Components of Dairy Manure Management Systems[1]

H. H. VAN HORN,[2] A. C. WILKIE,[3] W. J. POWERS,[2] and R. A. NORDSTEDT[4]
Departments of Dairy and Poultry Sciences,
Soil and Water Science,
and Agricultural Engineering
University of Florida
Gainesville 32611

ABSTRACT

Dairy manure management systems should account for the fate of excreted nutrients that may be of environmental concern. Currently, regulatory oversight is directed primarily at the assurance of water quality; N is the most monitored element. Land application of manure at acceptable fertilizer levels to crops produced on the farm by hauling or by pumping flushed manure effluent through irrigation systems is the basis of most systems. Nutrient losses to surface and groundwaters can be avoided, and significant economic value can be obtained from manure as fertilizer if adequate crop production is possible. Dairies with insufficient crop production potential need affordable systems to concentrate manure nutrients, thereby reducing hauling costs and possibly producing a salable product. Precipitation of additional nutrients from flushed manures with sedimented solids may be possible. Composting of separated manure solids offers a possible method to stabilize solids for distribution, but, most often, solids separated from dairy manures are fibrous and low in fertility. Manure solids combined with wastes from other sources may have potential if a marketable product can be produced or if sufficient subsidy is received for processing supplementary wastes. Solutions to odor problems are needed. Energy generated from manure organic matter, via anaerobic digestion, reduces atmospheric emissions of methane and odorous compounds. Use of constructed wetlands or harvesting of photosynthetic biomass from wastewater has the potential to improve water quality, making extensive recycling possible.

(**Key words**: manure, nutrient management, odors, environmental concerns)

Abbreviation key: GE = gross energy, TS = total solids, VS = volatile solids.

INTRODUCTION

Manure nutrients and decaying organic matter are natural components of the environment that ultimately contribute to the production of more plant and animal tissue. Thus, although they may be called wastes, these components are in fact resources to be recycled in the natural ecosystem. When these resources are in short supply, they are valued and reused as true resources. However, when they are in excess and result in detrimental environmental effects, they are truly wastes. In such circumstances, society chooses to pay for their management even if costs exceed the direct value of the resources recovered.

Currently, there are major concerns about the negative effects of nutrient losses from the manure of large dairy herds on ground and water quality. Most regions of the world with intensive, domestic livestock production (i.e., large numbers of food-producing animals maintained on small acreage) have begun monitoring farms to ensure that leakage of nutrients to the environment is avoided (52). Emissions of odorous compounds are regulated in all US states through nuisance legislation and, in several states, through odor measurements taken at the property line (56). Addi-

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[2]Department of Dairy and Poultry Sciences.
[3]Department of Soil and Water Science.
[4]Department of Agricultural Engineering.

tional regulations sometimes include standards for volatile emissions of ammonia (e.g., in The Netherlands), and studies of methane emissions (11) may lead to regulatory oversight in the future.

Fertilizer nutrients in manure are potentially recyclable through plants, thus avoiding excess nutrient losses to water and the atmosphere, if land applications are in balance with plant uptake. Salt nutrients (e.g., Na and Cl) in manure, however, are a potential limitation to use of manure as fertilizer in regions with inherently high salt content in soils and irrigation waters. Organic constituents of manures have potential for conversion by anaerobic digestion to biogas, which can be used for fuel or production of electricity, as well as reduced odor emissions and release of methane to the atmosphere.

This paper reviews design components of dairy manure management systems, including manure production and potential methods of processing that optimize resource recovery via environmentally accountable approaches. The design of an optimal manure management system should address the following major factors:

1. Manure production and characterization.
2. Environmental components.
3. Methods for processing and resource recovery.
4. Optimization of system options.

Thus, the optimal manure management system should be designed to minimize detrimental environmental impacts and to maximize resource recovery and reuse to the greatest extent possible.

PRODUCTION AND CHARACTERIZATION OF DAIRY MANURE

Predicting Amount and Composition

Considerable variation in the amount of excretion occurs, depending on DMI, nutrient concentration, and digestibility of diet. Experiments measuring fecal and urine P concentrations (39) and N excretion (60) with variable dietary concentrations confirm that total excretions of these elements are well predicted by subtracting P or N content in milk from the amounts consumed. Thus, excretion estimates for N, P, and a number of other mineral elements were developed based on dietary intake minus milk content (Table 1), assuming body stores of these elements to be constant, a reasonable assumption for cows for year-round herd average conditions. Table 1 also includes daily excretion estimates from the American Society of Agricultural Engineers (1) that are widely used by engineers designing waste management systems. Although near average for manure excreted by 635-kg dairy cows, these estimates do not reflect accurately the wide range of dietary intakes that occur across farms.

Data from experiments measuring fecal and urinary P excretion [93 individual P balances from lactating cows (39) and 15 from dry cows (68)] were used to develop estimates of P excretion (the Morse equation) based on P intake and milk yield (61). Estimates (Y) of daily P excretions (grams per day) obtained using the Morse equation ($Y = 9.6 + .472X + .00126X^2 + .323$ kg of milk/d, where X = grams per day of P intake) are included in Table 1 for cows that are dry or producing 45.4, 31.8, or 22.7 kg of milk/d and consuming diets of .40, .45, or .60% P (DM basis). These excretion estimates closely agree with estimates derived from P intake minus P excretion in milk (Table 1). Yearly excretion estimates were similar for the two methods of calculation based on DMI of 25.3, 21.0, 17.8, and 11.4 kg/d [from NRC (40)] and milk yields for 40 d at 45.4 kg milk/d, 130 d at 31.8 kg/d, 135 d at 22.7 kg/d, and 60 d dry; i.e., 18 kg of P/yr if diet DM was .40% P, 21 or 22 kg/yr with .45% P, and 31 or 32 kg/yr with .60% P. Excretion estimates using the Morse equation were greater than those using P intake minus milk P when milk yields were high and cows were in negative P balance but less when in late lactation and the dry period when cows were replenishing body reserves.

Similarly, Tomlinson (60) found that total N excretion estimates obtained using equations predicting daily urinary N and fecal N, which were developed from 59 complete daily collections of urine and feces from cows with variable N intake, were nearly identical to those obtained from NRC equations (40) and to those using N intake minus N in milk (Figure 1).

Estimates of excretion are important for total farm nutrient budgeting and point out the

TABLE 1. Daily and yearly excretion estimates of various fractions and nutrients by 635-kg Holstein cows.[1]

	Daily excretion per cow					Annual total
Fraction or nutrient	From (1)	40-d LC[2] 45.4 kg of milk 25.3 kg of DMI	130-d LC 31.8 kg of milk 21.0 kg of DMI	135-d LC 22.7 kg of milk 17.8 kg of DMI	60-d LC Dry 11.4 kg of DMI	365 d 9004 kg of milk 6827 kg of DMI
	(kg/d)					(kg/yr)
Raw manure (feces + urine)	54.6	88.4	72.6	56.7	36.3	22,805
Feces, wet		56.7	45.4	34.0	20.4	13,982
Urine	16.5	31.8	27.2	22.7	15.9	8822
Total solids (.33 DMI + urine DM)	7.6	9.8	8.2	6.9	4.5	2650
Water in manure	47.0	78.7	64.4	49.8	31.8	20,155
Volatile solids	6.4	8.1	6.8	5.7	3.7	2208
BOD,[3] 5-d	1.0	1.3	1.1	.9	.6	353
COD	7.0	8.9	7.5	6.3	4.1	2429
Total N (NRC, low)	.286	.408	.330	.273	.165	104
Total N (NRC, high)	.286	.467	.384	.317	.199	123
Urea + ammonia N (NRC, low)		.185	.140	.113	.057	44
Urea + ammonia N (NRC, high)		.227	.177	.144	.081	56
Ammonia N	.050					
P (diet .40% P)	.060	.056	.052	.048	.046	18
P (Morse model,[4] diet .40% P)	.060	.066	.053	.045	.032	18
P (diet .45% P)	.060	.068	.063	.057	.051	22
P (Morse model,[4] diet .45% P)	.060	.079	.064	.053	.036	21
P (diet .60% P)	.060	.106	.094	.084	.069	32
P (Morse model,[4] diet .60% P)	.060	.125	.098	.079	.050	31
Ortho P	.039					
K (diet .8% K)	.184	.134	.120	.108	.091	41
K (diet 1.2% K)	.184	.235	.204	.179	.137	68
Ca (diet .65% Ca)	.102	.110	.098	.088	.074	34
Ca (diet .90% Ca)	.102	.173	.151	.133	.103	51
Mg (diet .20% Mg)	.045	.046	.039	.033	.023	13
Mg (diet .35% Mg)	.045	.084	.070	.060	.040	23
Na (diet .35% Na)	.033	.066	.058	.051	.040	19
Cl (diet .55% Cl)	.083	.089	.081	.073	.063	28
S	.032					

[1]Adapted from data of Van Horn et al. (63). Crude protein percentage of total dietary DM for cows producing 45.4, 31.8, and 22.7 kg milk/d and dry cows for "NRC, low" diets were 16.0, 14.8, 13.8, and 11.0%; respective CP percentage for "NRC, high" diets were 17.5, 16.4, 15.3, and 12.0% of total dietary DM. Unless otherwise specified, daily excretion estimates are based on dietary intake (DMI × nutrient percentage of DM) minus amount secreted into milk. Milk quantities were based on quantity × nutrient composition of milk, which was estimated as .512% N, .10% P, .12% Ca, .15% K, .01% Mg, .05% Na, and .11% Cl.

