrations of pathogens might be acceptable before manure can be applied on pasture.

Kelley et al. (1994) found that indoor stored piles of poultry litter after four months showed significant reductions in pathogenic and indicator bacteria concentrations, and in most cases concentrations were below detection limits of approximately 30 CFU (colony forming units) / g dry weight.

Avoid water contamination
Farm and manure storage facilities should be in a location that is not susceptible to flooding or in a floodwater path, near wells, or in karst topography. Restrictions on the timing and location of manure application can reduce the risk of water contamination by pathogens. (See the Manure Storage and Handling section of this report, and the Water Quality report for more information).

What are the measures for pathogen control in edible crops?

There are no regulatory requirements that address manure handling for pathogen reduction or that provide some measures as guidelines. Feedlot rules in Minnesota (MPCA, August 3, 2000),

which established environmental regulations for feedlots, cover pathogen control only in composted manure. However, the Commission of the European Communities stated that manure should be stored for a minimum of 60 days, to reduce pathogen concentrations, before spraying on land (Kelly, 1978). Also there are regulations (Part 503 of 1993 U.S. Environmental Protection Agency (EPA) regulations) that govern pathogen reduction in sewage sludge. Sewage sludge is regulated because of its very high human pathogen content. The same processes required for pathogen reduction in sewage sludge could work for manure, so it is useful to look at these restrictions. It should be noted, however, that in general the risk for transmission of human disease organisms to food is less for manure than for sewage sludge. The EPA restrictions on sewage sludge are described below.

Measures for pathogen reduction in manure
The EPA regulations separate sewage sludge into A and B categories with respect pathogen content. Class A sludge must be treated to decrease to pathogens to essentially non-detectable. Class A sludge does not have any pathogen restrictions for land application. Class B sludge is only required to have a fecal coliform density of less that 2 million (MPN) per gram. This may require some form of treatment

Measures for pathogen reduction in manure application
Restrictions for the harvest of crops and turf on sites where class B sewage sludge is land applied are contained in Table 3-11, subpart D, part 503 of the EPA regulations. These are summarized below in Table 4.

Table 4. Restrictions for the Harvesting of Crops, Grazing of Animals, and Public Access on Sites Where Class B Sewage Sludge is Land Applied

Restrictions for the harvesting of crops:

- Food crops with harvested parts that touch the sewage sludge/soil mixture and are totally above ground shall not be harvested for 14 months after application of sewage sludge.
- Food crops with harvested parts below the land surface where sewage sludge remains on the land surface for 4 months or longer prior to incorporation into the soil shall not be harvested for 20 months after sewage sludge application.
- Food crops with harvested parts below the land surface where sewage sludge remains on the land surface for less than 4 months prior to incorporation shall not be harvested for 38 months after sewage sludge application.
- Food crops, feed crops, and fiber crops, whose edible parts do not touch the surface of the soil, shall not be harvested for 30 days after sewage sludge application.
- Turf grown on land where sewage sludge is applied shall not be harvested for 1 year after application of the sewage sludge when the harvested turf is placed on either land with a high potential for public exposure or a lawn, unless other wise specified by the permitting authority.

Restrictions for the grazing of animals:

- Animals shall not be grazed on land for 30 days after application of sewage sludge to the land.

Restrictions for public contact:

- Access to land with a high potential for public exposure, such as a park or ballfield, is restricted for 1 year after sewage sludge application. Examples of restricted access include posting with no trespassing signs, or fencing.
- Access to land with a low potential for public exposure (e.g., private farmland) is restricted for 30 days after sewage sludge application. An example of restricted access is remoteness.

[a] Adapted from part 503 U.S. EPA (1993) rules.

Summary

Pathogens that exist in manure can end up on fresh fruits and vegetables, causing illness or even death in the humans that consume them. Animal manure may contain bacteria, protozoa, and viruses that are transmissible to humans. Farm animals, birds, deer and many other animals can serve as reservoirs for pathogens, and often the exact source of food-borne disease outbreaks cannot be established.

Substantial scientific literature exists on the presence of pathogens in manure and in fresh fruits and vegetables. Almost every type of fruit and vegetable is prone to contamination by bacterial pathogens. However, only a few fruits and vegetables have been confirmed as vehicles for food-borne illness. Edible crops can carry pathogens if they have manure or manure-contaminated soil on their surfaces, or if they have been exposed to manure-contaminated water (through hydroponics or irrigation). We found no evidence of transmission of zoonotic pathogens to consumers via field crops that are processed before consumption.

Factors that contribute to pathogen transfer include the presence of pathogens in animals (influenced by age, herd size, housing, and feed), manure contamination of water, pathogen survivability during manure storage and handling and in water, and survivability on edible crops. Pathogen transfer can be reduced by promoting animal health, reducing pathogen levels in manure with storage and treatment, avoiding water contamination, using proper manure application procedures, and using proper harvest procedures. While there are no manure regulations that provide measures for pathogen control in application of manure to land used for edible crops, U.S. Environmental Protection Agency regulations for the use of sewage sludge on edible crops could be used for guidance in developing guidelines or regulations

References

Ajariyakhajorn, C., S.M. Goyal, R.A. Robinson, L.J. Johnston, and C.A. Clanton. 1997. The survival of Salmonella anatum, pseuorabies virus and porcine reproductive and respiratory syndrome virus in swine slurry. New Microbiol. 20:365-369.

Beuchat, L.R.1996. Pathogenic microorganisms associated with fresh produce. J. Food Prot. 59:204-216.

Bogosian, G., L.E. Sammons, P.J.L. Morris, J.P. O'Neill, M.A. Heitkamp, and D.B. Webber. 1996. Death of Escherichia coli K-12 strain W3110 in soil and water. Appl. Environ. Microbiol. 62:4114-4120.

Brackett, R.E. 1999. Incidence, contributing factors, and control of bacterial pathogens in produce. Postharvest. Biol. Technol. 15:305-311.

Brush, C.F., W.C. Ghiorse, L.J. Anguish, J. Parlange, and H.G. Grimes. 1999. Transport of Cryptosporidium parvum oocysts through saturated columns. J. Environ. Qual. 28:809-815.

Burnett, S.L., J. Chen, and L.R. Beuchat. 2000. Attachment of Escherichia coli O157:H7 to the surface and internal structures of apples as detected by confocal scanning laser microscopy. Appl. Environ. Microbiol. 66:4679-4687.

