21 Recycling of Nitrogen Through Land Application of Agricultural, Food Processing, and Municipal Wastes[1]

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I. INTRODUCTION

Crop residues, animal, municipal, and more recently, food processing wastes are among the many materials applied to soil as fertilizer or for disposal. Before waste treatment systems were developed, most food processing and municipal wastes were discharged to rivers, lakes, and oceans, often severely polluting these waters and wasting fertilizer nutrients. Many innovations in waste management have been developed and considerable research is underway on the beneficial use of wastes for crop growth while giving cost-effective disposal.

The objective of this chapter is to summarize and evaluate available information on the value and management of agricultural, food processing, and municipal wastes as they are applied to land for N fertilization of growing crops.

II. AGRICULTURAL WASTES

A. Crop Residues

1. CROP RESIDUE DECOMPOSITION AND NITROGEN MINERALIZATION

Crop residues are a valuable natural resource. Approximately 363 million metric tons of crop residues are produced annually in the USA and

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Table 1—Annual crop residue yields and N contents in the USA (Larson et al., 1978).

Residue	Total yield	Nitrogen content	
	10^6 metric tons	%	10^6 metric tons
Corn (*Zea mays* L.) and Sorghum (*Syricum granum* L.)	154	1.1	1.69
Cereal grain‡	96	0.7	0.67
Soybeans (*Glycine max* L.)	53	2.2	1.17
Rice (*Oryza sativa* L.)	4.7	0.6	0.03

† Wheat (*Triticum aestivum*), barley (*Hordeum vulgare*), oats (*Avena sativa*), and rye (*Secale cereale*).

these residues contain approximately 4 million metric tons of N (Larson et al., 1978). Most crop residues are returned to the soil following harvest.

Yield and N content figures provide perspective on crop residue value and management problems associated with crop residues in the USA (Table 1). The N content of residues varies widely. For example, wheat straw N may range from 0.2 to 1.0% or higher.

Nitrogen transformations in decomposing crop residues in soil are reasonably well understood. Bartholomew (1965) prepared a comprehensive review of the subject and this section is intended to describe and evaluate subsequent research developments (See also Chapt. 6, S. L. Jansson and J. Persson).

When crop residues are incorporated into soil, the N requirements for a maximum decomposition rate depend upon many factors. The environmental factors were discussed in some detail by Bartholomew (1965) and will not be dealt with here. One point, however, that should be considered is the fact that environmental conditions are frequently less than optimum in the field, resulting in slower decomposition rates than those observed in the laboratory. Under these conditions, less N is required for crop residue decomposition in the field than in the laboratory. We shall emphasize field results.

After crop residues are incorporated into the soil the initial phases of decomposition proceed rapidly. In a field experiment with wheat straw, Smith and Douglas (1971) observed 20% straw weight loss in 10 weeks in the fall while soil temperatures decreased to 4°C. During 13 months 75% straw weight loss occurred. Jenkinson (1971) surveyed the literature on plant residue decomposition and reported that the proportion of crop residues decomposed under different climatic conditions with different plant materials was remarkably similar. Excluding acid soils, approximately one-third of the residue remained after 1 year and one-fifth remained after 5 years. He reported that even fresh green plant materials behave in this way, contrary to the widespread belief that such residues decompose rapidly and completely in the soil.

Previous cropping history and soil fertility have considerable impact on crop residue decomposition and N mineralization. The N requirements of subsequent crops also can be important. Smith and Douglas (1968) noted that while residual N from previous cropping had little influence on wheat

straw decomposition in the field, straw buried in plots increased in N percentage in relation to previous N fertilization. Further, yields of non-N responsive Pinto beans (*Phaseolus vulgaris* L.) were not influenced by straw applications, while sugar beet (*Beta vulgaris* L.) yields decreased due to addition of straw residues. Soil and crop analyses showed that N immobilization was essentially completed in the fall. The addition of 7.5 kg N/metric ton of straw, with N applied in the spring for growing the sugar beet crop, compensated for straw-immobilized soil N (Smith et al., 1973).

Studies in Iowa to determine the long-term effects of crop residue additions on soil organic matter and on N immobilization and mineralization, showed that after the addition of 16 tons/ha per year of cornstalk residues for 11 years, soil organic N increased by 32%. Alfalfa residue addition increased soil organic N by 41%. Calculated break-even point values for maintaining soil organic matter and N at the original levels indicated that 6 metric tons/ha per year of either alfalfa or corn residues were required (Larson et al., 1972).

Many laboratory experiments have been run in which researchers attempted to develop threshold values for N immobilization and release. Generally, crop residues containing 1.5% N or more will decompose at "normal" rates without bringing about a net immobilization of soil N. Crop residues containing $<1.5\%$ N are considered deficient in N and will lead to depletion of available forms of soil N during decomposition. Such residues are often amended with N to compensate for net immobilization and to hasten crop residue decomposition. In most cases, this is an uneconomical, wasteful practice based on a false premise. Nitrogen addition seldom accelerates plant residue decomposition (Allison, 1973). Nitrogen addition did not change the amount of straw remaining in soil after 3 years cropping under dryland conditions in Montana and Nebraska (Greb et al., 1974).

Nitrogen fertilizer use, soil fertility levels, and crop productivity have increased since many of the original laboratory experiments on N immobilization were performed. As a result, the impact of N on crop residue decomposition has changed. Field experiments by Brown and Dickey (1970) on nonirrigated soils in Montana showed that 95% of the buried wheat straw decomposed in 18 months and the residue increased from 0.3 to 1.2% N without added N. Similar results were found by Smith and Douglas (1971) in irrigated soils in Idaho, who found that 70% of the applied N (to bring the straw N to 1.5%) was lost in the first few weeks of incubation and that straw N was mineralized at N concentrations $<1.0\%$ (See Fig. 1). Under field conditions, with irrigation or rainfall, leaching as well as the numerous factors that limit decomposition rates apparently decrease the amount of N required for maximum decomposition. Evidently, decomposition proceeds at some rate that is not limited by N when N percentages in the crop residues are below the theoretical equilibrium value of 1.5%.