[2]LC = Length of lactation cycle segment.

[3]BOD = Biological oxygen demand; COD = chemical oxygen demand.

[4]Morse model estimated P excretion (grams per day) = $9.6 + .472X + .00126X^2 + (.323 \times \text{kg milk/d})$, where X = P intake (grams per day). Original data from Morse et al. (39) and Wang and Beede (68) were fitted to equation by Van Horn (61).

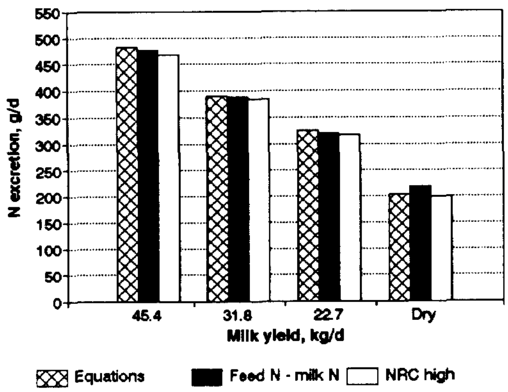

Figure 1. Estimated excretion of N by equations from Tomlinson (60), N intake in feed g/d minus N excretion in milk (at .512%), and NRC (40) CP standard (NRC, high) when N intake and DMI are same as that generated by NRC recommendations (40) for cows with milk yields as indicated.

need to avoid feeding cows diets that exceed nutrient requirements to minimize excretions because there usually is an environmental cost to managing nutrient excretions. As a corollary of a dairy's nutrition program, it is quite possible to predict available manure fertilizer nutrients from feed nutrients delivered to the cows adjusted for nutrients in daily milk yield. Inclusion of an adjustment for BW gain and production of a calf would be a refinement that could be helpful, but minor. Estimated nutrient excretions, however, do not replace the need for chemical analyses of manure relative to fertilizer value at application because of postexcretion losses, particularly for N.

Estimates of the composition and amounts of separate fecal and urinary components of year-round daily average excretion for typical Holstein cows are shown in Table 2. Several additional characteristics of manure are also included, such as ADF, NDF, protein, and nonstructural carbohydrates. In using Tables 1 and 2, it is important to consider that water content of manures is the most variable constituent. The amount of daily or yearly excretion can be more accurately predicted than percentage composition (60).

Manure Distribution

Many large dairies have insufficient acreage for recycling nutrients and, thus, must haul manure nutrients to locations off the farm. Dairies in dry regions frequently benefit from natural drying of manure in open lots, allowing manure to be more easily scraped and hauled. Some dairies contract with nearby neighbors who can utilize a portion of the dairy farm's solid manure (dry or wet) in place of commercial fertilizer. If environmental regulations require a total farm nutrient budget, records must be kept of where the farm's manure nutrients were distributed.

Animals distribute their manure naturally, usually assumed to be in proportion to time spent in respective locations. However, data on relative distribution are lacking. When cows have access to high intensity dirt lots or to pastures, separate accounting of nutrients for those lots or pastures may be required apart from feeding barns and milking parlors in which manure is completely collected. Similarly, estimation of the proportion of daily manure deposited in milking parlors and adjacent holding areas is critical if the manure management system for these areas is designed separately from feeding and loafing areas. Many extension specialists hypothesize that cows defecate and urinate less in the milking parlor area relative to amount of manure voided per unit of time in feeding and resting areas (Van Horn, 1993, personal communication with Western Region US Dairy Extension Specialists). More data are needed on behavioral patterns of manure deposition under different housing and management conditions.

Solids Separation

Removal of manure from animal pens by flushing with water is an easy and clean way to handle manure. However, this process results in a larger volume of manure slurry to be managed. Separation of the coarse solids from flushed manure is potentially important for several reasons:

1. To remove large particles and sand that could plug or damage distribution nozzles in irrigation equipment.
2. To reduce organic loading on anaerobic and aerobic lagoons.
3. To capture a fibrous by-product with some N and mineral content for uses such as bedding for free stalls, part of the feed for cattle on maintenance diets, plant-potting compost, and fertilizers.

The primary benefit of separation of solids from liquid is the production of two fractions that are inherently more manageable than the original slurry. The most popular systems used to remove a portion of the solids from manure slurries are mechanical separation and sedimentation basins.

Screening of Manure Solids. Stationary screens, which are most common, usually remove 20 to 30% of the organic matter from liquid dairy manure (36). Auvermann and Sweeten (4), using flushed dairy manures, evaluated several mechanical separators being used on farms and found that the three most effective screening systems reduced organic solids by 21% [total solids (TS) by 16.4%].

Pain et al. (46) evaluated the use of a vibrating screen for dairy and swine waste slurries containing up to 12% TS and found that the screens were ineffective above 8% TS because the slurry accumulated on top of the screen. Holmberg et al. (19) separated flushed swine manure averaging 2.9% TS using vibrating screens of 45.7-cm diameter with five mesh

TABLE 2. Feces, urine, and combined characteristics for typical Holstein cow consuming 17.8 kg/d of DM and producing 22.7 kg/d of milk.[1]

	Composition			Daily excretion		
Characteristic or nutrient	Feces	Urine	Combined	Feces	Urine	Combined
	(%)			(kg)		
Total excretion, wet				34.0	22.7	56.7
DM	17.4	4.3	12.2	5.92	.98	6.90
Water content	82.6	95.7	87.8	28.1	21.7	49.8
	(% of DM)					
VS	89	47	83	5.27	.46	5.73
ADF, kg	31.2	0	26.8	1.85	0	1.85
NDF	53.2	0	45.6	3.15	0	3.15
Estimated NSC	15.9	26.9	17.5	.94	.26	1.21
Crude fat	3.8	0	3.3	.23	0	.23
Carbon[2]	42.4	20.0	38.9	2.5	.2	2.7
				(g)		
N	2.7	14.0	4.3	160	136	296
NH_3 N	.14	10.7	1.63	8	104	112
Estimated true protein N	2.6	3.3	2.7	151	32	183
P	.85	.37	.78	50	4	54
K	.53	13.96	2.43	31	137	168
Ca	2.00	.023	1.72	118	.3	119
Na	.16	4.65	.80	9	46	55
Mg	.70	.46	.67	41	5	46
Fe	.11			7.4		7.6[3]
	(ppm)					
Zn	89			.6		1.1[3]
Cu	14			.1		.29[3]
Mn	62			.4		1.2[3]
Mo	2			.02		.05[3]
Bo						.16[3]
Cd						.001[3]
Ni						.065[3]
				(Mcal)		
Estimated GE				25	2	27

[1]Data, except as noted, adapted from Tomlinson (60), Morse et al. (39), and Table 1 to cows consuming diets with 50% of DM from corn silage; diets at 14.5% CP, .43% P, .82% Ca, 1.13% K, .37% Na, and .27% Mg. VS = Volatile solids, NSC = nonstructural carbohydrate, and GE = gross energy.

[2]Calculated from estimated carbon content of nutrient fractions obtained from Maynard et al. (30).

[3]From American Society of Agricultural Engineers (1).

sizes, ranging from 2.4 mm down to .1 mm. Flow rates of the slurry were varied from 37.5 to 150 L/min. Increased flow rate and decreased screen size increased the amount of solids removed; organic solids removal ranged from 14 to 70%, N removal from 2.5 to 50.9%, and P removal from 2.5 to 58.9%.

Powers (49) wet-screened feces of individual cows over a series of vibrating screens of 3.35, 2.00, 1.40, 1.00, and .50 mm to evaluate DM, N, and P recovery in screened solids. The percentages of fecal DM (TS) recovered were 14.6, 9.4, 2.8, 4.3, and 8.6%, respectively, for 39.7% total removal. The percentages of N associated with DM on the screens were 5.7, 3.1, .8, 1.3, and 2.8% (13.7% total); the percentages of P removal were 2.2, 1.2, .3, .6, and 1.5%, respectively (5.8% total). The total for the two larger screen sizes, which could be considered equivalent to that obtained with screen sizes currently being used on dairy farms, shows removal of 24.0% of DM, 8.8% of N, and 3.4% of P. The conclusion is that screens remove a much smaller percentage of N and P than DM; most manure fertilizer nutrients are soluble and stay with the liquid effluent.

The expected DM percentage of dairy manure fiber recovered from commercially available stationary screens usually ranges from 13 to 20%, which, after drainage or use of a screw press, may be increased to 25 to 28%. Approximate ranges in composition of screened manure solids from systems removing approximately 20% of total DM are shown in Table 3.

TABLE 3. Composition of screened manure solids.[1]

Nutrient	(% of DM)
Ash	7.0–13.4
N	1.0–1.6
P	.12–.15
K	.16–.22
NDF	77.7–83.5
ADF	50.0–60.0
ADL	12.9–15.1

[1]Moisture content usually about 75.0% (DM, 25%) after drainage from stack. ADL = Acid detergent lignin. References (4, 21, 36, 41, 49) and individual farm analyses.