CAST. 1978. Feeding animal waste. Council for Agricultural Science and Technology. Report No. 75. June 22, 1978. Iowa State University. Ames, Iowa.

CAST. 1996. Integrated animal waste management. Council for Agricultural Science and Technology. Report No. 128. November, 1996. Iowa State University. Ames, Iowa.

Chapman, P.A., C.A. Siddons, A.T. Cerdan-Malo, and A.M. Harkin. 1997. A 1 year study of Escherichia coli O157 in cattle, sheep, pigs and poultry. Epidemiology and Infection. 119:245-250.

Deng, M.Y. and D.O. Cliver. 1992. Inactivation of poliovirus type 1 in mixed human and swine waste by bacteria from swine manure. Appl. Environ. Microbiol. 58:2016-//

Deng, M.Y. and D.O. Cliver. 1995. Persistance of inoculated hepatitis A virus in mixed human and animal wastes. Appl. Environ. Microbiol. 61:87-91.

Diesch, S. L. 1970. Disease transmission of water-borne organisms of animal origin. In Agricultural Practices and Water Quality. Eds: T. L. Willrich and G. E. Smith. Iowa State University Press. Ames, Iowa.

Diez-Gonzalez, F., G.N. Jarvis, D.A. Adamovich, and J.B. Russell. 2000. Use of carbonate and alkali to eliminate *Escherichia coli* from dairy cattle manure. Environmental Science and Technology 34(7):1275-1279.

Dingman, W.D. 2000. Growth of Escherichia coli O157:H7 in bruised apple (Malus domestica) tissue as influenced by cultivar, date of harvest, and source. Appl. Environ. Microbiol. 66:1077-1083.

Entringer R.A. and J. Strepelis, 1996. Health concerns resulting from the effects of animal agriculture on water resources. In Proceedings from the Animal Agriculture and the Environment; Nutrients, Pathogens, and Community Relations. North American Conference. Rochester, NY. NRAES-96. Pg 47-55.

Fisher, T.L. and D.A. Golden. 1998. Fate of Escherichia coli O157:H7 in ground apples used in cider production. J. Food Prot. 61:1372-1374.

Folsom, J.P. and J.F. Frank. 2000. Heat inactivation of Escherichia coli O157:H7 in Apple Juice exposed to chlorine. J. Food Prot. 63:1021-1025.

Galloway, J.H. 1974. Farm animal health and disease control. Lea & Febiger. Philadelphia.

Gaudy, Jr., A.F., and E.T. Gaudy. 1980. Microbiology for environmental scientists and engineers. McGraw-Hill Book Company. New York, NY. 736 pg.

Hammer, M.J., and M.J. Hammer, Jr. 2001. Water and Wastewater Technology. 4th edition. Prentice Hall. New Jersey.

Hara-Kudo, Y.H. Konuma, M. Iwaki, F. Kasuga, Y. Sugita-Konishi, Y. Ito, and S. Kumagai. 1997. Potential hazard of radish sprouts as a vehicle of Escherichia coli O157:H7. J. Food Prot. 60:1125-1127.

Hunter, C., J. Perkins, J. Tranter, and P. Hardwick. 2000. Fecal bacteria in the waters of an upland area in Derbyshire, England: The influence of agricultural land use. J. Environ. Qual. 29:1253-1261.

Inami, G.B. and S.E. Moler. 1999. Detection and isolation of Salmonella from naturally contaminated alfalfa seeds following an outbreak investigation. J. Food Prot. 6:662-664.

Jeffrey, J.S., J.H. Kirk, E.R. Atwill, and J.S. Cullor. 1998. Prevalence of selected microbial pathogens in processed poultry waste used as diary cattle feed. Poultry Science 77:808-811.

Jones, D.L. 1999. Potential health risks associated with the persistence of Escherichia coli O157:H7 in agricultural environments. Soil Use and Mang. 15:76-83.

Kearney, T.E., M.J. Larkin, and P.N. Levett. 1993. The effect of slurry storage and anaerobic digestion on survival of pathogenic bacteria. Journal of Applied Bacteriology 74:86-93.

Keene, W.E., E. Sazie, J. Kok, D.H. Rice, D.D. Hancock, V.K. Boden, T. Zhao, and M.P. Doyle. 1997. Outbreak of Escherichia coli O157:H7 infections traced to jerky made from deer meat. JAMA. 277: 1229-1231.

Kelley, T.R., O.C. Pancor, S.A. Martin, and M.A. McCann. 1998. Microbiological survey of Georgia poultry litter. Journal of Applied Poultry Research 7:90-98.

Kelley, T.R., O.C. Pancorbo, W.C. Merka, S.A. Thompson, M.L. Cabrera, and H.M. Barnhart. 1994. Fate of selected bacterial pathogens and indicators in fractionated poultry litter during storage. Journal of Applied Poultry Research 3:279-288.

Kelly, W. R. 1978. Animal and human health hazards associated with the utilization of animal effluents. EUR 6009 EN. 304 pages. Brussels: Commission of the European Communities.

Kemp, J.S., S.E. Wright, and Z. Bukhari. 1995. On farm detection of Cryptosporidium parvum in cattle, calves and environmental sample. In: W.B. Betts et al. (ed). Proc. of a conf. on Protozoan Parasities and Water. Univ. of York. Royal Soc. Of Chemistry, Cambridge, U.K.

Kudva, I.T., K. Blanch, and C.J. Hovde. 1998. Analysis of Escherichia coli O157:H7 in ovine or bovine manure and manure slurry. Appl. Environ. Microbiol. 64:3166-3174.

Kudva, I.T., P.G. Hatfield, and C.J. Hovde. 1996. Escherichia coli O157:H7 in microbial flora of sheep. J. Clin. Microbiol. 34:431-433.

Kudva, I.T., P.G. Hatfield, and C.J. Hovde. 1997. Characterization of Escherichia coli O157:H7 and other Shiga toxin-producing E. coli isolated from sheep. J. Clin. Microbiol. 35:892-899.

LeChevallier, M.W., W.D. Norton, and R.G. Lee. 1991. Occurrence of Giardia and Cryptosporidium spp. in surface water supplies. Appl. Environ. Microbiol. 57:2610-2616.

Lin, J., I.S. Lee, J. Fey, J.L. Slonczwski, and J.W. Foster. 1995. Comparative analysis of extreme acid survival in Salmonella typhimurium, Shigella flexneri, and Escherichia coli. J. Bacteriol. 177:4097-4104.