Several reports are available on the influence of above-ground crop residues on soil organic matter and N transformations, but few deal with roots. Powers (1968) reported that the effect of N content of bromegrass (*Bromus inermis* L.) on the rate of decomposition was similar to that of

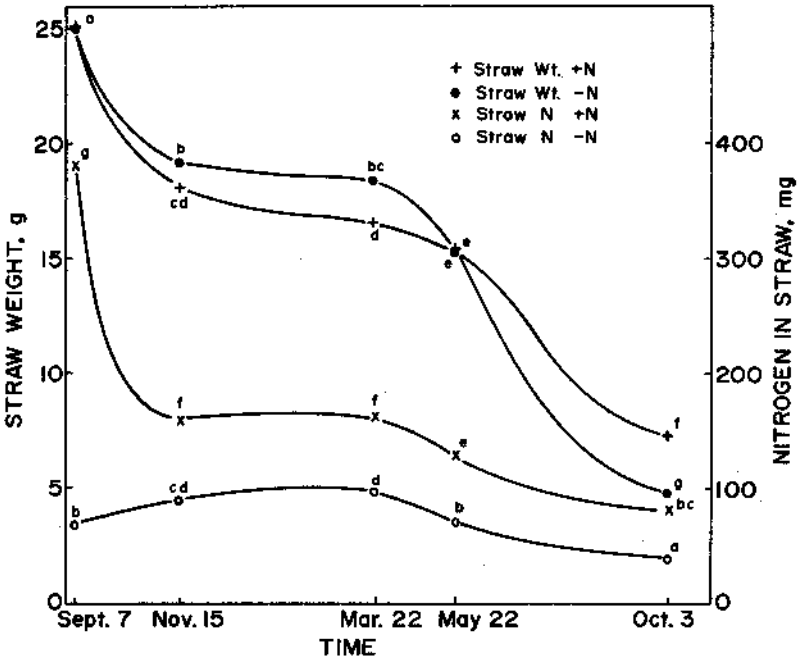

Fig. 1—Weight of Nugaines wheat straw remaining in soil at indicated dates, and N content of straw. Points associated with different letters are different at the 0.05 significance level (from Smith and Douglas, 1971).

most other crop residues. Roots containing 0.8% N immobilized some soil N for a time while roots containing 1.4% N caused little or no N immobilization. Smith (1966) using ^{14}C-labeled plant parts of corn, wheat, and soybeans (*Glycine max* L.), showed that roots decomposed more slowly than tops. He also reported that corn roots decomposing in soil decreased the decomposition of soil organic matter by almost 50%. This protective effect would also decrease the rate of soil N mineralization and, therefore, corn may be grown continuously when adequately fertilized.

With other low N plant materials such as wood chips, sawdust, or wood bark, the decomposition rate and N mineralization is limited by composition. Wood products contain less soluble carbohydrates and higher lignin concentration than most crop plants, and thus, have lower decomposition rates. Most of the wood products investigated by Allison (1965) contained an average of 0.1% N, and the decomposition rates of most of the 28 wood species investigated were not influenced by N addition. These wood materials would not be expected to mineralize enough N to supply growing crops for a long time after incorporation into soil and would only slowly decompose in the soil.

The importance of rice straw (*Oryza sativa* L.) as a crop residue and its impact on soil fertility worldwide have been largely overlooked until recently. In the past 10 years, several definitive publications pointing out the differences between rice straw and wheat straw management have appeared.

While rice culture in the USA produces a relatively insignificant 4.7 million metric tons of residue annually, the remainder of the world probably produces more rice residues than any other cultivated crop residue. In Asia, excluding mainland China, the annual production of rice straw is approximately 454 million metric tons (Tanaka, 1973). Rice straw contains an average of 0.5% N and a total of approximately 2.3 million metric tons of N. Rice straw is used for a variety of purposes including building materials and furnishings, paper, fuel, animal feed, culture media, and composts, but its major use is for incorporation in soil to provide plant nutrients. Rice culture included both lowland (flooded) and highland (nonflooded) rice. This distinction is made because there is a striking difference between straw decomposition and N requirements under these two conditions.

Williams et al. (1968) determined that under lowland field conditions, net immobilization of N, as determined by yield responses, occurred at 0.54% N in the original straw. Straw with higher N content increased yields on N deficient soils, while straw with lower N content decreased yields. They reported that under upland conditions, N requirements for rice straw decomposition were similar to that of wheat straw.

Rao and Mikkelsen (1976) in laboratory experiments found that rice straw immobilized about half as much N as wheat straw. Sain and Broadbent (1977) enclosed rice straw in nylon bags and buried the bags in the soil. Decomposition of the buried rice straw was influenced very little by N additions. Yoneyama and Yoshida (1977a, 1977b, 1977c) conducted a series of experiments using ^{15}N to trace N transformations during rice straw decomposition. They reported that the N percentage in the rice straw increased during the first 2 weeks because of N immobilization while further increases in percentage of N resulted from C loss. The mature rice plants grown on soils enriched with ^{15}N-labeled rice straw derived approximately 6% of their N from the straw. Sampling with stage of growth showed that the percentage of N in the plants derived from straw was relatively constant. These authors also observed that N mineralization and immobilization by plant residues appears to be simultaneous, beginning at an early stage of decomposition even when the N content of the residue is low and net mineralization is not detectable.

Thus, N addition to rice straw for decomposition under lowland conditions is neither necessary nor desirable and, as found by Williams et al. (1968), sometimes N may suppress rice straw decomposition or cause yield depression. Most rice straw contains adequate N for decomposition and will provide part of the N needed to grow succeeding rice crops.

Krantz et al. (1968) have summarized the practical considerations in handling crop residues in the field. Early crop residue incorporation into moist soil was regarded as a key factor in residue management. Further, fertilizing the growing crop at the time N is needed by the crop is much more efficient than fertilizing the crop residue. When N is added to crop residues, part of the N is immobilized, but has little effect on decomposition rates. Nitrogen immobilization by decomposing crop residues decreases N use efficiency because losses occur in each subsequent turnover. A portion of the immobilized N is remineralized during the immediate cropping year and be-

comes available to growing crops. The balance of the immobilized N mineralizes at about the same rate as the native soil N in subsequent years.

2. DENITRIFICATION

Four factors must be present simultaneously for denitrification to occur in the field (see Chapt. 8, M. K. Firestone). They are (i) anoxic conditions, (ii) an organic energy source, (iii) presence of denitrifying microorganisms, and (iv) presence of NO_3^-.

Crop residues decomposing in soil may promote conditions contributory to denitrification. This depends primarily on the nature of the crop residue. Most crop residues, when mixed with warm moist soil, will decompose rapidly enough to deplete O_2 and, thus, lower the redox potential in the immediate area. Soil water content and texture will probably determine whether or not the threshold conditions for denitrification will be reached. Crop residues low in N that are not fertilized with N will not yield NO_3^- during the stage of rapid decomposition and NO_3^- in the soil at the time of residue incorporation will probably be mostly immobilized. Therefore, denitrification is unlikely in the presence of low N crop residue, providing N fertilizer has not been added. Approximately 85% of the crop residues produced in the USA are low enough in N that denitrification of the residue N will be minimal. One exception is soybean straw, which contains an average of 2.2% N. Incorporating this residue in a warm, moist soil will cause rapid decomposition, and nitrification is likely. With irrigation or rainfall, the redox potentials will be lowered and some denitrification could occur. How frequently this will happen in the field is open to speculation. The same conditions would develop with plowdown of green manure crops, alfalfa, bean or pea straw, or other high N crop residues.

Crop residues decomposing in soil can either decrease or increase ground water pollution by NO_3^-. For example, Smith and Douglas (1971) showed that approximately 70% of the N added to straw in the field was leached out of the straw into the soil. Nitrogen not immobilized will be susceptible to leaching. If the field is irrigated or rainfall is greater than the soil water holding capacity, the applied N may leach below the root zone and ultimately reach the ground water causing pollution. The potential for pollution is similar to that of low rate fertilizer N as described by Schuman et al. (1975).