Sedimentation of Manure Solids. Moore et al. (38) measured settling efficiency with time for manures from several livestock species and reported that over 60% of TS from a dairy slurry can be removed in the first 10 min of settling. Sedimentation of swine slurry containing 4 to 5% TS was evaluated for solids and P_2O_5 removal by Voermans and de Kleijn (65). Under controlled conditions, TS in the effluent was reduced 24 to 59% coupled with a 23 to 57% reduction of P_2O_5. Greater removal of both P_2O_5 and TS occurred when ambient temperature was below 16°C. Voermans and de Kleijn (65) combined settling with the use of polyelectrolytes to determine the extent to which solids and nutrient removal could be improved and found an average reduction of 70% TS and 88% P_2O_5, regardless of ambient temperature.

Safley and Owens (51) sedimented poultry slurry of 5 to 6% TS and observed a 57% reduction in TS. They found that, with poultry manure above 7% TS, little or no settling occurred because the product remained a homogeneous mixture.

Powers (49) simulated flushed dairy manure for individual cows by adding excreted proportions of urine and feces, diluting them to a 1-L volume (approximately 1.5% TS), agitating, and evaluating sedimentation at various times. Powers (49) found that 65% of solids in the slurry, which contained 40% of N, settled out after 1 h. Volumetric readings within the hour indicated that, although not sampled, of the 65% solids recovered after 1 h, 89% settled after only 5 min, 91% after 10 min, and 95% after 20 min. Using similar procedures, Montoya (35) found a 65% reduction in TS by sedimentation with dairy manure diluted to an average of 1.5% TS and obtained mean recoveries of 30% N, 12% P, and 15% K.

If nutrient reduction in the effluent is the objective, the potential exists for greater recovery of TS and nutrients from flushed manures with sedimentation than with screening. Wastewaters need to be held for about 10 min to allow for adequate settling. Simply slowing flow causes sedimentation of solids, but not to the extent demonstrated as possible by Montoya (35) and Powers (49).

Jones et al. (24) demonstrated the potential for flocculating agents to increase precipitation of solid materials and soluble mineral elements from manure slurries. Montoya (35) added agricultural lime ($CaCO_3$) and $Fe_2(SO_4)_3$ or CaO and $Fe_2(SO_4)_3$ to diluted manure in 1-L graduated cylinders and obtained recoveries with the sediment of approximately 90% of

TS, 70% of N, 75% of P, and 30% of K. The potential for increasing sedimentation of solids and nutrients from flushed manures with addition of flocculating agents or $CaCO_3$ and $Fe_2(SO_4)_3$ appear promising but need further study.

Resource Potential

Fertilizer Value. Current extension education programs for dairy manure management primarily emphasize the fate of N and P because of regulatory concerns with these nutrients. Nutrient budgeting programs account for manure nutrient production and plan for environmentally acceptable nutrient losses, using adequate crop production to utilize fully the most environmentally sensitive nutrient [e.g., (63)]. Large dairies frequently have limited acreage on which to apply manure. Hence, only the value of the most sensitive nutrient (most often N) is fully realized, and other nutrients, which may be applied in excess, are not utilized most effectively. In such systems, the fertilizer value of nutrients used for crop production is the resource recovered plus the intangible benefits associated with environmental compliance.

One method of estimating the resource value of manure is to assign a fertilizer value to the yearly production of N, P, and K, the most valued fertilizer nutrients. For example, based on assumed values of \$.66/kg of N, \$1.32/kg of P, and \$.33/kg of K, the range in value for N, P, and K in manure illustrated in Table 1 would be \$107 to \$146/yr per cow. In practice, realized values probably are only about half these amounts because of N volatilization and less than optimal use of other nutrients.

The organic matter in manure has some value in fertilizer, but this value is difficult to quantify. Manure organic matter aids water retention, and organically bound nutrients do not leach easily.

Energy Value. Manure, in a relatively dry form, may be burnt directly as fuel. The use of manure as fuel is an ancient practice that is still utilized in many developing countries. In the 1800s, westward pioneers crossing the prairies of the US used buffalo chips for fuel. The first large-scale resource recovery project in the world to burn cattle manure as fuel is in the Imperial Valley of southern California (H. H. Van Horn, 1990, personal communication with Western Power Group and National Energy Associates, El Centro, CA). The \$46.2 million plant was developed jointly by the Western Power Group and the National Energy Associates. It was designed by employees of the Lurgi Corporation (part of Metallgesellschaft AG, Germany), an international process design, engineering, and construction firm, who have successfully designed plants in Europe to burn sewage sludge for electricity generation. Scraped, feedlot manure of beef cattle is delivered to the plant and is then piled and compacted to allow moisture to become uniformly distributed, retarding degradation and reducing the potential for spontaneous combustion. Approximately 80,000 tons of manure are on site at all times. The power generator is connected to the Imperial Irrigation District's electrical power transmission system. In addition to supplying in-house electrical needs, the plant generates about 15 MW of power, a quantity sufficient to meet the electrical needs of 20,000 homes.

The energy value of manure is a potential resource that, however, is usually discarded. Figure 2 shows how a typical cow producing 22.7 kg milk/d partitions DM, volatile solids (VS, organic matter), C, and Mcal of gross energy (GE) during digestion and metabolism. Of the nutrients consumed that yield dietary energy, approximately 5% is eructated from the rumen as methane, 20% is secreted into milk at overall mean yield, 40% is lost as heat (maintenance energy plus heat of fermentation), and 35% is excreted in manure, of which approximately 93% is in feces.

An important question to be answered is whether the potential energy in manure is economically recoverable. Anaerobic digestion of manure to produce biogas, which can be captured and used as a fuel, is the most feasible method to recover the energy value from manure on individual farms. Fabian (12) estimated biogas production of .35 L of biogas/g of VS input when hydraulic retention times are ≥20 d. Thus, with VS production, as in Table 2, biogas potential for the typical cow was estimated at

$$5.73 \text{ kg of VS/d} \times 1000 \text{ g/kg} \times .35 \text{ L/g of VS} = 2005 \text{ L of biogas.}$$

Considering the biogas to contain 60% methane and methane to contain 8.90 kcal/L (16), daily production of kilocalories would be 8.90

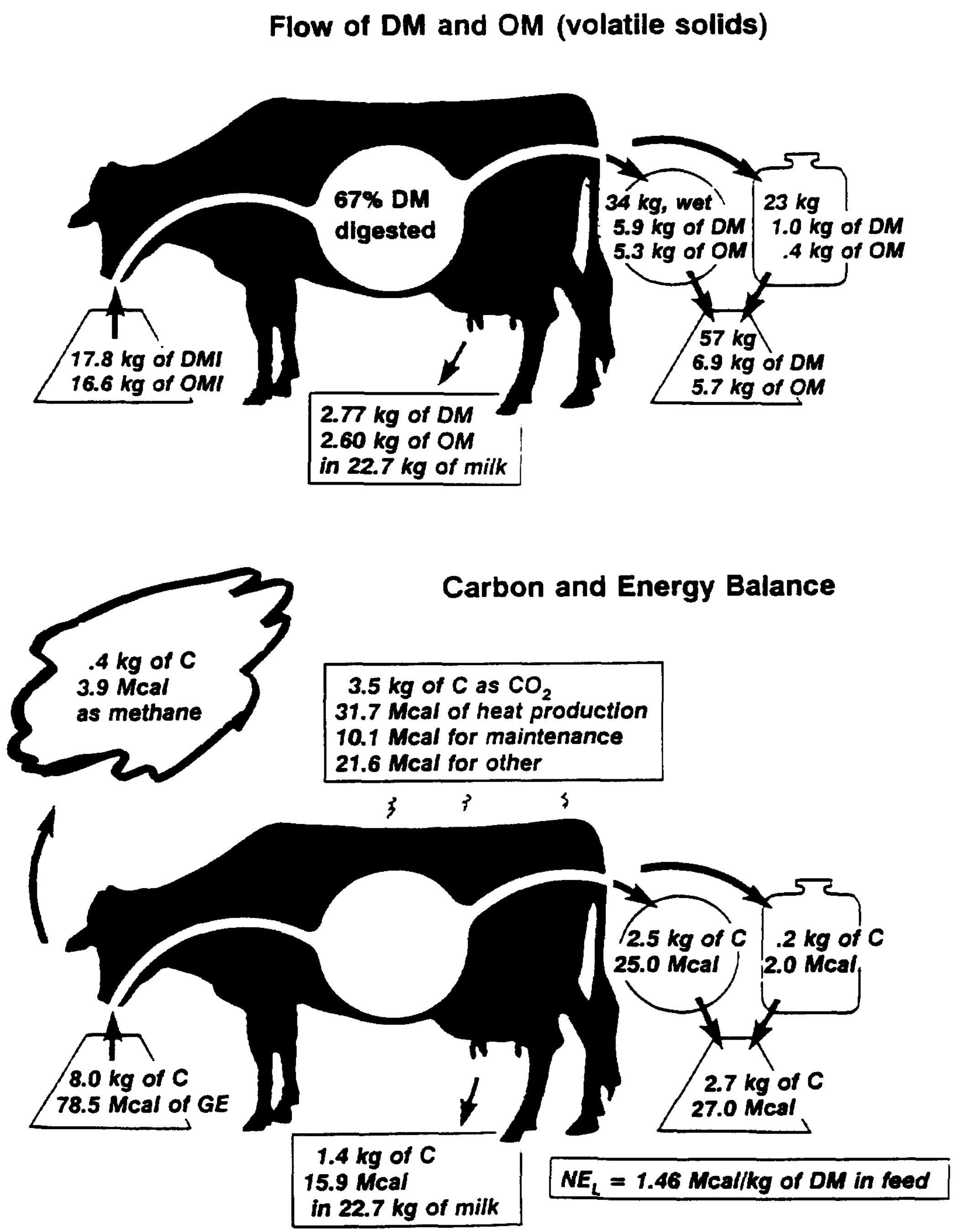

Figure 2. Estimated daily flow of DM, organic matter, energy, and carbon through a typical Holstein cow (typical of year-round amounts when extrapolated to 365 d). Energy and carbon balance data adapted from Flatt et al. (14) to agree with DM and organic matter excretion data from Tomlinson (60); some carbon compositions were estimated from average carbon content of nutrients (30). OMI = Organic matter intake; GE = gross energy.