Lin, J., M.P. Smith, K.C. Chapin, H.S. Baik, G.N. Bennett, and J.W. Foster. 1996. Mechanisms of acid resistance in enterohemorrhagic Escherichia coli. Appl. Environ. Microbiol. 62:3094-3100.

Marshall, M.M., D. Naumovitz, Y. Ortega, and C.R. Sterling. 1997. Waterborne protozoan pathogens. Clin. Microbiol. Rev. 10:67-85. (not mentioned in the text)

Martin, S. A., and M. A. McCann. 1998. Microbiological survey of Georgia poultry litter. Journal of Applied Poultry Research 7:90-98.

Mawdsley, J.L., A.E. Brooks, and R.J. Merry. 1996a. Movement of the protozoan pathogen Cryptosporidium parvum through three contrasting soil types. Biol. Fert. Soils 21:30-36.

Mawdsley, J.L., A.E. Brooks, R.J. Merry and B.F. Pain. 1996b. Use of a novel soil tilting table apparatus to demonstrate horizontal and vertical movement of the protozoan pathogen Cryptosporidium parvum. Biol. Fert. Soils 23:215-220.

Mawdsley, J.L., R.D. Bardgett, R.J. Merry, B.F. Pain, and M.K. Theodoru. 1995. Pathogen in livestock waste, their potential for movement through soil and environmental pollution. Appl. Soil Ecol. 2:1-15.

Metcalf and Eddy, Inc. 1979. Wastewater Engineering: Treatment, Disposal, Reuse. 2nd edition. McGraw-Hill Book Company. New York.

MidWest Plan Service. 2000. Manure storages. MWPS-18, Section 2 (Review Draft), Ames, IA.

Miller, L.G. and C.W. Kaspar. 1994. Escherichia coli O157:H7 acid tolerance and survival in apple cider. J. Food Prot. 57:760-464.

Miner, J.R., F.J. Humenik, and M.R. Overcash. 2000. Managing livestock wastes to preserve environmental quality. Iowa State University Press. Ames, Iowa.

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

Munch, B., H.E. Larsen and B. Aalbaek. 1987. Experimental studies on the survival of pathogenic and indicator bacteria in aerated and non-aerated cattle and pig slurry. Biological Wastes 22:49-65.

NRAES. 1999. Earthen manure storage design considerations. NRAES-109. Ithaca, NY.

NRCS. 1992. Agricultural waste management field handbook. Part 651. Natural Resource and Conservation Service (old Soil Conservation Service, SCS). Washington, D.C.

O'Donoghue, P.J. 1995. Cryptosporidium and Cryptosporidiosis in man and animals. Int. J. Parasitol. 25:139-142.

O'Handley, R.M., C. Cockwill, T.A. McAllister, M. Jelinski, D.W. Morck, and M.E. Olson. 1999. Duration of naturally acquired giardiosis and cryptosporidiosis in dairy calves and association with diarrhea. J. Am. Vet. Med. Assoc. 214:391-396.

Olson, M.E., J. Goh, M. Phillips, N. Guselle, and T. McAllister. 1999. Giardia cyst and Cryptosporidium oocyst survival in water, soil, and cattle feces. J. Environ. Qual. 28:1991-1996.

Pell, A.N. 1997. Manure and microbes: public and animal health problem. J. Dairy Sci. 80:2673-2681.

Pesaro, F., I. Sorg, and A. Metzler. 1995. In situ inactivation of animal viruses and a coliphage in nonaerated liquid and semiliquid animal wastes. Appl. Environ. Microbiol. 61:92-97.

Peterson, E.W., R.K. Davis, and H.A. Orndroff. 2000. 17â-estradiol as an indicator of animal wast contamination in mantled karst aquifers. J. Environ. Qual. 29:826-834.

Pugh, D.G., J.G.W. Wenzel, and G.D. Andrea. 1994. A survey on the incidence of disease in cattle fed broiler litter. Vet. Med. 89:665-667.

Pyle, B.H., S.C. Broadaway, and G.A. McFeter. 1999. Sensitive detection of Escherichia coli O157:H7 in food and water by immunomagnetic separation and solid-phase laser cytometry. Appl. Environ. Microbiol. 65:1966-1972. (not mentiond in text)

Riordan, D.C.R., G. Duffy, J.J. Sheridan, R.C. Whiting, I.S. Blair, and D.A. McDowell. 2000. Effects of acid adaptation, product ph, heating on survival of Escherichia coli O157:H7 in pepperoni. Appl. Environ. Microbiol. 66:1726-1729.

Rose, J.B., C.P. Gerba, and W. Jakubowski. 1991. Survey of potable water supplies for Cryptosporidium and Giardia. Environ. Sci. Technol. 25:1393-1400.

Smith, J. L. 1994. Taenia solim neurocysticercosis. Journal of Food Protection 57(9): 831-844

Strauch, D., and G. Ballarini. 1994. Hygienic aspects of production and agricultural use of animal wastes. Journal of Veterinary Medicine Series B 41:176.

Stehman, S.M., C. Rossiter, P. McDonough, and S. Wade. 1996. Potential pathogens in manure. In Proceedings from the Animal Agriculture and the Environment; Nutrients, Pathogens, and Community Relations. North American Conference. Rochester, NY. NRAES-96. Pg 47-55.

Varnam, A.H. and M.G. Evans. 1999. Foodborne pathogens. Wolfe Publication, Ltd. London.

Wallace, J.C., T. Cheasty, and K. Jones. 1997. Isolation of Verocytotoxin-producing Escherichia coli O157 from wild birds. J. Appl. Microbiol. 82:399-404.

Wang, G. and M.P. Doyle. 1998. Survival of enterohemorrhagic Escherichia coli O157:H7 in water. J. Food Prot. 61:662-667.

Zhao, T., M.P. Doyle, and R.E. Besser. 1993. Fate of enterohemorrhagic Escherichia coli O157:H7 in apple cider with and without preservatives. Appl. Environ. Microbiol. 59:2526-2530.

Zhao, T., M.P. Doyle, J. Shere, and L. Garber. 1995. Prevalence of enterohemorrhagic, Escherichia coli O157:H7 in a survey of dairy herds. Appl. Environ. Microbiol. 61:1290-1293.