Low N crop residues in soil may also decrease pollution potential by acting as a sink and immobilizing excess fertilizer N or mineralized soil N, retaining it in a nonmobile form until later when it will become available to growing crops.

High N crop residues, returned to the soil when N is not needed by growing crops, may pose a pollution hazard. Soybean straw, edible bean straw, pea (*Pisum sativum* L.) residue, or alfalfa (*Medicago sativa* L.) when incorporated into soil will decompose rapidly and the N will be subject to nitrification. This NO_3^-, like any fertilizer N that is applied at the wrong time, can leach and pollute ground water (Burwell et al., 1976; Letey et al., 1977). Crop residues can and should be managed to minimize N pollution hazards.

B. Animal Manures

1. ANIMAL MANURE DECOMPOSITION AND NITROGEN MINERALIZATION

Domestic animals in the USA produce over 1 billion metric tons of fecal wastes each year. Liquid effluents amount to over 360 million metric tons. Used bedding, paunch manure from slaughtering facilities, and dead carcasses make the total annual animal waste production close to 2 billion metric tons (Wadleigh, 1968). For centuries, animal wastes have been used as fertilizers for crop production. With the advent of low-cost commercial fertilizers, the use of animal manures became less cost effective. Since 1973, energy shortages have increased commercial fertilizer costs and made manure use more economical.

Modern animal feeding practices, where thousands of animals are concentrated in small areas, have made manure handling difficult. Waste must be stored so that its application corresponds with favorable weather, crop, and land conditions. Nutrient loss, especially NH_3, can occur during storage and can cause some social and economic problems. However, with careful management, the benefits outweigh the problems, making animal manure a valuable resource (Elliott & Swanson, 1976).

Many pathways are followed by N during animal waste use and conversion (Fig. 2). The processes affecting these conversions are physical, chemical, and biological. Microorganisms play a major role in N conversions in decomposing manure in the feedlot and soil.

The N forms in fresh excreta are mainly urea, NH_3, organic, and from birds, uric acid, with virtually no NO_3^-. Manure composition will vary with feed ration composition and with the animal species. For example, the N content of fresh chicken manure is much higher than that of the other animals (Table 2). Even though these variabilities exist, many manure decomposition pathways are common to all types of manure (Fig. 2).

When manure and urine are deposited, their composition changes immediately. For instance, urine contains large quantities of urea, which is converted rapidly to CO_2 and NH_4^+ by the enzyme urease. The N in urine comprises 21% (hog), 35.5% (horse), 52% (cattle), and 63% (sheep) of the total N excreted (Elliott & McCalla, 1973).

Mathers et al. (1972) determined the N concentration in manure from 23 Texas high plain feedlots. The N percentage ranged from 1.16 to 1.96% and averaged 1.34%. The way manure is handled affects its chemical content. As decomposition proceeds, the mineral percentage in the manure increases, and the mineralized N is converted to NO_3^-. When excessive rates of manure are added to soil, (100 metric tons/ha or more) NH_3 tends to accumulate in the soil, temporarily slowing NO_3^- formation.

Mathers et al. (1972, 1975) and Mathers and Stewart (1970), in a series of experiments where varying rates of manure were applied to soil, found that manure applications up to 22 metric tons/ha supplied enough N for growing corn or sorghum without excess NO_3^- in the soil profile. Higher

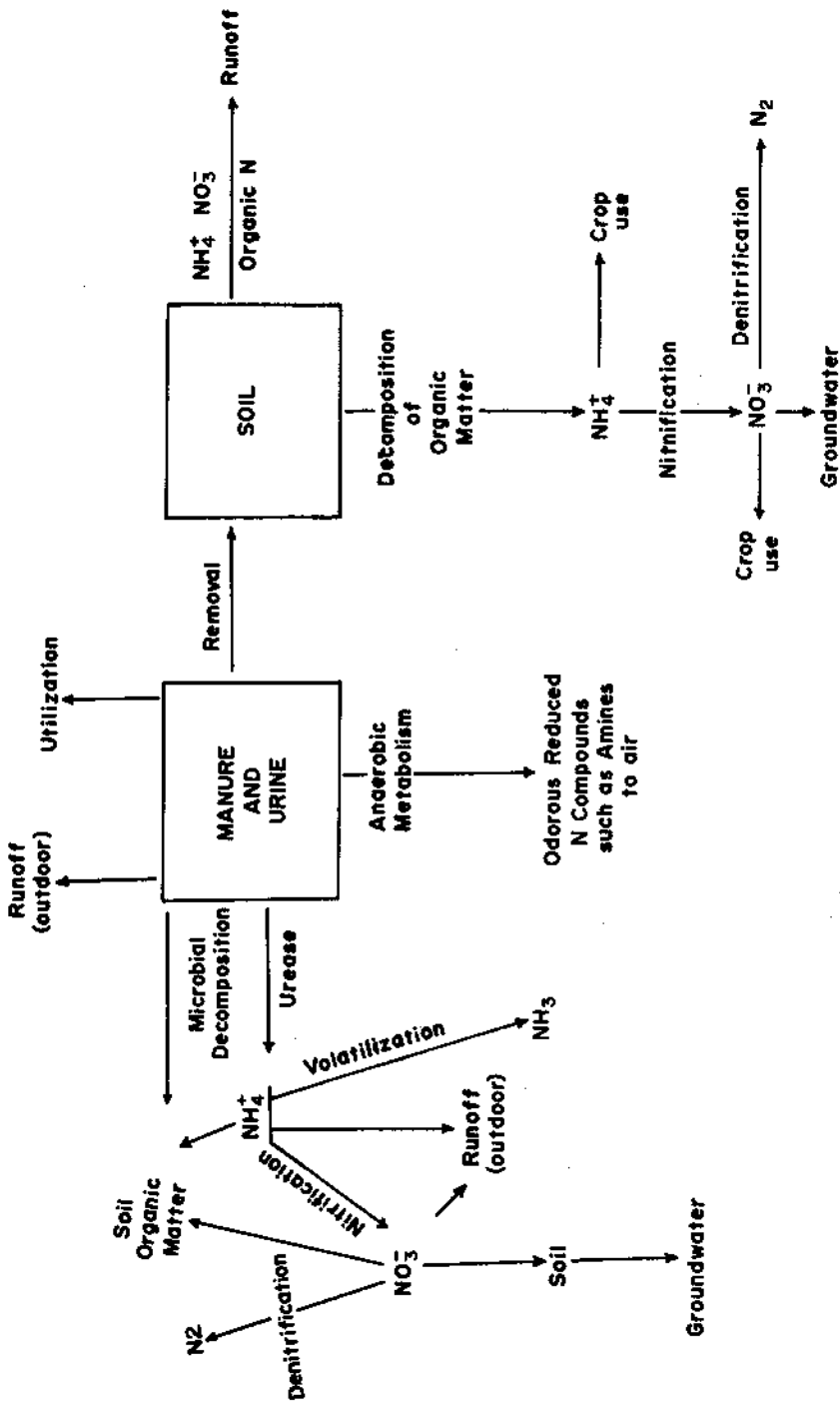

Fig. 2—Manure decomposition pathways (from Elliot and McCalla, 1973).