kcal/L × .60 × 2005 L = 10,707 kcal = 10.71 Mcal. The energy recovered in relation to megacalories of GE in the original manure can be calculated; original manure contained 27.0 Mcal in VS, and estimated recovery in anaerobic digestion = 10.71 ÷ 27.0 = 39.7%. If a large dairy had 1000 cows generating 2005 m^3 of biogas/d and converted this energy to electricity with an efficiency of 1.0 kWh/.934 m^3 of biogas (25), 2005/.934 = 2150 kWh/d per 1000 cows would be generated; estimated value would be $129 ($.06/kWh) to $215 ($.10/kWh). This value converts to $47 to $78/yr per cow. Relative returns may be even greater if the biogas can be utilized as a substitute for other fuels used to produce heat.

Feed and Bedding Values. Anthony (2) and Fontenot (15) reviewed research on the value of animal wastes as ruminant feedstuffs. The ranking of animal wastes for ruminant feed in descending order of nutritive value was excreta of young poultry, deep litter of young poultry, hog feces, excreta of laying hens, hog and layer manure solids, and excrement of cattle (15). The estimated feeding value of wastes, based on in vitro cell-wall digestibility, was in general agreement with that ranking, except that swine wastes ranked lower than wastes of caged laying hens. Cell walls from feces of nonruminants are generally more digestible than those from ruminants (15). However, cell-wall digestibility is similar for wastes from cattle fed all-concentrate diets and wastes from nonruminants, resulting in similar TDN content.

Fibrous solids, separated from dairy manure by screening, are more amenable to use as feed than is manure. However, digestibility is low, and solids of this type have not been of much value as a source of digestible nutrients because the digestible energy value is too low to support production above maintenance (21, 41). However, screened solids may have potential as a diluent for use with dry cows or heifers fed corn silage or other high energy feedstuffs that promote overfattening if offered free choice and unamended.

Use of fibrous solids for bedding (e.g., in free stalls) is feasible (5, 13, 55) and currently is substantial on dairy farms [e.g., (45)].

ENVIRONMENTAL COMPONENTS

Livestock manures are considered to be significant pollutants of the nation's waters [e.g., (29)]. For this reason, regulatory oversight by the US Environmental Protection Agency and cooperating state agencies has become a significant force to ensure that dairies manage their manure so that surface and groundwater qualities are not compromised. Odors and other emissions from dairy manure management systems are further causes of environmental concern.

Water Management

Water use is essential in all dairies. Drinking water is indispensable; some water is necessary for cleaning and sanitation procedures; moderate amounts are important in periods of heat stress for evaporative cooling of cows to improve their production and health; water can be used in labor-saving methods to move manure and clean barns by flushing in properly designed facilities; and additional amounts may be used for irrigation of crops grown to recycle manure nutrients. Extensive water use, however, increases the potential for surface runoff and penetration to groundwater, with possible environmental impacts offsite. Heightened environmental concerns and the need for resource conservation have resulted in many regions in the implementation of permits for water use and in other regulatory controls. Thus, it is important to quantify essential water uses and various other uses that are important to dairy farm management and to consider whether reduction of water use in one practice reduces overall water consumption. For example, reduction or reuse of some of the water used for manure flushing, cow cooling, and cow washing may not save total water if all of the wastewater is currently directed to irrigation of crops because water needed for irrigation would then have to be supplied from other sources (62).

The sample budgets for water use (Table 4) illustrate that water usage on dairies is probably small compared with irrigation needs when there are 12.1 ha (30 acres) of spray field available per 100 cows, a common minimum area for adequate crop production to utilize the manure N from 100 cows (63). Conversely, the amounts used in most dairy systems in warm climates would be large and unmanageable if application through irrigation were not an option or if less acreage were available for irrigation than would be needed for application of all manure nutrients. If the water and manure

. TABLE 4. Estimated water budgets for management systems (all values in liters unless otherwise noted).[1]

Water use by the dairy	System	
	Flush, typical needs during hot season	Minimum water use
Drinking water for cows	95	95
Cleaning cows	120	0
Cleaning milking equipment	11	11
Cleaning milking parlor	114	24
Sprinklers for cooling	95	45
Flushing manure	230	0
Total use per cow per d	665	175
Total use per 100 cows per d	66,500	17,500
Use per 100 cows per wk	465,500	122,500
Water in milk per 100 cows per wk	17,000	17,000
Estimated evaporation at 20% of use	93,100	24,550
Average rainfall and watershed drainage into storage facility per 100 cows per wk	100,000	50,000
Wastewater produced from 100 cows per wk	455,400	131,000
m^3/wk per 100 cows	455.4	131.0
Water, cm/wk if 12 ha in spray field	.38	.11

[1]Adapted from Van Horn et al. (62).

nutrients cannot be used through irrigation, a system based on flushing manure should not be utilized. However, some irrigation usually is possible. Costs for construction of storage structures for holding wastewater until used for irrigation warrant consideration.

Table 4 includes a theoretical minimum amount of water use on a dairy. This system implies that cows are clean enough and cool enough that sprinkler washers are not needed while the cows are being held for milking and that all of the manure is scraped and hauled to manure disposal fields or transported off the dairy. Intermediate steps between flush and other systems that might be taken include 1) scraping and hauling manure from high use areas such as the feeding barn so that this manure can be managed off the dairy; 2) using wastewater in lieu of fresh water to flush manure from feeding areas and free-stall barns; and 3) using a housing system that will keep cows clean enough that cow washers are not needed prior to milking. Such a system, however, may still require use of alternating sprinklers and fans to keep crowded cows cool during hot weather.

Odor Control

Volatile odorous compounds emitted from manure during transport, storage, treatment, and disposal have become an acute public relations problem for animal agriculture. Odorous compounds usually are present at such low levels (parts per million or parts per billion) that they are not toxic at the concentrations found downwind of livestock production facilities. Thus, the problem depends largely on subjective factors, such as how much the smell bothers people or the "nuisance value" of the odor. Nuisance is generally defined as "interference with the normal use and enjoyment of property" (58). Odor nuisance is dependent on odor quality and quantity, visual perception of odor from applied or stored manure, and exposure interval and frequency. Flies often add to an odor nuisance, and the two problems may be difficult to separate in the minds of complainants. Odor complaints range from casual comments, indicating displeasure, to major lawsuits and court orders that have the potential to terminate the affected food animal enterprises (58).

The US Environmental Protection Agency does not regulate odors. However, odor is regulated as a nuisance in every state in the US. A distinction is often made between a "public nuisance" (an infringement on the rights of numerous people) and a "private nuisance" (an infringement on the rights of a small number of people). States regulate odors as a public nuisance through air pollution control and public health protection statutes. Sweeten (56) listed 11 states that had, by 1988, adopted quantitative criteria for regulating odor intensity at the property line (i.e., Colorado, Connecticut, Illinois, Kentucky, Louisiana, Minnesota, Missouri, North Dakota, Nevada, Oregon, and Wyoming). Many other states are considering similar regulations.

Definitive measures of odor are needed to evaluate the extent of an odor nuisance. One problem is in defining what to measure. Over 75 odorous compounds, in varying proportions, have been identified around manure storage areas (31, 32, 56). Volatile fatty acids, phenols, and sulfides are thought to be the major odor-causing compounds. However, typical chemical analyses measure concentrations of only a small number of constituents in the complex mixture that contributes to the odor people identify by smell. A second problem is how to obtain consensus on which odors are strong enough to be a nuisance (33).

One approach to odor measurement has been to determine concentrations of individual odorous compounds by gas chromatography or adsorption techniques and to correlate these concentrations with perceived odor intensity. For example, concentrations of ammonia and propionic acid were correlated with odor intensity in some studies; hydrogen sulfide and ammonia and methyl amine were highly correlated in another study (56). Such measurements often correlate well with observed odor intensity at specific sites but do not transfer well from one site to another. Consequently, current odor measurement technology has turned to sensory methods (i.e., using the human nose).

One device for the estimation of odor intensity that can be used on-site is the scentometer (33), which is based upon an evaluator acclimating his or her sense of smell to odor-free air and progressively introducing higher proportions of odorous air, mixed with odor-free air, until an odor is first detected coming through the device (termed the threshold concentration). Threshold is usually defined in terms of the number of dilutions of odorous air with odor-free air. The scentometer has received widespread application in animal waste odor evaluation. Sweeten (56) reported that eight states and seven municipalities had adopted odor regulations based on the scentometer. However, use of the scentometer has several limitations. Individuals differ greatly in the ability to detect odors.