Northern New York Agricultural Development Program

The Effect of the Timing of Animal Manure Application on Nutrient Fate Under Maize and Grass

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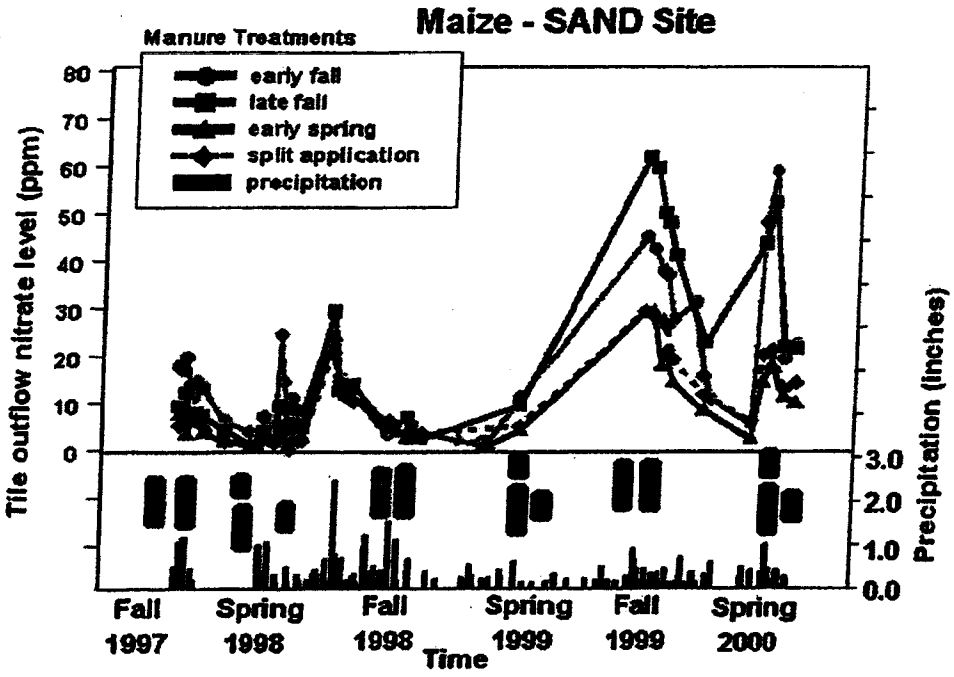
Results from the leaching experiments show that the timing of manure application significantly affects nitrate and phosphorus leaching. On the loamy sand, flow-weighted mean nitrate levels in tile outflow (Appendix 1) were highest for early-fall application on corn (23.4 mg L^{-1}) and second-highest for late fall application (19.3 mg L^{-1}). Spring application (either prior to planting or split) resulted in significantly lower nitrate leaching losses. Nitrate leaching losses from manure application on grass were minimal and apparently do not pose serious concerns. This is explained by the higher N uptake potential and a longer growing season for the grass. Also, with non-incorporated manure application on grass, much N is lost through ammonia volatilization. A very similar pattern was observed for the clay loam (Appendix 2), although the nitrate concentrations were consistently lower. Previous research on these plots attributed this to higher denitrification losses. The figures also indicate a strong effect of weather conditions. Both very wet (1998) and very dry (1999) growing seasons resulted in excessive N leaching losses. Wet growing seasons result in immediate losses of nutrients, while dry seasons result in high levels of residual nitrate in the soil, which is leached in the following fall and winter.

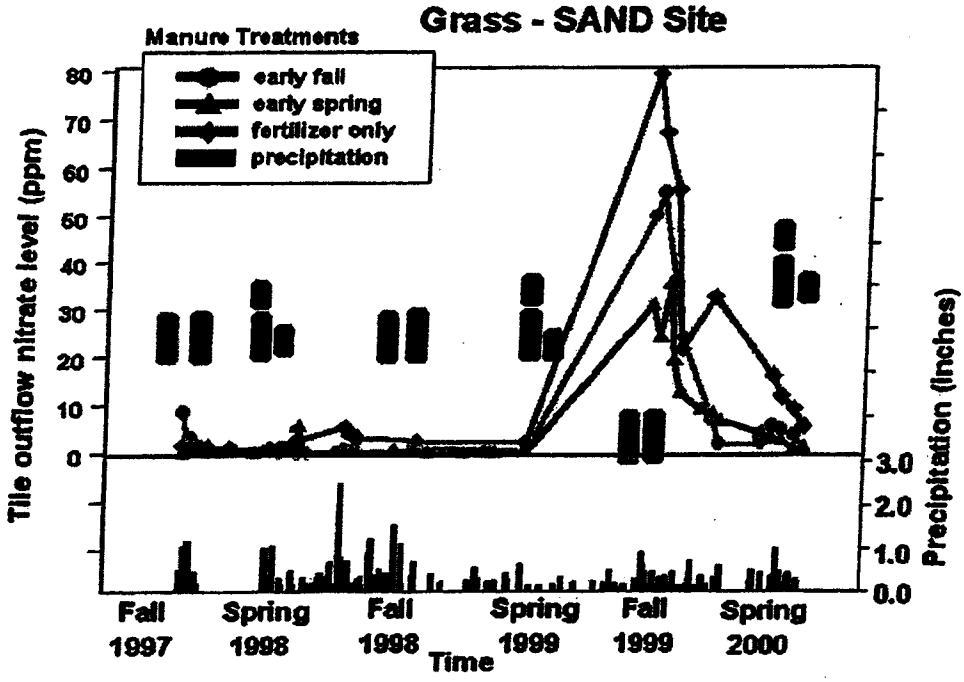
Unlike the nitrate losses, P leaching losses were very low under the loamy sand, both for corn and grass. Conversely, P leaching losses from the clay loam soil are high, especially with non-incorporate manure on grass. This is explained by the dominance of preferential flow in these soils, which allows the nutrients to rapidly move to the tile lines. Levels are 10 to 70 times the level of concern in surface water bodies.

These preliminary results corroborate the hypothesis that the mechanisms and quantities of leaching losses vary greatly among nutrients, soil types, crop type and timing of manure application. Environmental losses of nutrients can be significantly reduced by optimizing the timing of manure application. These data also proivide input for N and P Index models.