Table 2—Average moisture and N content of animal manures (from Elliott and McCalla, 1973).

	Moisture in fresh manure	N in fresh manure		Total dry solids		N/animal per day
	%		kg/ metric ton	kg/animal per day	%	g
Dairy cattle	79	1.0	10	3.7	4.6	172
Beef cattle	80	0.9	9	4.1	4.5	186
Swine	75	1.6	16	0.32	6.4	20
Horse	60	0.7	7	6.4	1.8	118
Sheep	65	1.7	17	0.5	4.6	21
Broiler	80	5.2	52	0.03	6.9	2.2
Hen	75–80	4.3	43	0.03	6.2	1.7

rates caused NO_3^- accumulation with up to 1,200 kg of NO_3^--N/ha in 180 cm of soil when 224 metric tons/ha manure were applied.

Nitrate pollution of surface or ground water is one of the major concerns in concentrated livestock holding areas or in soils receiving large amounts of manure (Gilmour et al., 1977). Mielke et al. (1974) described three layers in the soil profile under cattle feedlots. The surface layer is manure overlaying a mixed layer of organic and mineral material. Below this is the top of the mineral soil. The two upper layers are relatively impermeable when moist and allow little downward movement of NO_3^-. These authors reported higher NO_3^- concentrations to 1.7 m in one feedlot profile than in adjacent cropland, but little NO_3^- occurred in another location in the same feedlot. Some ammonification and nitrification occurs in feedlots. Adriano et al. (1971) found higher NO_3^- concentrations under California dairy corrals than under irrigated pastures or cropland at the 3- to 6-m depths. Marriott and Bartlett (1972) in Pennsylvania applied up to 85 metric tons/ha of manure and found relatively high NO_3^- concentrations in the soil to depths of 1.5 m when more than 17 metric tons of manure/ha were applied.

The literature on manure application to soils is voluminous. Pertinent recent references concerned with the probability of high soil NO_3^- concentrations accumulating with large applications of animal manure include: Webber and Lane (1969); Olsen et al. (1970); Concannon and Genetelli (1971); Hensler et al. (1971); Mathers and Stewart (1971); Peterson et al. (1971); Carlile (1972); Kimble et al. (1972); MacMillan et al. (1972, 1975); Murphy et al. (1972); Weeks et al. (1972); Bielby et al. (1973); Walter et al. (1974); Meek et al. (1974); Stewart and Chaney (1975); Loehr et al. (1976); and Pratt et al. (1976).

Although wastes have been used for centuries as nutrient sources for plant growth, a rational basis for their use has not been developed. Application rates have been based on experience and little information has been available about decomposition rates or nutrient availability. In recent years, renewed interest in using manure for its fertilizer value has led to several experiments to obtain information on decomposition rates and to develop a

rational basis for using manure. Pratt et al. (1973) proposed a theoretical basis for matching manure applications with crop needs. This theory is being tested by them and others to evaluate and refine estimates of N availability.

Organically combined N must be mineralized before it can become available to plants. Therefore, the mineralization rate is the key to the application rate of any given material. The yearly mineralization rates are expressed as a series of fractional mineralizations of any given application, or the residual of that application. These will be referred to hereafter as the decay series. For example, the decay series, 0.30, 0.10, 0.05, means that for any given application, 30% is mineralized the first year, 10% of the unmineralized residual is mineralized the second year, and 5% of the residual is mineralized the third and all subsequent years. The same series is applied individually to each yearly application of organic N (Pratt et al., 1973).

Constant annual manure applications that will supply enough N for the present crop will ultimately cause excessive fertilization. Therefore, to meet annual crop N requirements with manure applications, decreasing amounts should be applied each year. For example, based on 1.5% N, 30% moisture, and a 0.35, 0.15, 0.10, 0.05 decay series, to supply 200 kg of available N/ha each year, it would be necessary to add 54 metric tons the 1st year, and decrease the rate to <27 metric tons for the 20th year (Pratt et al., 1973).

Decomposition patterns of manures differ, requiring development of several decay series. The decay series 0.90, 0.10, 0.05 is typical of organic wastes such as chicken manure containing N, primarily in the rapidly mineralized urea and uric acid forms. The N in these materials is nearly as available as inorganic N. The decay series 0.75, 0.15, 0.10, 0.05 represents materials in which about 50% of the N is present as urea or uric acid and the balance is slowly mineralizable organic compounds. Fresh wastes from dairy or beef cattle are in this category. Other decay series are used to represent more stable organic materials such as accumulated dried dairy or beef cattle manure found in corrals (Pratt et al., 1973).

Table 3—Ratio† of yearly N input to annual N mineralization rate of organic wastes at constant yearly mineralization rate for five decay series for various times after initial application (Pratt et al., 1973).

		Time, years							
Decay series	Typical material‡	1	2	3	4	5	10	15	20
		N input/mineralization rate							
0.90, 0.10, 0.05	Chicken manure	1.11	1.10	1.09	1.09	1.08	1.06	1.05	1.04
0.75, 0.15, 0.10, 0.05	Fresh bovine waste, 3.5% N	1.33	1.27	1.23	1.22	1.20	1.15	1.11	1.06
0.40, 0.25, 0.06	Dry corral manure, 2.5% N	2.50	1.82	1.74	1.58	1.54	1.29	1.16	1.06
0.35, 0.15, 0.10, 0.05	Dry corral manure, 1.5% N	2.86	2.06	1.83	1.82	1.72	1.40	1.23	1.13
0.20, 0.10, 0.05	Dry corral manure, 1.0% N	5.00	3.00	2.90	2.44	2.17	1.38	1.13	1.04

† This ratio equals kg of N required to produce 1 kg of available N annually.
‡ The N content is on a dry weight basis.

The decay series can be used to obtain a constant N output each year or a particular N output in the current year by adjusting the manure applications. A constant annual N output can be obtained by using Table 3 which presents the ratio of yearly inputs to the annual N mineralization rate, at a constant yearly mineralization rate for five decay series for 20 years. In this case the application for any specific year can be obtained by multiplying the ratio for that year times the yearly mineralization rate desired. For example, if a constant yearly mineralization rate of 100 kg N/ha were desired, using a decay series of 0.40, 0.25, 0.06, the input rates would be 250, 182, 154, and 109 kg/ha of manure N each year, respectively, for the 1st, 2nd, 5th, and 20th years. Nitrogen input rates for years not listed will be between those listed. Using these ratios, the required amounts of any of the five manures can be calculated if the N and water contents are known (Pratt et al., 1973).

The decay series used in Table 3 were largely the results of the combined judgments of Pratt, Broadbent, and Martin, except the decay series 0.40, 0.25, 0.06, which was taken from a field trial in the Coachella Valley, in which the availability of the N was compared with the availability of inorganic sources. These series should be tested further for verification or modification.