A drawback to testing by dilution to threshold is that, by definition, evaluator disagreement is widest at the threshold level. To adjust data scatter around threshold values, it is possible to compare the odor to be measured with a reference compound such as butanol (butyl alcohol). Sweeten et al. (59) described an alternative procedure whereby a panel of odor evaluators are sequentially presented with six to eight concentrations of butanol vapor in air and are asked to determine whether the butanol odor has greater, equal, or lesser intensity than the ambient odor sample. The responses define the concentration of butanol that best matches the intensity of the odorous ambient air. One state (Louisiana) has adopted property line odor regulations based on a butanol olfactometer (56).

Because odors, as a nuisance, are being defined relative to threshold concentrations, control of odorous emissions to levels less than threshold is the primary objective of odor control. Odor control methods fall into three broad categories: 1) control of odor dispersion, 2) odor capture and treatment, and 3) treatment of manure (32, 56).

Control of odor dispersion is primarily a function of site selection, system design and construction, and manure handling methods (e.g., spray field application may provoke more odor drift than soil incorporation). Odor capture and treatment methods include containment, wet scrubbing, packed-bed adsorption, and soil filter fields (56, 57). These methods are not well suited for open housing conditions on most dairies. Manure treatment methods include anaerobic digestion, aeration, and biochemical treatment.

Manure odors are caused principally by intermediate metabolites of anaerobic decomposition. Anaerobic degradation is initiated within the animal and continues in feces droppings, manure piles, and storage facilities. If odorous compounds can be confined within the fermentation medium until the fermentation is far along, many of the intermediary odorous compounds will be metabolized to less odorous compounds or will exist in lesser concentrations. For example, anaerobic lagoons, although not free of odors, are seldom the cause of an odor problem. However, overloaded or shock-loaded lagoons are more likely to have objectionable odors. Anaerobic lagoon odors are most common in the spring when the temperature rises and when manure, accumulated during the winter, undergoes rapid decomposition. Where practical, lagoons should be located as far as possible from neighboring residences, roads, and other odor-sensitive areas. Shielding lagoons from view is also helpful in reducing the perception of an odor nuisance.

Anaerobic digestion systems in which biogas fuel is generated do an excellent job of processing odorous compounds. In some cases, such systems are being installed with odor control as the primary objective and energy recovery a by-product that helps defray the cost of installation. Anaerobic digestion of swine manure for methane gas production reduced the odor emission rate from land-applied digested slurry by 91% compared with that from slurry stored in untreated pits. Anaerobic digestion also reduced the time for odor dissipation (as measured at the 50% panelist threshold, the point at which 50% of the panel detected odor) from 72 to 24 h [cited by (56)].

Mechanical aeration in oxidation ditches or in the second stage of a lagoon system has long been recognized as an effective method for odor control. Phillips et al. (48) reported that aeration rapidly reduced odors that were due to hydrogen sulfide and methanthiol from anaerobic swine manure. However, aeration had little effect on less volatile and less offensive compounds such as phenols. Aeration of manure was recommended just prior to land spreading to reduce odors from field application.

Biochemicals used to control odors have included masking agents (disguising one odor with another, more acceptable odor), oxidizing agents (e.g., ozone, potassium permanganate, and chlorine-containing compounds), digestive deodorants, and feed additives (34, 56). Digestive deodorants are the most prevalent and necessitate that the added bacteria become the predominant strain. Some of the products currently marketed seem to be helpful in controlling odor, but others are not. In cases in which they were effective, costs were high, e.g., $6 to $23/yr per hog (69). More research is needed in the area of additives to control odors to improve efficacy and to evaluate cost effectiveness.

Ammonia Emissions

Two primary forms of N exist in manure, ammonia and organic N. The major source of ammonia is urea from urine, or uric acid in the case of birds, which can be easily converted to NH_3, a gas. Urea plus ammonia N from urine usually accounts for 41 to 49% of total N excreted in manure (Tables 1 and 2). In aqueous solution, NH_3 reacts with acid (H^+) to form an ion (NH_4^+), which is not gaseous. Thus, the chemical equilibrium in an acid environment promotes rapid conversion of NH_3 to NH_4^+ with little loss of NH_3 to the atmosphere. However, most animal manures, lagoons, and feedlot surfaces have a pH >7.0, making H^+ scarce and, thus, permitting rapid loss of NH_3 to the atmosphere. As a consequence, N losses from animal manures can easily reach 50 to 75% [e.g., (53, 54)], most as NH_3 before NH_3 is converted to NO_3^- through nitrification. If large amounts of manure are being applied to fields at one time (e.g., injected or plowed down in winter before corn is planted), possible leaching losses of NO_3^- to groundwater can be reduced through use of nitrification inhibitors (54).

In addition to ammonia volatilization, airborne losses include denitrification, an anaerobic process in which bacteria convert nitrate to nitrous oxide and then to N gas. Denitrification is a major process of N removal from soils. However, enhancing N loss by denitrification on farms is not feasible, and nitrous oxide, an intermediary gas that may be emitted in the process, has been implicated as a significant contributor to global warming. Moore and Gamroth (37) found denitrification losses from 7 to 28% of applied N in wet or poorly drained soils.

Thus far, concern with manure N in the US has been to avoid losses to ground and surface waters in order to avoid nitrate contamination of drinking water supplies and overfertilization of surface waters. Most manure management systems have been designed to recycle manure N through crop production, and gaseous ammonia losses have been considered to be an economic rather than an environmental concern. To the extent that N is needed by crops produced on available land at the livestock production unit, strategies to conserve N and to maximize the fertilizer value of manure should be utilized. However, for farms with insufficient crop acreage and, hence, excess N, encouraging emissions to the atmosphere is one way to reduce potential transfer to aquatic systems.

An important question to be answered is whether it is important to minimize low emissions of ammonia to the atmosphere, and, if not, whether livestock producers be encouraged to use manure management procedures to volatilize more ammonia? In Europe, atmospheric ammonia concentrations have become a public concern through their perceived contribution to acid rain and the destruction of forests (3, 47, 50). Consequently, European livestock and poultry operations are being required to utilize practices to minimize ammonia losses to the atmosphere.

Ammonia can be toxic to cells, and the potential exists for plant damage if excessive ammonia is released after manure application. Also, excessive ammonia concentrations in closed buildings used to house large numbers of animals may lower animal performance and may be a potential health hazard for workers. Atmospheric ammonia has been known to cause blindness in chicks and turkey poults. Thus, it is important to avoid ammonia buildup where animals are confined and people work.

To understand the environmental impacts of volatilized ammonia, it is important to know the fate of ammonia once it is emitted from storage structures, pastures, and manured fields. Most of the following information was taken from an extensive review by Elliott et al. (10), prepared for use in development of manure management policy for the Chesapeake Bay area.

The fate of ammonia is linked to three possible processes: dry deposition, wet deposition, and movement into the upper atmosphere above the cloud layer. The latter process represents a very small percentage of total volatilized N because almost all of the volatilized N returns to the earth, mostly within 3 to 5 d (3, 10).

Some ammonia gas is sorbed directly (dry deposition) by aquatic systems, soils, or settling particulate matter in the atmosphere and is deposited locally before ammonia can dissolve in atmospheric water vapor. Plant species vary in the ability to absorb ammonia directly. In general, agricultural plants scavenge atmospheric ammonia, and corn has the highest rate of absorption. Elevated gaseous ammonia concentrations are thought to be detrimental to forests; for example, in Europe, when the needles of *Pinus nigra* take up NH_4^+, K and Mg deficiencies occur, and premature shedding of needles is promoted (50). Additionally, excess ammonia has been linked to increased susceptibility to frost damage and fungal diseases.

Most volatilized ammonia is dissolved in water vapor in the lower atmosphere and washed back to earth by rainfall. During this process, ammonia neutralizes the acidity of the rainwater. In industrial regions with somewhat acid rainfall (e.g., Pennsylvania), neutralization is one potential benefit of ammonia release (10). If techniques were used to promote ammonia volatilization, a portion would be redeposited from the atmosphere to nonagricultural areas that are poor in N, such as forests. The resulting increase in soil fertility would be a potential benefit of increased volatilization. However, soil pH would drop over time, just as continued application of fertilizers containing ammonia acidifies agricultural soils. Researchers in The Netherlands think that long-term soil acidification can alter forest species distribution and vitality (50).

Prediction of the overall, global consequences of increased manure ammonia emissions is hampered by a lack of historical data on changes in the atmospheric content of ammonia. However, 130 yr of data on rainwater composition [data cited by (10)] indicates that deposition of ammonia has remained relatively constant, despite dramatic increases in fertilizer use and livestock numbers. Some researchers think that input of ammonia to the atmosphere from human activities is small relative to that from breakdown of materials in the soil that naturally contain N. If this hypothesis is true, a sizeable increase in ammonia released from livestock wastes would be necessary to effect a substantial change in the global atmospheric ammonia inventory.

Current data do not prompt concern about negative effects on the environment caused by diffuse ammonia emissions from animal manures, at least in North America. However, local concern about animal, human, and plant health is warranted when ammonia concentrations are high (10).