Appendix 1. Nitrate leaching losses under maize and grass for the loamy sand site. Shown are

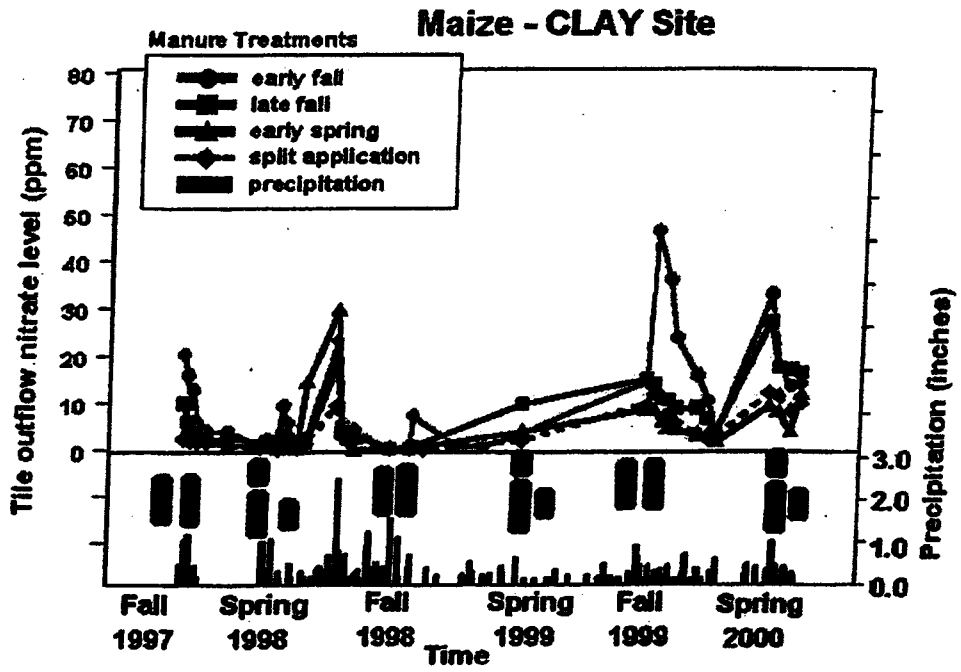
measured concentrations (figures) and flow-weighted averages (table). Units are mg L^{-1}.

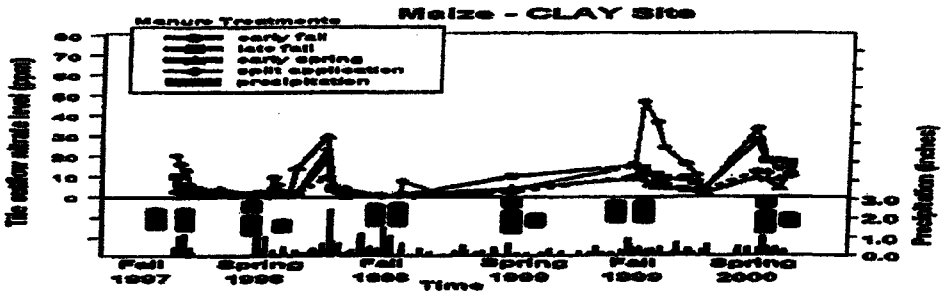


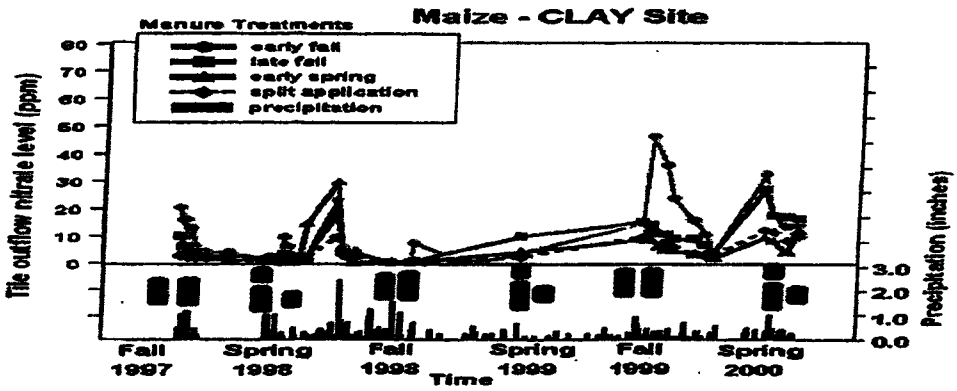


	Fall / Winter 1997	Growing Season 1998	Fall / Winter 1998	Fall / Winter 1999	Spring 2000
Maize					
Early Fall	15.29	14.15	8.01	36.41	33.28
Late Fall	8.11	10.35	6.63	51.85	26.00
Early Spring	7.13	9.27	3.83	28.16	13.31
Split Applic	5.35	9.02	4.67	25.41	15.05
Grass					
Early Fall	3.54	0.70	0.62	34.06	2.01
Early Spring	0.46	0.57	0.33	20.54	1.54
Fertilizer	1.33	2.92	1.82	57.66	5.48
Precip (mm)	**216**	**518**	**183**	**140**	**127**

Appendix 2. Nitrate leaching losses under maize and grass for the clay loam site. Shown are measured concentrations (figures) and flow-weighted averages (table). Units are mg L^{-1}.