Nitrogen losses in manure handling are an important consideration when determining the fertility value of the manure. Turner (1976) evaluated N loss in various storage, treatment, and handling systems (Table 4). Nitrogen losses in storage and handling ranged from 10 to 84%. Management systems can be used that will increase N losses when large quantities of manure are available to spread on limited land areas or can be minimized in cases where efficient N use for growing crops is desired.

Turner (1976) developed a less complicated approach to manure N availability than the decay series previously discussed. His availability coefficient A indicates that portion of the total soil-incorporated manure N that is, or becomes available. The value of A is influenced by the portion of the total N that is already in the inorganic form when the manure is incorporated, the amount of organic N in the current-year application that is miner-

Table 4—Nitrogen remaining after accounting for storage, treatment and handling losses (Turner, 1976).

Manure storage, treatment, and handling system	N remaining, %
Oxidation ditch, anaerobic lagoon, irrigation or liquid spreading	16
Anaerobic lagoon, irrigation or liquid spreading	22
Deep pit storage, liquid spreading	34
Aerobic lagoon, irrigation or liquid spreading	40
Open lot surface storage, solid spreading	40
Roofed storage in manure pack, solid spreading	65
Fresh manure, directly field spread:	
(time between application and incorporation)	
1-4 days; warm, dry soil	65
7 days or more; warm dry soil	50
1-4 days; warm, wet soil	85
7 days or more; warm, wet soil	70
7 days or more; cool, wet soil	90

Table 5—Availability coefficient A for the first 5 application years (Turner, 1976).

Type of manure	Availability coefficient for the year of application				
	1	2	3	4	5
Poultry, fresh	0.75	0.80	0.85	0.90	0.93
Poultry, aged, covered	0.60	0.75	0.80	0.84	0.87
Dairy, fresh	0.50	0.65	0.70	0.74	0.78
Dairy, liquid manure tank (1–3 weeks storage)	0.42	0.54	0.60	0.64	0.68
Dairy, anaerobic lagoon storage	0.30	0.38	0.45	0.50	0.53
Beef, feedlot stockpiled	0.35	0.45	0.50	0.53	0.55

alized during the year, and the amount of residual organic N from previous applications that is mineralized during the current year. Estimated values for A (Table 5) are based on the assumption that manure will be applied each year on the same land for an indefinite number of years. The availability coefficient, therefore, approaches 1.0 with time.

In the case of poultry waste and fresh bovine waste, the initial year recovery rates proposed by Turner (1976) appear to be lower than those of Pratt et al. (1973). This may be partly explained by the climatic differences between the Pacific Northwest and California. This also points to the necessity of obtaining data from other areas to enable more precise evaluation of manure fertilization requirements in areas of widely differing climates.

Other approaches have been made to estimate the amount of available N from manure. Walter et al. (1974) prepared a computer model for predicting the soil profile NO_3^- distribution resulting from heavy application rates of anaerobic liquid dairy waste to coarse-textured soil. Gilmour et al. (1977) introduced a half-life concept of manure decomposition. The half-life is the time required for 50% of the manure incorporated into the soil to decompose. The half-life increases with stage of decomposition and time as the more resistant materials remain in the soil. They described this relationship with the equation:

$$t_{1/2} = \frac{0.693}{k \times (M_2/M_1) \times 0.933^{-[(\text{Annual heat units}/365) - (T_2 - T_1)]}}$$

where

$t_{1/2}$ = manure decomposition half-life in days;
k = reaction velocity constant at optimum temperature (27°C);
M_1 = soil moisture for optimum mineralization (% of water-holding capacity, WHC);
M_2 = actual soil moisture (% of WHC);
T_2 = experimental reference temperature, °C; and
T_1 = 5°C = temperature at which annual heat units = 0.

The factor 0.933 applies when the velocity of the reaction is decreased 50% for each 10°C temperature decrease.

This mathematical model was developed to predict C loss during organic matter decomposition in soil, but because C loss and N mineralization are closely related in waste materials with narrow C/N ratio, as in animal

manures, it should also be useful for predicting N mineralization. Additional testing in the field will determine how useful mathematical models may be for predicting N availability from manures.

Meek et al. (1975) prepared guidelines for manure use and disposal in the western USA. The first objective of manure application should be designed to minimize pollution and maximize plant nutrient conservation.

2. DENITRIFICATION

Fresh manure contains very little NO_3^--N, therefore, the potential for N loss through denitrification is nearly zero initially. However, soon after manure is incorporated into the soil, decomposition begins, and with large applications of manure, the NO_3^- concentration in the soil sometimes increases to high levels. Denitrification of NO_3^- from animal manures is affected by the same factors that were discussed earlier in this chapter and in Chapter 8 (M. K. Firestone).

Larson and Gilley (1976) stated that considerable N will be lost by denitrification in moist soils after application of manures, sewage sludges, and sewage effluents. Rolston and Broadbent (1977) found that denitrification was greater in manured plots than in similar nonmanured cropped field plots, probably because of an abundant energy source. Carlile (1972) studied the transformation and movement of N from slurry manure applied to soil. He developed a system for denitrifying soil solutions containing NO_3^- by adding energy materials. Two such materials that worked successfully were methanol and milk, either of which removed up to 90% of the NO_3^- in 1 hour retention time in his anaerobic filter. There is an interest in denitrifying excess NO_3^- under field conditions without the use of additives.

Guenzi et al. (1978) conducted greenhouse investigations in which they studied nitrification and denitrification with 45- and 90-metric ton/ha manure rates and with ^{15}N-labeled N sources. They determined that oxygen levels in the soil were decreased, but not completely depleted by manure decomposition. However, anaerobic microsites were developed which permitted denitrification to occur. They showed that nitrification and denitrification were proceeding simultaneously. In field experiments, Concannon and Genetelli (1971) showed that soluble organic matter from manure applied to soil was leached 90 cm or deeper. This organic matter provided energy for denitrification. Murphy et al. (1972) reported greater NO_3^- concentrations in fields that had received one manure application than after two annual treatments. They attributed this to leaching of organic materials to greater depths and in greater concentrations with additional manure applications, resulting in enhanced denitrification.

Kimble et al. (1972) attempted to estimate denitrification by measuring leached NO_3^- and the NO_3^-/Cl ratios in plots receiving 66 metric tons/ha of manure. The ratio decreased with depth to 182 cm, indicating that denitrification had occurred. Meek et al. (1974) showed that with 180 metric tons/ha manure, soluble organic C leached to 80 cm or deeper. Nitrate was leached to considerable depths in the soil and denitrified, and the disappearance of NO_3^- correlated with the appearance of soluble manganese. They

Table 6—Denitrification coefficient for four drainage conditions (Turner, 1976).

Degree of soil drainage	Denitrification coefficient
Excessive or somewhat excessive	--
Well to moderately drained	0.85
Somewhat poorly to poorly drained	0.70
Very poorly drained	0.60

also measured redox potentials and correlated them with NO_3^- loss by denitrification.