Methane Emissions

Methane emissions from animal production systems do not present an odor control problem because methane is odorless. The concern with methane relates to its contribution to global warming (11, 22, 66). The earth is blanketed by a layer of gases that is relatively open to penetration by incoming short-wave solar energy. The percentage of this energy that is radiated from the earth back to space as long-wave radiation is determined by the concentration in the atmosphere of several of these gases. The principal long-wave, energy-absorbing gases are carbon dioxide, methane, chlorofluorocarbons, and nitrous oxide. These gases are termed "greenhouse" gases because they absorb the long-wave radiation, just as glass in a greenhouse absorbs radiation, rather than allowing the heat to be radiated away from the earth. The steady enrichment of the atmosphere with greenhouse gases creates a warming effect, referred to as global warming. The actual contribution of greenhouse gasses to global warming is not precisely known because the extent to which their emissions affect global warming is still a topic of much debate.

Carbon dioxide is the most abundant greenhouse gas and is being added in the greatest quantity; carbon dioxide is expected to cause about 50% of the global warming occurring in the next half century. Methane is generally held to be the second most important greenhouse gas and is expected to contribute 18% of future warming (22). Indeed, molecule for molecule, methane traps 25 times as much of the sun's heat in the atmosphere as does carbon dioxide. Thus, methane is estimated to contribute 18% of future warming from <1% of the total greenhouse gas emissions. In addition to warming effects, increased atmospheric methane will likely be detrimental by increasing ozone pollution near the earth's surface and, conversely, by decreasing ozone in the stratosphere, which shields the earth from harmful solar ultraviolet radiation.

Samples of air taken from deep arctic ice cores show that atmospheric methane has been stable for thousands of years until about 200 to 300 yr ago, after which time its concentration has more than doubled. Recent years show an increase of approximately 1%/yr. A total of about 550 million tonnes of methane are entering the atmosphere each year, and about 460 million tonnes are consumed in the atmosphere and by soils. The largest single source of atmospheric methane (about 23%) appears to be that naturally produced by bogs, swamps, and wetlands. This, plus an estimated 11.8% from oceans, wild animals and termites, and incomplete natural burning constitute naturally occurring sources of methane; the remaining 65% is caused by anthropogenic activities [data cited by (22)].

The origin of methane produced by animals is microbial action in the gastrointestinal tract, which occurs to varying degrees in all animals. Major fermentative digestion, allowing utilization of fibrous dietary components, occurs in ruminants. This digestion, coupled with large body sizes, dry matter intakes, and animal numbers, results in 95% of animal methane emissions arising from ruminants, about 80% from the Bovidae family alone. Sheep and goats account for another 12%, and horses and pigs contribute about 2 and 1%, respectively (9).

Energy losses through methane produced in the rumen are usually 6 to 8% of gross energy intake in cattle consuming high forage diets; the greatest percentage of losses occurs when forage is of low digestibility (11). Dairy cows fed moderately high concentrate diets convert about 5% of their gross energy intake into methane and belch this methane into the atmosphere.

The methane produced by animals and animal manures constitutes 16.4% of estimated annual methane emissions [from (22)], which translates roughly to 2.9% of the estimated contribution of all greenhouse gases to global warming (i.e., 16.4% of 18%, the projected contribution of all methane sources). Although an extremely small part of the total, some agencies are investigating the feasibility of reducing animal-related methane emissions.

METHODS FOR PROCESSING AND RESOURCE RECOVERY

Manure Processing on Pasture

To allay fears that unprocessed animal manures are a risk factor to humans, it is important to point out the many natural biological pathways at work in pastures and in soils utilized for field crop production that effectively process manures. Anaerobic processing (degradation) of undigested fecal organic matter is already underway in the lower gut of animals before feces are voided. This process continues after defecation to the extent that anaerobic conditions are maintained; e.g., the centers of manure droppings firm enough to stack may remain anaerobic. Wherever oxygen permeates, aerobic microbes take over the degradation process, and anaerobic activity decreases. Insects, such as dung beetles, and worms also contribute. Aerobic systems and soil associations effectively oxidize most odorous compounds rapidly, and, when manure applied to pastures is spread across enough area that remaining odorous compounds are effectively diluted, odors under pasture conditions are usually not a problem.

Thus, agriculture is based on biological systems that effectively process manure nutrients and other biomass in cost-effective, environmentally acceptable ways. The public sector needs to be aware of this process and to monitor agricultural systems based on real concerns and not perception to avoid imposing unnecessarily costly processing methodology. In fact, many municipal systems are turning to agricultural methods and contracting with agricultural units to receive wastewater or sludge to reduce the cost of municipal waste disposal [e.g., (20, 64)].

Total nutrient budgets for pasture conditions seldom show excessive nutrient applications unless commercial fertilizer nutrients are applied in addition to manure or unless pastures become holding areas to accommodate a relatively large number of cattle being fed primarily from feed sources obtained off site. Cattle cannot excrete more nutrients than they take in, at least for extended periods. However, problems in surface water quality can occur when cattle congregate in or near waterways. Examples of pasture N and P budgets for late pregnant, dry cows are shown in Figure 3. In the illustrated system, nutrients were imported in 2.0 kg of concentrates/d per cow and some commercial fertilizer. Nutrient exports from the pasture in the form of conceptus and weight gain were small in relation to nutrients consumed. Losses of N through volatilization make the use of commercial fertilizer or N fixation via legumes a necessity to prevent N depletion and to maintain forage production. The case for P is somewhat different. Little or no commercial fertilizer P is needed to maintain the balance of P in the soil. In most grazing situations, cattle are supplemented with enough P in mineral supplements to keep the field in balance if manure P is evenly returned to the land area.

Fertilizer for Crop Production

Nutrient budgeting for farms and regions has been proposed as an approach to avoid nutrient loss to groundwaters and to surface waters when combined with soil and water conservation practices [e.g., (26, 27, 67)]. For a farm to be sustainable, its nutrient budget must balance. If a net loss of nutrients occurs, the farm's soils will eventually become depleted. Wallingford (67) indicated that the national N budget for the major US crops has been fairly stable since 1980 but that the P budget is now negative after being positive for most of the 1960s and 1970s. The K budget continues to be strongly negative. Thus, manure nutrients are needed at some locations even though these nutrients are in surplus on some dairy farms. Transportation of surplus manure nutrients off the farm or out of the region is expensive. Because of manure transportation costs, most large dairies try to intensify crop production to utilize manure nutrients efficiently enough to avoid violation of environmental standards. Many agronomists and dairy extension specialists have developed nutrient budgeting materials for dairy farmers to use in planning the amount of crop production (or acreage) needed to utilize manure nutrients effectively [e.g., (63)].

An example of N budgeting for one system is illustrated in Figure 4. Chosen were typical Holstein cows, 22.7 kg/d of milk yield year-round basis, consuming NRC diets with low N (Table 1). The yields for the triple crop program in Figure 4 were selected from data based on the maximum application rate for manure N that was in balance with N in the harvested corn silage, bermudagrass hay, and rye silage (23). The manure resource value in this system is the equivalent commercial fertilizer value of N actually applied to the crop plus the value of additional manure nutrients and water that were needed and utilized by the crops produced. In this example, 565/862 = 66% of the N originally excreted was assumed to have been actually applied. Also, this system maximized N application and assumed that excess P associated with the N would not be detrimental and that manure applications would not be restricted to a P budget; these assumptions would require approximately twice the crop acreage for manure disposal as is required with N budgeting (63).

Nutrient budgets must be developed for each farm based on crops and yields potentially available in that region. In most N budgets, volatilization plus denitrification losses of N should be estimated at >50% of N in original manure excretions, leaving <50% available for crop production (37, 53, 63).