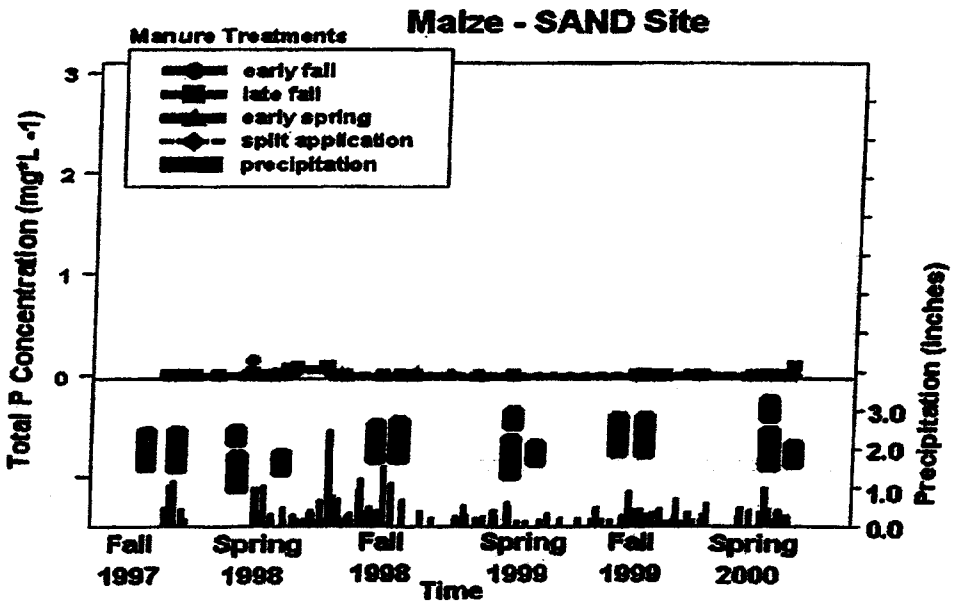


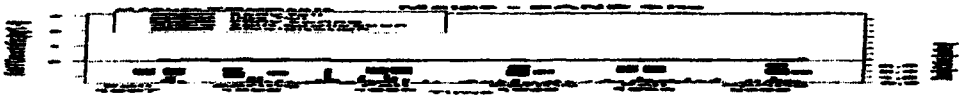


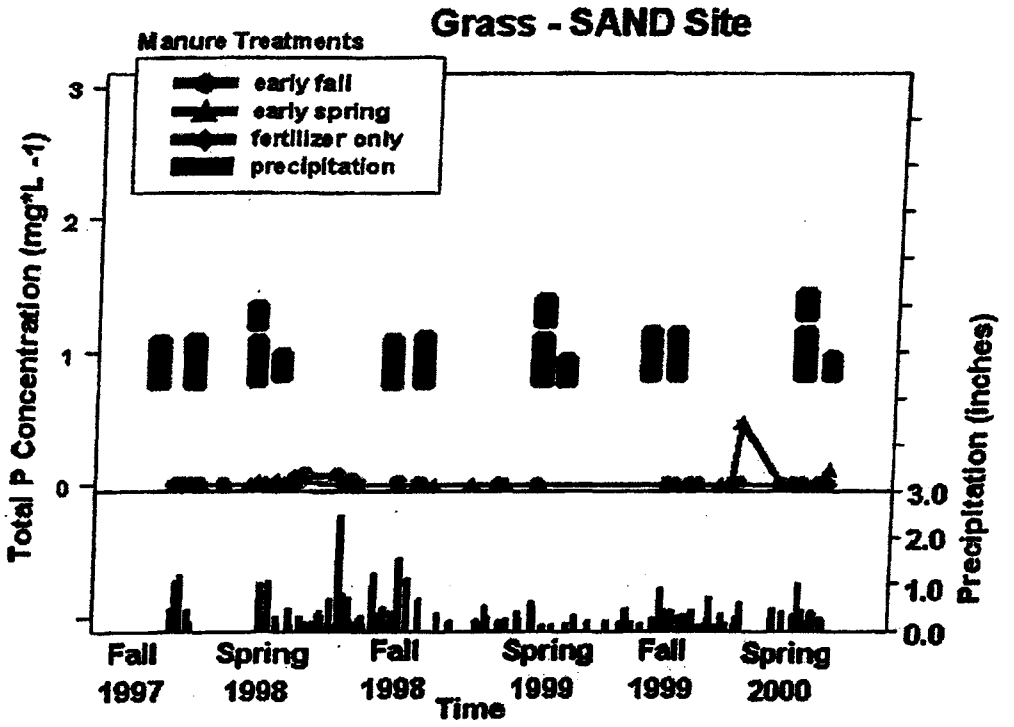


Manure Applic Time	Fall / Winter 1997	Growing Season 1998	Fall / Winter 1998	Fall / Winter 1999	Spring 2000
MAIZE					
Early Fall	14.26	7.86	6.10	42.43	16.46
Late Fall	5.45	5.42	2.96	12.85	17.47
Early Spring	2.51	7.26	1.10	8.01	6.84
Split Applic	2.02	2.99	1.37	9.38	10.38
GRASS					
Early Fall	0.92	0.38	0.49	1.02	1.56
Early Spring	0.39	0.51	0.17	2.47	2.90
Fertilizer	1.44	0.37	0.74	3.07	3.03
Precip (mm)	**216.0**	**510.0**	**183.0**	**140.0**	**127.0**

Appendix 3. Total P leaching losses under maize and grass for the loamy sand site. Shown are measured concentrations (figures) and flow-weighted averages (table). Units are mg L^{-1}.