This research has provided background for greater quantification of nitrification and denitrification of manure N in soil. Further, Turner (1975, 1976) developed a denitrification coefficient. This coefficient indicates that portion of the available inorganic N remaining in the soil after accounting for denitrification loss. Aeration, one of the main factors in denitrification, is related to soil drainage. Therefore, excessively drained soils show little or no denitrification potential and this potential increases as soil drainage decreases to the very poorly drained soils (Table 6).

Nitrogen loss from denitrification following manure applications to land will probably be associated primarily with systems where manure is applied at high rates. In summarizing available data, manure application rates below 20 metric tons/ha will probably produce fairly low NO_3^- concentrations in soil and low concentrations of leached organic material below the manure mixing layer. Therefore, denitrification losses would be expected to be less than at high disposal rates (100 or more metric tons/ha) where high concentrations of leached organic materials, high NO_3^- concentrations, and low redox potentials resulting from intensive microbiological activity would maximize denitrification. This is a favorable factor in decreasing ground water and soil pollution from NO_3^- in manure disposal sites and under corrals.

3. NITROGEN POLLUTION POTENTIAL FROM ANIMAL MANURES

During decomposition of manure, NO_3^- is the main end product from the nitrogenous portions of the manure, and NO_3^- may accumulate to very high concentrations in the soil. Several unfavorable conditions may develop in soil and ground water as a result of high NO_3^- concentrations. High NO_3^- in the soil may result in high to very high NO_3^- concentrations in forage, feed, or food. High NO_3^- concentrations can be toxic to animals or humans consuming the products grown on land. Nitrate concentrations in the ground water may increase above the limits set by the U.S. Public Health Service (see Chapt. 16, D. R. Keeney). The movement of NO_3^- through the soil profile varies with soil, climate, and management (Chapt. 11, D. R. Nielsen and J. W. Biggar). Generally, NO_3^- will move through porous, well-drained soils with water from either rainfall or irrigation. Although the rate of movement varies, Adriano et al. (1971) suggested that with 18 to 25 cm of drainage water per year, NO_3^- may reach a water table at a 30-m depth in 10–50 years. Nitrate does not readily move through tight soils, uncleaned animal yards that have become sealed, or in low rainfall nonirrigated areas.

Stagnant water or poorly aerated soil layers encourage denitrification and much of the NO_3^- is lost rather than moved into the ground water (Meek et al., 1975).

We will discuss a few animal waste disposal related experiments that point out the problems associated with NO_3^- leaching. Bielby et al. (1973) treated lysimeters with slurry manure to provide 560, 1,120, and 1,685 kg N/ha, and found that in 2 years of cropping to corn, 26.6, 26.2, and 24.0% of the added N leached out of the lysimeters. Concannon and Genetelli (1971) applied manure to field plots at rates of 0, 33, 66, and 99 metric tons/ha and found that the 66 and 99 metric ton rates produced NO_3^--N concentrations above 10 mg/liter in the soil solution for August or October applications, but not for June applications. Marriott and Bartlett (1972) found that manure applications containing 250–1,460 kg N/ha produced NO_3^--N concentrations in the soil solution ranging from 3 to 140 mg/liter.

In evaluating soil and water pollution potential from manure applications, the greatest need is to determine leaching losses for various manure rates for specific cropping sequences, water management systems, soils, and climates. When soils are loaded with large amounts of manure, denitrification is the key mechanism in preventing or decreasing NO_3^- leaching beyond the root zone. Pratt et al. (1976) indicated that organic loading was more important than water management in creating O_2-deficient systems necessary for denitrification. They also reported that the NO_3^- concentration in the unsaturated zone and the amount of NO_3^- leached were linearly related to available N in the soil and also to the amount of dry or liquid manure added. High organic loading favors denitrification and adds a buffering effect to NO_3^- leaching from high manure applications. In most well-aerated soils, this buffering probably is not sufficient to prevent large leaching losses and high NO_3^- concentrations in drainage waters from large manure applications. Conversely, in heavy, poorly drained soils, there is evidence that denitrification will greatly decrease or eliminate NO_3^- leaching losses when large amounts of animal manure are applied. When annual applications cease, the lower soil profile becomes aerated, denitrification stops, and NO_3^- leaches from the profile until the added organic source is depleted. In a humid area with 125 cm of annual precipitation, Fordham and Giddens (1974) found that feedlots remained wet on the surface most of the time, which was conducive to denitrification, and they found only 3 mg/liter NO_3^--N at the 120-cm depth under the feedlot.

Of nuisances related to manures, odor is probably the most readily noticeable, but the least definable and the most difficult to control. Although manure odors can be characterized chemically, odor nuisance judgements are subjective. The odor of fresh livestock manure is inoffensive to most people, but odors produced by anaerobic bacterial activity during fermentation of wet manure can be offensive (McCalla et al., 1977).

Ammonia from manures is usually not as offensive as anaerobically produced volatile non-N compounds. Stewart (1970) in soil column studies showed that 25% of the applied urine-N was lost as NH_3 from wet and 90% from dry soil surfaces. Peters and Reddell (1976) reported a 10–20% loss of total N as volatilized NH_3 from heavy manure applications on soil columns.

Adriano et al. (1974a) found that manure application rate did not affect NH_3 loss, but showed volatile losses approaching 50% of manure N. McCalla et al. (1970) in a laboratory experiment showed up to 90% of surface-applied urine N was lost by volatilization of NH_3.

Hutchinson and Viets (1969) measured NH_3 concentrations in the air downwind from a 90,000 cattle feedlot. They postulated potential water pollution of nearby lakes with NH_3. Luebs et al. (1973) measured NH_3 concentrations in the air in a high concentration dairy area of California and calculated that a lake in that area could absorb up to 190 kg N/ha of lake surface annually. In a 6-m-deep lake this would constitute enough N to make an NH_3 concentration of 0.75 mg/liter. This is in excess of the U.S. Environmental Protection Agency (USEPA, 1973) recommended permissable NH_3 level for public water systems of 0.5 mg/liter. Rainfall in these areas also contributes additional NH_3-N to soil and water and increases the pollution potential from high concentrations of livestock. These observations suggest that large livestock enterprises, with their associated pollution problems from odors and NH_3, should not be located in highly populated areas.

III. FOOD PROCESSING WASTES

A. The Nature and Composition of Food Processing Wastes

Food processing wastes may be liquid as in the cases of effluents from processing potatoes, sugar beets, vegetables, fruits, meats, or dairy products or they may be solid or semisolid materials such as soil, rocks, peel wastes, pulps, or paunch manure. They vary in composition from very low nutrient liquid wastes from dairy washing operations to very high nutrient concentrations from whey or meat processing. Almost all food processing waste effluents may be used for irrigating agricultural land to supply water and nutrients for growing crops. Waste water containing a high salt or Na concentration that would damage crops and soil cannot be used for irrigation.