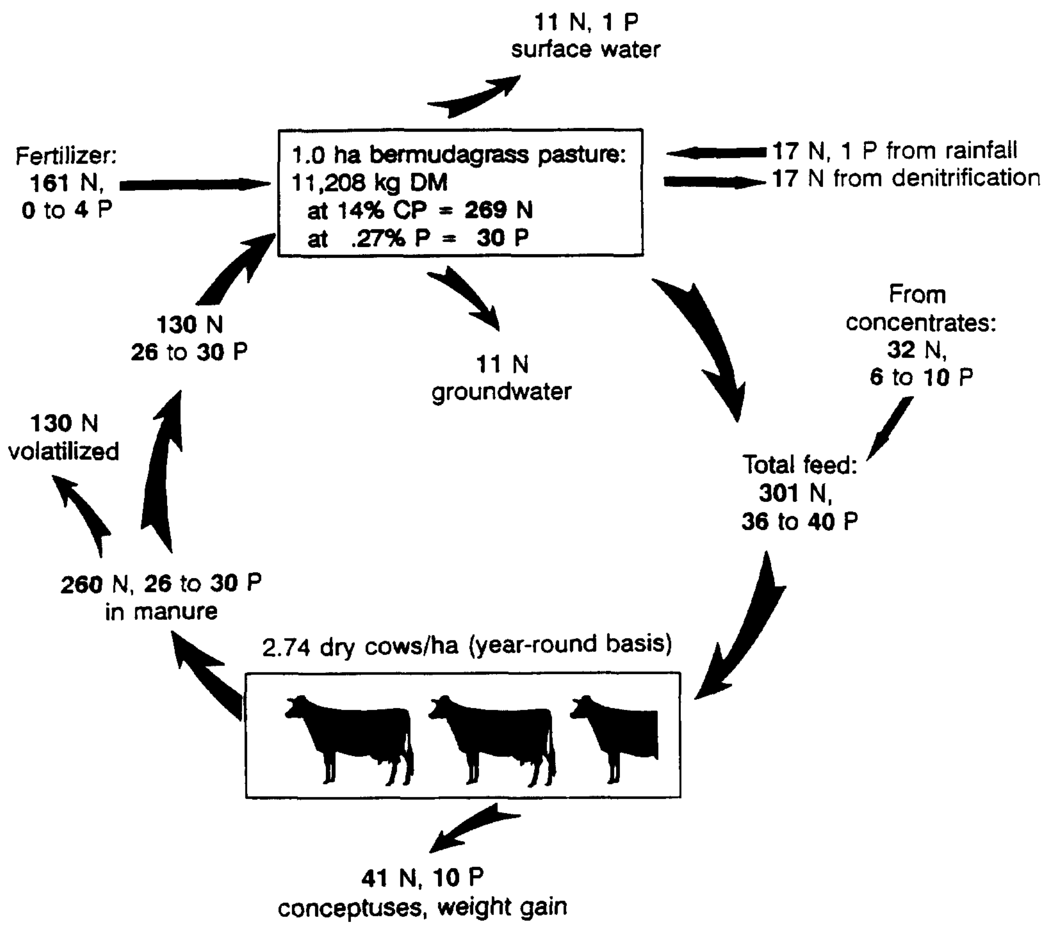

Figure 3. Estimated N and P budgets (kilograms) for dry, pregnant cows consuming 11,208 kg of DM (1.25 Mcal/kg) from bermudagrass pasture annually. Cows were supplemented with 2.0 kg of concentrates/d per cow (1.65 Mcal of NE_L/kg of DM, 10% CP, and .30% or .50% P). Energy was fed to meet NRC (40) standards for 635-kg cows, averaging 250 d in gestation and with a body weight gain of .60 kg/d; N was in excess of minimum CP needs. Accumulation of N in conceptus and body weight gain were from NRC (40) equations, and P was estimated at 6 g/d in conceptus and 4 g/d in body weight gain. Denitrification losses of N were assumed to be offset by N returning in rainfall; P in rainfall is assumed to be from particulate matter. Losses of N to groundwater was the amount calculated to equal 5 ppm of NO_3^- N added to estimated throughput groundwater.

Anaerobic Treatment

Anaerobic lagoons probably represent the most common method of anaerobically processing dairy manure. A regional research project on animal waste as nutrient and energy resources in warm, humid climates summarized many experiments that utilized anaerobic lagoons as a major processing step in the treatment of animal wastes (18). That project evaluated systems of lagoon management that enhanced the use of recycled water. Other factors evaluated were 1) estimates of sludge build-up, 2) crystallization (struvite) build-up in water recycle systems, 3) potential use of lagoons as energy (methane) sources, and 4) lagoon-overland flow treatment.

Overall reductions of chemical oxygen demand, TS, VS, total N, total P, total plate count, coliform count, fecal coliform count, and fecal *Streptococcus* count in three-pond systems were 75, 48, 46, 69, 47, 85, 99, 98, and 99%, respectively. Reduction of VS (organic matter) implies that lost carbon was transferred to the atmosphere in volatile, carbon-containing gases (mostly CO_2, CH_4, and VFA) or retained in the lagoon sludge. Recycling of dairy wastewater resulted in a 20% reduction in potable water usage on the farm (18). Recycled water from a three-pond system did not add to the counts of total plate, total coliform, fecal coliform, fecal *Streptococcus, Salmonella, Shigella, Klebsiella, Staphylococcus,* or other *Streptococcus* on the surfaces of concrete holding and feed lots. Also, the recycled water did not increase the counts of these same microorganisms on the teats of cows confined to these lots.

Nordstedt and Baldwin (44) found that sludge accumulated at an average rate of 1.4%/mo or 16.8%/yr of lagoon volume at an average loading rate of .115 kg of VS/m^3 (.007 lb

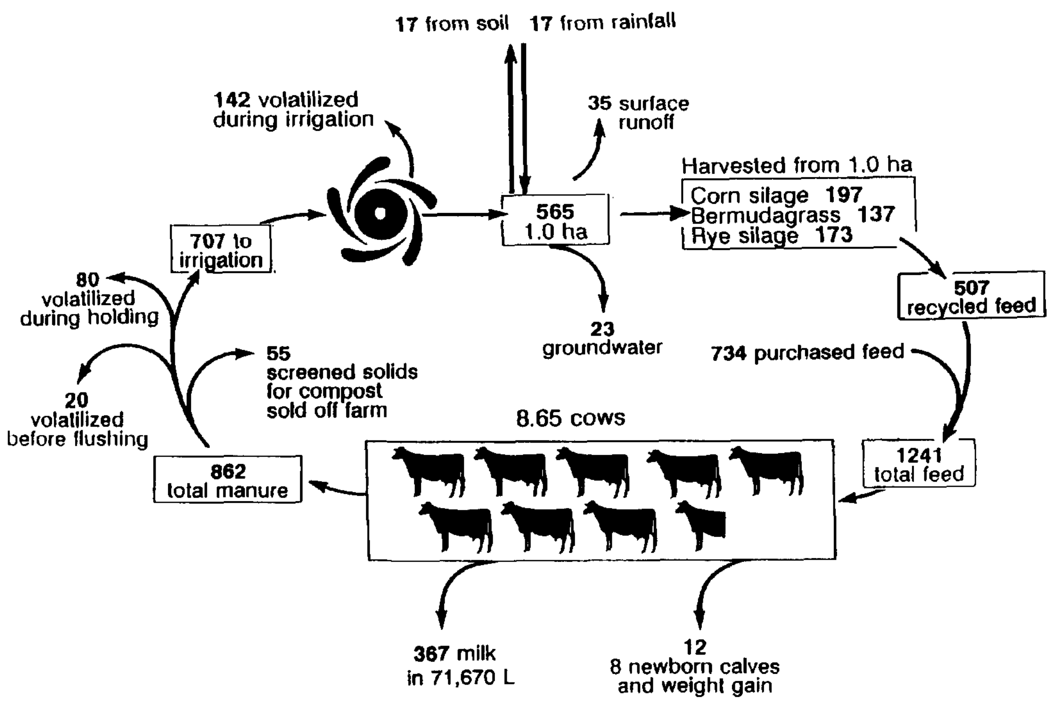

Figure 4. Example of N budget for dairy manure system where N was environmentally balanced [adapted from Van Horn et al. (62)]. Numbers represent kilograms of N. Crop N harvested per hectare was adapted from Johnson et al. (23). Losses of N from soil by denitrification were assumed to be offset by N returning in rainfall and losses of N to groundwater was the amount calculated to equal 10 ppm of NO_3^- N added to throughput groundwater. No commercial fertilizer was applied to crops.

of VS/ft^3) per d. The VS content of sludge averaged 67.9% of TS, much of the VS comprising microbial biomass. Sludge also contained some of the N and all of the P and other mineral nutrients lost from the influent. Thus, lagoon sludge serves as storage for fertilizer nutrients and periodically the sludge needs to be removed and hauled off the dairy farm or calculated into the total nutrient budget for the dairy. Sludge management alternatives include 1) batch removal, which would involve nearly complete sludge cleanout when the lagoon becomes full or begins to discharge high sludge effluent; 2) periodic sludge removal, particularly near the outlet structure, to reduce the amount of sludge in the effluent; and 3) continuous removal in lagoons that are destratified by mixing or recirculation (43).

Many dairy lagoon systems in current use are mismanaged (18). There was a tendency to dispose of trash, waste feed, and spilled silage or hay by flushing them into the lagoon. In half of the lagoon systems studied, such mismanagement resulted in excess floating plant residues and clogged drains. The crystalline build-up in anaerobic lagoon recycle plumbing and tanks is thought to consist mainly of magnesium ammonium phosphate ($MgNH_4PO_4 \cdot 6H_2O$), commonly referred to as struvite. No easy method was found to predict struvite problems. Any anaerobic lagoon system seemed to have the potential to form struvite, especially with swine and poultry manures. The recommended solution was to design the system so that lines and pumps could be cleaned with an acid solution (18).

Measured biogas production varied widely for several anaerobic lagoons, ranging from .2 to .5 m^3/m^2 per d (lagoon surface area) [0 to .23 m^3/m^3 per d (lagoon volume)]. The methane concentration of the lagoon biogas was about 60%. Biogas production was influenced by concentrations of organic acids in the lagoon, the position of the collection cover on the lagoon surface, and lagoon temperature. Biogas production from normally loaded lagoons (<.06 kg of VS/m^3) was not of sufficient quantity, nor was the production rate consistent enough throughout the year to be considered as a reliable source of energy (18).