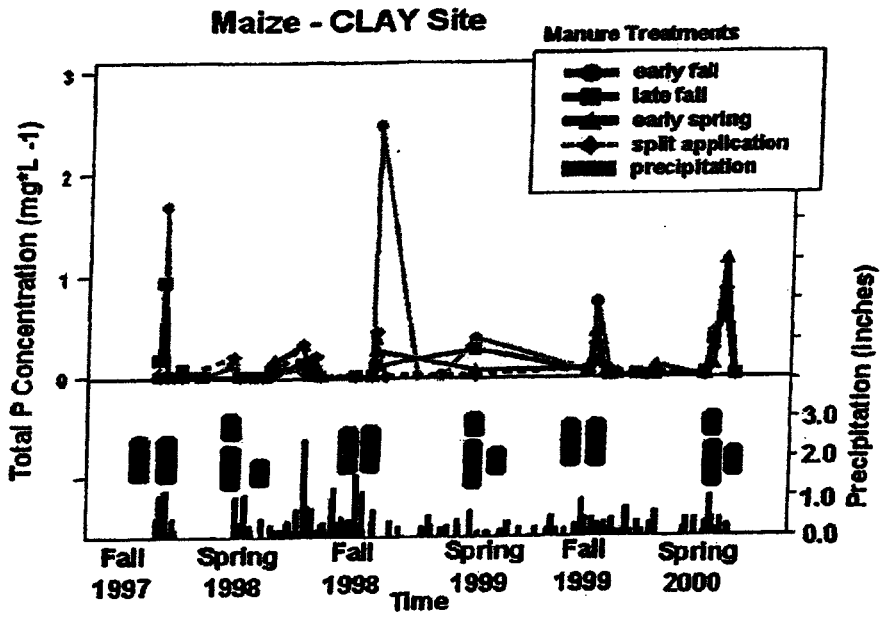


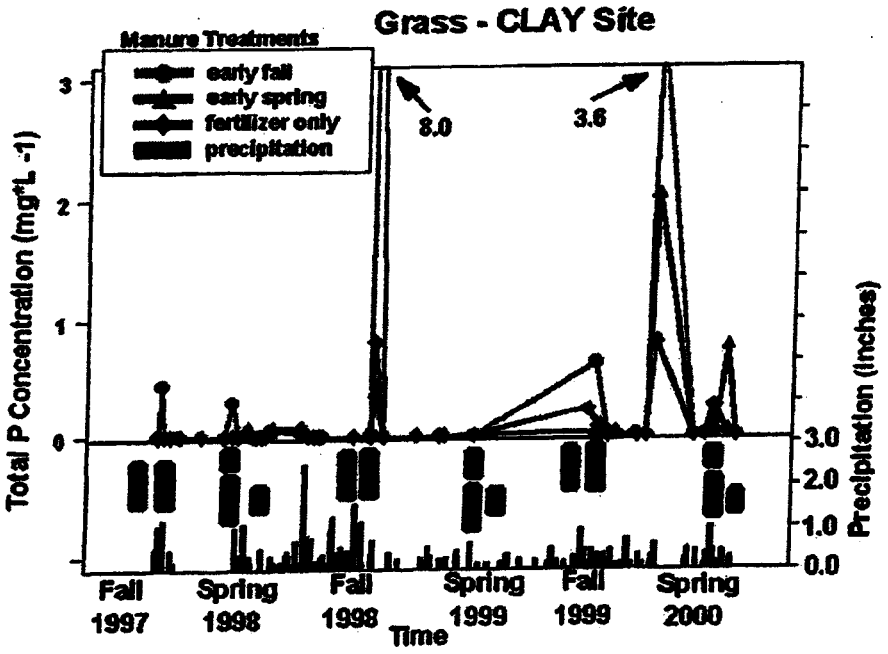




Manure Applic Time	Fall / Winter 1997	Growing Season 1998	Fall / Winter 1998	Fall / Winter 1999	Spring 2000
MAIZE					
Early Fall	0.00	0.02	0.00	0.00	0.00
Late Fall	0.00	0.01	0.00	0.00	0.09
Early Spring	0.00	0.02	0.00	0.00	0.01
Split Applic	0.00	0.01	0.00	0.00	0.00
GRASS					
Early Fall	0.00	0.02	0.00	0.00	0.00
Early Spring	0.00	0.01	0.00	0.01	0.05
Fertilizer	0.00	0.00	0.00	0.00	0.00
Precip (mm)	**216.0**	**510.0**	**183.0**	**140.0**	**127.0**

Appendix 4. Total P leaching losses under maize and grass for the clay loam site. Shown are measured concentrations (figures) and flow-weighted averages (table). Units are mg L^{-1}.





Manure Applic Time	Fall / Winter 1997	Growing Season 1998	Fall / Winter 1998	Fall / Winter 1999	Spring 2000
MAIZE					
Early Fall	0.75	0.03	1.86	0.62	0.49
Late Fall	0.40	0.03	0.12	0.09	0.39
Early Spring	0.00	0.09	0.19	0.31	0.42
Split Applic	0.00	0.11	0.39	0.19	0.40
GRASS					
Early Fall	0.15	0.01	5.87	2.88	0.02
Early Spring	0.00	0.01	0.26	0.92	0.18
Fertilizer	0.00	0.01	0.61	0.81	0.19

PATHOGENS IN ANIMAL WASTES AND THE IMPACTS OF WASTE MANAGEMENT PRACTICES ON THEIR SURVIVAL, TRANSPORT AND FATE

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Introduction

Manure and other wastes (such as respiratory secretions, urine and sloughed feathers, fur or skin) of various agricultural (livestock) animals often contain high concentrations (millions to billions per gram of wet weight feces) of human pathogens (disease-causing microorganisms). Per capita fecal production by agricultural animals such as cattle and swine far exceeds that of humans, and the trend for production facilities to harbor thousands to tens of thousands of animals in relatively small spaces results in the generation of very large quantities of concentrated fecal wastes that must be effectively managed to minimize environmental and public health risks.

Pathogens

As shown in Table 1, animal pathogens posing potential risks to human health include a variety of viruses (such as swine hepatitis E virus), bacteria (such *Salmonella* species), and parasites (such as *Cryptosporidium parvum*), some of which are endemic in commercial livestock and difficult to eradicate from both the animals and their production facilities. Hence, pathogens in animal manure and other wastes pose potential risks to human and animal health both on and off animal agriculture production facilities if the wastes are not adequately treated and contained. There are also growing public health concerns about the high concentrations of antibiotic-resistant bacteria in agricultural animals resulting from the therapeutic and growth-promotion use of antibiotics in animal production. This report reviews: (1) the types of pathogens potentially present in the manure of swine and other agricultural animals, (2) the levels of some important microbial pathogens and indicators for them that have been detected in animal wastes, (3) the potential for off-farm release or movement of pathogens present in manure and other wastes under current or proposed management practices, and (4) the extent to which these pathogens are reduced by currently used and candidate manure treatment and management technologies.

Table 1. Some Human Pathogens Potentially Present in Animal Wastes

Viruses/Groups:	Hepatitis E virus (swine), Reoviruses, Rotaviruses, Adenoviruses*, Caliciviruses*, Influenza viruses (Orthomyxoviruses)*
Bacterium/Group:	*Salmonella* spp., *Campylobacter* spp., *Escherichia coli* **, *Aeromonas hydrophila***, *Yersinia enterocolitica*, *Vibrio* spp., *Leptospira* spp., *Listeria* spp.
Parasites (Protozoans):	*Cryptosporidium parvum*, *Giardia lamblia*, and *Balantidium coli*

*Humans and animals (including swine) usually have distinct strains of these viruses, but not always.
**Some strains of these bacteria are non-pathogenic and others are pathogenic. The extent to which pathogenic strains occur in animal wastes varies with the animal species and other factors.

Some of the important pathogens potentially present in animal manures are not endemic in the United States, but there are growing concerns that such non-endemic pathogens may be introduced either accidentally or deliberately. Newly recognized or emerging livestock animal pathogens with uncertain host ranges continue to be discovered, and there are concerns that these pathogens, such as hepatitis E virus and orthomyxoviruses (influenza viruses), may be able to infect humans.

Pathways for Pathogen Movement on and off Farms

Pathogens from animal manures and other wastes have the potential to contaminate water, land and air if containment and treatment do not adequately manage the wastes. Pathogens are capable of persisting for days to weeks to months, depending on the pathogen, the medium and the environmental conditions. Many treatment and management systems for animal manure are based on the principle of no discharge and the recycling of manure constituents on the farm. However, off-farm movement or transport of animal waste pathogens has occurred via water, air and other media and is an infectious disease concern within the animal industry. Pathogen contamination of farm workers is also possible, and infection of farm workers can lead to further transmission of pathogens to family members and other contacts.