The amount of vegetables, fruits, and meats processed in the USA annually was estimated by Hunt et al. (1976), and the processed product, the waste water produced, and the resulting biochemical oxygen demand (BOD) load are shown in Table 7. The total N added by irrigation with these waste

Table 7—Wastes from processed vegetables, fruits, and animal slaughtering (Hunt et al., 1976).

Product	Amount processed	Waste water	BOD
	10^6 metric tons	$10^6/m^3$	10^6/kg
Vegetables	12.8	185	232
Fruits	10.9	129	131
Meat	32.5	327	557
Total	56.2	641	920

waters is significant, ranging from 6,500 metric tons N/year for vegetable processing wastes, somewhat less from fruit processing, and 2,400 metric tons annually from meat-processing wastes.

1. VEGETABLE PROCESSING WASTES

Large volumes of water are used for washing, transporting, blanching, and cooking vegetables. Each of these processes extracts some vegetable constituents, enriching the water with organic material and plant nutrients. As water is used and recycled to lower level steps in the vegetable processing, its nutrient and organic concentration increases and its quality decreases until the water can no longer be used for processing. Then it is discarded. Table 8, compiled from several publications on waste water quality, shows NO_3^-, total N, and chemical oxygen demand (COD) concentrations in the waste waters from processing several vegetables. Nitrate-N concentrations are generally low (<3 mg/liter) in vegetable-processing waste waters. However, the organic N in the waste water will be converted to NO_3^--N during treatment or in the soil when it is used for irrigation. When using vegetable processing waste water for irrigation, the organic N must be considered as the main N component and evaluated as fertilizer for crops and for its soil and ground-water pollution potential.

The total N content of the vegetable-processing waste waters reported in Table 8 range from 6 to 66 mg/liter with tomato (*Lycopersicon esculentum* Mill.) and brussels-sprouts (*Brassica oleracea* Gemmifera) processing producing low N waste water and potato (*Solanum tuberosum* L.), corn,

Table 8—Nitrogen and COD composition of vegetable processing wastes.

Crop	NO_3^--N	Total N	COD
	In wastewater		
	mg/liter		
Snap beans†	0.9	31.2	176
Sweet corn†	1.7	61.9	1,043
Brussels-sprouts†	0.4	5.7	15
Beets†	2.9	66.4	854
Peas†	0.1	44.7	707
Tomatoes†	0.6	6.3	95
Cabbage†	1.6	31.3	229
Tomatoes‡	0.4	6.8	47
Corn‡	trace	27.3	316
Potatoes§	0.9	55.0	1,680
	In solid waste		
	% N		
Tomato¶	–	2.33	–

† Shannon et al., 1968.
‡ Stanley Assoc., 1977; values are biochemical oxygen demand, BOD.
§ Smith et al., 1977.
¶ Timm, H., N. B. Akesson, M. O'Brien, W. J. Flocker, and G. York. 1976. Soil and crop response to variable loading of canning wastes. Presented at Am. Soc. of Agric. Eng. Meeting, Davis, Calif., June 1976.

cabbage (*Brassica oleracea* Capitata), and beet (*Beta vulgaris* L.) processing relatively high N waste waters. The COD concentration of the waste water is also included because in some waste materials, the COD and N concentrations have some interesting relationships. For example, potato-processing waste waters have a COD/N ratio in the range of 25:1 to 30:1 representing from 3.3 to 4.0% N in the organic waste materials. Other vegetable processing wastes may have predictable COD/N ratios, but these data are not available.

Applying food processing waste water to land can supply large amounts of N. Smith et al. (1975, 1978) and Smith (1976) showed that potato processors applied from 160 to 490 cm of waste water annually, which supplied from 1,080 to 2,200 kg N/ha. Potatoes are processed most of the year and large amounts of waste water are discharged from the processing plants. Daily discharge ranges from 1.9 to 19 million liters. This long processing season often results in excessive N applications to the land used for waste water discharge. Other vegetables such as peas, green beans, sweet corn, tomatoes, and brussels-sprouts are processed for a much shorter season each year. Waste water from these processing operations is discharged to the land for only a few months, therefore, their fertilizer potential is much less than that from potato processing. The actual fertilizer N obtained from processing these other vegetables is not known, but can be calculated from the general data in Table 8 if the amount of water available from their processing is known.

Another factor related to N fertilization with waste water from vegetable processing is the efficiency of conversion from the organic to available N. De Haan et al. (1973) stated that waste water applications must be adjusted so that plant nutrient as well as purification requirements are met. To meet plant nutrient requirements, the availability of plant nutrients in the waste water must be known. They developed a "relative efficiency index" for N and other nutrients utilizing potato starch waste. The "relative efficiency index" of potato starch waste N compared to commercial fertilizer N was as follows: potatoes, 0.5; beets, 0.5; cereals, 0.2; and grass, 0.8. These values need further verification in other climatic areas and with other soils, waste waters, and crops than those in The Netherlands, but the concept is good.

Some vegetable processing operations generate a large quantity of solid or semisolid waste. In the potato processing industry, several kinds of solid waste are generated. Waste water from the processing plants is usually passed through a primary clarifier, where the settleable solids are removed from the clarifier underflow and concentrated on a vacuum filter. These wastes contain from 6 to 15% dry matter with a N concentration similar to that of the soluble COD that passes through the filter. The filter cake is usually fed to livestock as a substantial part of the fattening ration. Some potato processors do not use a clarifier, but apply the solid wastes directly to the fields. In this case, organic C and N loading is increased by the additional solid materials resulting in a higher rate of N addition than with clarified waste effluent alone. Additional land should be used to assimilate the additional nutrients. Other solid wastes include substandard products

that must be discarded, soil, rock, and mud from transporting and washing the potatoes. These waste materials are fed to livestock or are discarded on land or in landfills as appropriate. Other solid wastes encountered in vegetable processing include peels from tomatoes and other vegetables, and pomace or pulp from tomato juice processing, or other similar materials. These are generally disposed of on land and contribute substantially to N fertilization because they contain relatively high N concentrations. For example, Timm et al. (see footnote Table 8) reported that tomato wastes contain 2.3% N on a dry weight basis.

2. SUGAR BEET PROCESSING LIQUID WASTES

Sugar beet processing waste water is being evaluated for irrigation use by Smith and others in Idaho (Smith & Hayden, 1980). Preliminary data are presented in Table 9. In Idaho, sugar beets are processed mostly during the winter season. Organic materials in the waste water pass through the soil more readily than do potato processing wastes and the N also readily leaches through the soil. Additional research is underway to improve cleanup and utilization of the sugar beet waste waters. The concentration of waste constituents in the waste water depends in part on the type of processing operation; however, the N rates in all these applications are high, but when divided by two or three seasons of fertilization they do not give excessively high N fertilization rates.

3. FRUIT PROCESSING WASTES

Fruit processing waste water data are rather scarce, but those few available data indicate that except for grape wastes, these wastes are usually low in N (Table 10).

Solid wastes from the fruit processing industry consist of fruit peels, pits, pomace, seeds, and stems. These wastes as applied to land for disposal are high in water. The N concentrations range from 0.77 to 1.37% and when applied to land, will supply from 1.2 to 1.8 kg N/metric ton of the wet

Table 9—Sugar beet processing plant waste water applied October 1975 to December 1977 (Smith et al., 1977; and unpublished data).