Some research indicated that VS from dairy manures did not produce methane as efficiently as VS from other animal manures (17) and that anaerobic digestors for purposes of cogeneration were not cost effective on moderate-size dairies (8). However, recent research has resulted in the development of a new generation of anaerobic digestor designs based on biomas recycling or on biomass retention independent of waste flow (71). These designs have reduced reactor volume requirements and improved process stability and control, counteracting the early unreliability associated with anaerobic treatment. Major advances have also been made during the past decade in the understanding of the microbiology of anaerobic digestion. Greater appreciation of the importance of bacterial interactions is already providing more informed guidelines for anaerobic digestor operation and control (70).

Flushed manure wastewater is too dilute for conventional anaerobic digestion systems. The fixed-bed anaerobic reactor immobilizes bacteria on a matrix within the reactor, thereby preventing washout of microbial biomass. Hence, the fixed-bed reactor is capable of treating larger volumes of dilute wastewater per unit of time than conventional systems. This technology has been applied successfully at full scale to treat swine wastes but not to treat dilute dairy manures (A. C. Wilkie, 1993, personal communication).

Although previous research has been primarily concerned with the energy aspects of anaerobic digestion, a digestor functions as an integral part of the total waste management system, and its advantages and disadvantages should be reviewed considering the overall system. A primary advantage of an anaerobic digestor is its ability to stabilize raw manure almost completely. A digestor can be designed to produce a stable, relatively low odor effluent, although this design might not be optimal for economic methane production. Another advantage of anaerobic digestion is nearly complete retention in the effluent of the fertilizer nutrients N, P, and K that were in the raw manure entering the digestor. Nutrient losses may occur in subsequent handling of the effluent. This advantage may become more significant in the future if fertilizer shortages become more acute.

An anaerobic digestor has the ability to stabilize more waste per unit volume than other treatment facilities, such as lagoons. This advantage is offset in most cases by the fact that a lagoon will probably be required for storage of digestor effluent until it can be used in irrigation or otherwise distributed over the land. Thus, a digestor is not a complete disposal tool in itself. The liquid volume of waste to be handled is not appreciably reduced by the action of the digestor. The digestor does reduce the amount of solids to be handled and provides relative odor-free treatment. Digestor effluent is not suitable for direct discharge into streams because it is still relatively high in nutrients.

Composting

Aerobic composting is an old technology for stabilizing manures through controlled microbial action, primarily that of bacteria, actinomycetes, and fungi. However, composting is relatively costly and labor intensive, so dairies usually consider the process only if a marketable product is created that will remove excess nutrients from the farm. Aerobic composting converts biodegradable materials into stable end products. Thermophilic temperatures of 54°C (130°F) to 71°C (160°F), achieved in the process, kill most weed seeds and plant pathogens. Some VS reduction occurs, and some ammonia is volatilized, but mineral elements are retained. If partially anaerobic conditions develop, odorous compounds can be produced as a result of incomplete anaerobic fermentation. Anaerobic conditions can be created in the composting materials by excess moisture content, fine particle size, or compaction. The most commonly used method of composting is the windrow process, which involves stacking organic wastes into windrows that are turned periodically. Other, more mechanized methods, are also used (42).

The physical form of dairy manures often does not provide optimal composting conditions. Fresh manure is too wet, and screened solids are usually too low in N content and fertilizer nutrients. Thus, mixing materials from other sources may be helpful. Supplies of manure and bulking agents, as well as market demand for the finished compost, should be investigated before a dairy invests in composting equipment.

Water Cleanup

Municipal waste processing systems are built to treat water sufficiently to meet accepted water quality standards before discharge. These methods generally are not needed by production agriculture because economical biosystems are utilized for manure processing and utilization. However, many dairies exist in locations where lack of agricultural land and relationships with urban neighbors demand unique processing if the dairy is to continue operating in that location. Constructed wetlands and naturally occurring wetlands are being researched and utilized for water quality improvement in several regions (6). In some cases, wetlands have been effective for this use. However, unharvested wetlands accumulate nutrients that may have to be removed at some point if the nutrient content of effluents approaches levels demanding regulatory action.

An example is the Lake Okeechobee area in Florida, where one or more dairy farms were unable to reduce P concentrations in surface waters exiting the farm with normal nutrient management practices. Commercial systems have been installed that utilize chemical treatment in combination with biological systems to remove P from large volumes of runoff water from pastures and from anaerobic lagoon effluents. Some farms utilized a combination of precipitation of nutrients with $FeSO_4$ and constructed wetlands from which produced forages can be harvested periodically. Although removal of P by this system was relatively expensive compared with the value of P from other sources, such systems have reduced the P content of regulated waters by more than 90% in specific cases and, thus, have made it possible for dairies to continue operating at those sites (V. R. Hoge and O. P. Miller, 1993, Okeechobee County Extension, Okeechobee, FL, personal communication).

Effluents from lagoons and anaerobic digestors, and perhaps from some wetlands, contain sufficient nutrients to require further treatment if not used in irrigation for nutrient and water recycling. Such wastewaters can be used as a growth medium for microalgae, which could be harvested as a high protein by-product. Production of algal biomass (*Spirulina* spp.) that contained 60% CP (DM basis) effectively removed 73.6% of ammonia N in anaerobic lagoon effluent (7). Potential uses of harvested algal biomass include biogas generation, animal feed, and fertilizer.

In the future, as regulatory pressures intensify, many dairies in particularly sensitive locations will likely package all manure nutrients in a form that can be transported off the dairy and will process remaining water to allow complete reuse on the dairy.

OPTIMIZATION OF SYSTEM OPTIONS

Dairy farms operate under widely differing constraints, such as amount of cropland, types and number of crops per year (single, double, or triple), opportunity to irrigate, local hauling costs to alternative fields, and N versus P application restrictions. Computer software has been and is being developed that can help farmers to use manure nutrients to minimize fertilizer costs [e.g., (28)]. Lanyon (26) outlined a number of equations that might be considered in a linear programming optimization of decisions about nutrient loads on farms. The crop production solutions derived may be adjusted further by evaluating the resulting forage production programs selected using models for least cost or optimal profit ration formulation to obtain even better solutions to optimize crop production and feeding for the farm.

One major limitation on many large dairy farms to optimizing total manure nutrient use as fertilizer is the availability of enough crop production acreage so that nutrients can be applied at rates such that no nutrient will be applied in excess of crop requirements. In these cases, manure must be applied at the maximum acceptable rate for the most environmentally sensitive nutrient for that location, most often N, and the excess nutrients must be disposed of off the farm. Thus, choosing the optimal method to export nutrients off the farm is vitally important. Is this method to transport scraped manure to other farms, make compost from screened or sedimented solids, sequester nutrients in lagoon sludge for transport when the lagoon is cleaned periodically, or sequester nutrients via algal biomass? Other factors also affect the system choice, e.g., the need to reduce odors to avoid nuisance complaints, to reduce nutrient load in surface water exiting the farm, to avoid mud, and to minimize annual net costs of manure management. Figure 5 shows many of the components previously discussed and indicates combinations of the management options that could be selected.

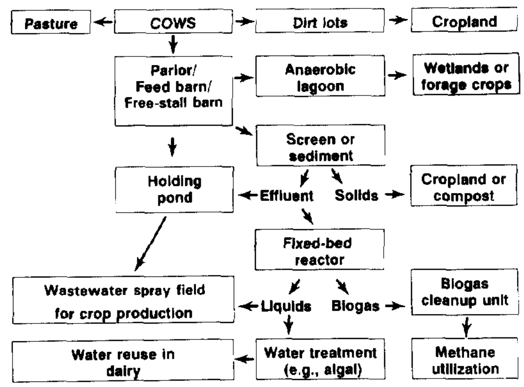

Figure 5. Several manure management system options showing potential for directing or redirecting flow of manure components depending on regulatory and economic constraints.

Currently, data have not been assembled to develop an extensive optimization model that would have general applicability. Often the constraints on a particular dairy are so specific, based on preexisting conditions, that decisions are based on one or two stages in the possible system. These conditions often deal with trying to retrofit the manure management system to accommodate previous or planned expansion.

CONCLUSIONS

Regulatory oversight of water quality standards has stimulated development of sustainable programs to manage total farm nutrients. For dairies with insufficient potential to utilize all of the manure nutrients through crop production, there is a great need to utilize technology that partitions fertilizer nutrients from manure organic matter and water so that surplus nutrients can be transported economically to other farms or to regions that are in deficit. Odors from manure have emerged as one of the primary public relations problems facing dairy farms if they are to coexist with urban neighbors. The energy resource potential of manure organic matter may be sufficient to stimulate some farms to employ fixed-bed anaerobic reactors for the combined benefits of energy recovery and odor control. Dairies that have difficulty in meeting water quality standards may find it necessary to invest in technology, such as algal ponds or constructed wetlands, to remove excesses of regulated nutrients and to improve quality sufficiently to permit reuse of all water and attain zero runoff of nonrain water.

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APPENDIX

Units

The following conversions are often needed: 1 gal of water = 8.346 lb = 3.785 L, 1 ft^3 = 7.48 gal = 28.31 L, 1 m^3 = 35.32 ft^3, 1 ha = 2.471 acres = 10,000 m^2, 1 acre = 43,560 ft^2, 1 acre-inch = 27,152 gal, 1000 m^3/ha = 3.938 acre-inches, BTU = heat to raise 1 lb of water by 1°F = .252 kcal, 1 kcal = 4.184 kJ, or 1 kJ = .239 kcal.