Pathogen Reductions by Manure Treatment and Management Processes

Estimated pathogen reductions in animal manures are summarized in Table 2. The reductions of some pathogens by some animal waste treatment processes have been determined in laboratory and pilot scale field studies. In general, thermophilic processes, such as pasteurization, thermophilic digestion and composting, are capable of producing extensive (>4 $\log_{10}$) pathogen inactivation, and therefore, resulting treated residuals are likely to contain only low pathogen concentrations. Further studies are recommended to better characterize pathogen inactivation in thermophilic processes for manure treatment and to define the optimum conditions to achieve extensive pathogen reductions.

Drying of some animal manures is a widely practiced management approach in some places. However, little is known about the extent to which pathogens are inactivated in manure drying processes or during dry storage because there have been few if any studies to document their effectiveness. Desiccation or drying to very low moisture levels (<1%) has been shown to result in extensive (>4 $\log_{10}$) inactivation of pathogens in municipal biosolids and in soils. Therefore, studies are recommended to determine the rate and extent of pathogen inactivation in drying and desiccation processes for animal manures.

Most mesophilic biological treatment processes for animal manures are not likely to reduce pathogen levels by more than 1-2 $\log_{10}$ or 90-99% unless several treatment reactors or processes are used in series. Therefore, treated manures, effluents or biosolids from such processes may still contain high concentrations of pathogens. The fate of these pathogens in subsequent management operations, such as land application or prolonged storage, is uncertain and has not been adequately determined. Therefore, further studies on effectiveness of mesophilic treatment processes in reducing pathogens and on the fate of pathogens in these post-treatment management processes are recommended.

Chemical treatments of animal manures are typically by lime or other alkaline treatment. Such treatment is widely practiced for municipal biosolids but less so for animal wastes. Alkaline stabilization for pathogen inactivation has been highly effective in municipal biosolids, and promising results have been obtained when it has been applied to animal biosolids. Therefore, further studies are recommended to better characterize pathogen inactivation by alkaline treatments of animal biosolids with respect to solids composition, pH and storage and handling conditions.

Summary, Conclusions and Recommendations

Pathogen reduction by animal waste treatment processes and management systems has been studied only for a few microbes, primarily indicator bacteria such as fecal coliforms. Therefore, removal and inactivation of the many different

TABLE 2. Summary of Animal Waste Treatment Processes and Estimated Pathogen Reductions

Treatment Process	Est. Pathogen Reduction (log_{10})	Comments
Physical		
Heat/Thermal Processes		
Mesophilic	Typically, 1-2	Depends on temperature, pathogen, contact time, pH, etc.
Thermophilic	Typically, >4	Depends on temperature, pathogen, contact time, pH, etc.
Freezing	Variable	Depends on pathogen, waste composition and conditions, temperature, etc.
Drying or desiccation	Typically >4 at <1% moisture; Typically <1 at >5% moisture	Depends on pathogen, contact time, pH, etc.
Gamma Irradiation	Typically >3	Varies with pathogen, dose, waste, etc.
Chemical		
High pH (>11)	Inactivation at high pH, e.g., alkaline/lime stabilization; >3-4	Varies with pathogen, contact time, pH, etc.
Low pH (<2 to <5)	Inactivation at low pH; acidification: typically, <2	Depends on pathogen, contact time, pH, etc.
Ammonia	Inactivation at higher pH where NH_3 predominates	Varies with pathogen, contact time, pH, other waste constituents
Biological Processes		
Aerobic, mesophilic	Typically 1-2	Varies with pathogen, solids separation, contact time, reactor design, temp.
Aerobic, thermophilic (composting)	Typically >4	Depends on pathogen, solids separation, contact time, reactor design, mixing methods, temperature
Anaerobic, mesophilic	Typically 1-2	Depends on pathogen, contact time, reactor design, solids separation, temperature
Anaerobic, thermophilic	Typically >4	Depends on pathogen, contact time, reactor design, solids separation, temperature
Silage treatment, mesophilic	Variable	Depends on ensiling conditions and pathogen
Land application	Highly variable and largely unknown; potentially high	Depends on site-specific factors: temperature, precipitation, vadose zone, loading, sunlight, riparian buffers, etc.

kinds of pathogens in various waste treatment processes and management systems is uncertain and needs further investigation. Although land application systems also influence pathogen survival and movement, this has not been extensively studied either. Stored manure also can attract vectors, and these vectors can either introduce or spread pathogens. Therefore, there are considerable uncertainties about the extent to which various pathogens survive waste treatment processes, are released into the environment and are available to be transported off of farms. Off-farm contamination can potentially occur inadvertently, such as in unplanned and uncontrolled releases by runoff, aerosolization or infiltration into soils and groundwater, or it can occur pur-

posefully when biosolids and other manure residuals are transported off of farms to be land applied, marketed or for other beneficial uses.

The ultimate fate of manure pathogens remains especially uncertain for large-scale, multi-stage systems employing treatment or storage followed by land application at production facilities with large numbers of animals and minimum acreage (confined or concentrated animal feeding operations). Because of the magnitude of the quantities of animal wastes generated by these facilities and the potentially high pathogen loadings that can result if the treated manure residuals still contain high pathogen concentrations, further investigation of the fate of pathogens in these systems and their surrounding environments is recommended.

Definitive or reference methods to recover and detect many of the pathogens in animal manures and their treated residual solids and liquids are lacking, especially for hyper-endemic or emerging pathogens, such as hepatitis E virus, bacteria such as *E. coli* O157:H7, *Salmonella typhimurium* and *Yersinia enterocolitica*, and parasites such as *Giardia lamblia* and *Cryptosporidium parvum*. Therefore, the extent to which these pathogens are removed, inactivated or persist in animal waste treatment processes and management systems remains uncertain due to the limitations of the recovery and detection methods. The development, evaluation and application of reliable, sensitive and affordable methods to recover and detect pathogens in animal manures and their treated residual solids and liquids are recommended.

Methods are available to recover and detect some fecal indicator microbes in animal manures and their treated residual solids and liquids. However, the methods for some indicators, such as bacterial viruses (coliphages) and spores of *Clostridium perfringens*, have not been adequately verified and collaboratively tested in these types of samples. Such verification and performance characterization studies are recommended. Also recommended are comparative studies on the removal, inactivation and fate of indicator microbes and animal pathogens in manure treatment processes and management systems. If such studies show that indicator microbes reliably reflect or predict the responses and fates of animal pathogens in manure treatment processes and management systems, then the indicators can be used in practical, rapid and affordable monitoring and surveillance activities to assess treatment process and system performance and the pathogen quality of treated residuals.

The full text of the White Papers is available for $25 from Midwest Plan Service,

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