Processing	Waste water applied	COD applied		Nitrogen applied	
	cm	mg/liter	kg/ha	mg/liter	kg/ha
		Twin Falls, Idaho			
Sugar beets	142	3,580	42,930	104	1,330
Juice	--	1,990	--	53	--
		Rupert, Idaho			
Sugar beets	111	1,616	20,830	65	717
		Nampa, Idaho			
Sugar beets	170	1,758	24,238	55	719
Juice	--	660	--	14	--

Table 10—Liquid fruit processing waste N and COD in wastewater.

Crop	NO_3^--N	Total N	COD
	mg/liter		
Apple products†	trace	2.2	170
Pear (*Pyrus communis* L.)†	trace	2.6	1,230
Grapes (*Vitus* sp.)‡	2.0	49.9	909
Citrus (*Citrus* sp.)§	3.0	7.8	150

† Stanley Assoc., 1977; values are BOD.
‡ Shannon et al., 1968.
§ Koo, 1974.

Table 11—"Solid" wastes from fruit processing (Reed et al., 1973).

Crop	Water	Total N
	%	
Peach-pear	86.7	0.77
Peach [*Prunus persica* (L.) Batsch]	85.1	0.75
Pear	87.2	1.37
Mixed fruit†	88.9	0.85

† Mixed fruit was peach, pear, plum, grape, and cherry.

fruit processing waste material (Table 11). From 50 to 200 metric tons/ha of the wet waste material can be applied to meet the fertilization needs of crops grown on these lands.

4. MEAT AND DAIRY PROCESSING WASTES

Meat processing liquid wastes are generally low in NO_3^--N, but they may increase in NO_3^- during conventional waste water treatment (Table 12). The N and COD concentrations in meat packaging waste water are relatively high and probably inversely related to water use efficiency in processing plants. The N and COD concentrations in waste water are highest for slaughter wastes and lowest for meat cutting wastes (Table 12). Meat processing waste water is generally easily treated in conventional waste treatment systems and is well suited for use on irrigated cropped land.

Dairy manufacturing wastes are extremely variable in composition (Table 12). Whey has the highest NO_3^-, total N, and COD concentrations. While whey is perhaps best suited for use as a livestock feed rather than land application, considerable whey is disposed of by irrigation. The high N and COD concentration pose some special problems when irrigating with whey. Extremely high N fertilization can result. The high COD causes rapid microbial growth and can produce odor problems and anaerobic conditions in and on the soil which damage growing crops. Dilution of whey with water is usually necessary for irrigation to decrease the severity of the problems in the field.

Other dairy manufacturing waste waters contain much less N and COD than whey and should be well suited for irrigation with no greater problems than balancing the N content of the waste water with the cropping require-

Table 12—Meat and dairy processing waste N and COD concentrations.

Source	NO_3^--N	Total N	COD
	mg/liter		
In meat packing wastes			
Catch basin effluent†	--	124	4,180
Extended aeration influent‡	0.4	92	1,630
Extended aeration effluent‡	2.6	10	122
Slaughter waste, nontreated§	0	69	2,029
Slaughter waste, treated§	0.4	10	139
Custom meat cutting§	0	20	139
In dairy manufacturing wastes			
Whey§	7.8	685	53,225
Fresh milk packaging§	0	24	2,290
Curds (waste not whey)§	0	18	725
Condensed milk§	2.5	1	96
Ice cream§	0	1	21

† Tarquin et al., 1974.
‡ Witherow, 1975.
§ J. H. Smith, Kimberly, Idaho, USDA-ARS, unpublished data from 24 meat and dairy processing plants in Idaho.

ments. Nitrogen efficiency factors for meat and dairy processing waste water N are not yet available.

Paunch manure is probably the major waste from livestock slaughtering that would be used for fertilizer. According to Baumann (1971) paunch manure contains approximately 2% total N on a dry weight basis. This material can be applied to fields and would be expected to produce an almost immediate fertilizer benefit. Values for availability of paunch manure N should be similar to those of fresh livestock manure discussed earlier. There are no solid wastes associated with dairy manufacturing that would normally be used for fertilizer.

B. Irrigating Agricultural Land

Application systems for food processing waste water fall into three different categories:

1) *Irrigation*—Irrigating agricultural land to produce crops is the system most often used and has the advantage of conserving and using at least part of the nutrients contained in the waste water. The water may be applied by surface methods or by sprinkling. Surface application lessens NH_3 volatilization compared to sprinkling and can be used on some soils even in midwinter, as the heat in the water thaws the soils and maintains infiltration. Sprinkling increases volatile NH_3 losses and creates some aerosol problems without utilizing the temperature advantages of surface application.
2) *High-rate Infiltration*—When large volumes of waste water are applied to sandy or gravelly soil where infiltration and percolation

rates are very high, many nutrients are lost by percolation and plant nutrients are used inefficiently.

3) *Overland Flow*—This is a method of water application where soils are relatively impermeable and the water is purified to some extent by contact with the growing vegetation, the organic matter laying on the surface, and by limited contact with the soil. This method has been studied by Gilde et al. (1971).

The most desirable waste water application method will be determined by several factors that must be evaluated at each site when the systems are designed. The system used will influence nutrient recovery and the utilization of N. Two concepts of waste handling in relation to the nutrient content are the disposal concept (the main consideration in infiltration-percolation systems), and the recycling–reuse concept. The recycling–reuse concept is becoming much more important with developing energy shortages and increased fertilizer cost. Many food processing waste treatment and disposal systems were designed to utilize the maximum rate of water and nutrients. Frequently, there was no other consideration than that a crop could be grown to keep the area looking acceptable and to avoid waste water applications that would create nuisance situations such as ponding and objectionable odor. High land cost and proximity, and the necessity for landleveling and water retention or recycling on the disposal and treatment site have led to the concept of applying as much waste water as possible. In many cases, N loading is extremely excessive and pollution of soil and ground water and the production of high NO_3^- forage results.

C. Nitrogen Loading and Utilization on Land

When food processing waste water is applied to agricultural land for treatment and disposal, the first limiting factor in the system design is usually the N application rate. Reasonable predictions of acceptable loading rates can be made for a given site if soil conditions, type of crop to be grown, depth to water table, frequency and intensity of rainfall, and similar pertinent information is available (Loehr, 1974). Nitrogen concentrations in food processing waste water varies widely, as discussed earlier. When designing or evaluating a food processing waste water irrigation field it is necessary to obtain data on N concentrations in the waste water and the amount of water being applied.

Loading rates for food processing waste water irrigation are available for a few cases. Smith et al. (1975) studied waste disposal at five potato processing waste water irrigation systems in Idaho and calculated N application rates for the fields. Nitrogen applications ranged from 800 to 2,200 kg/ha annually. These values are higher than the grass crops grown on the fields can be expected to utilize and will increase soil NO_3^- and possibly pollute ground water under the fields. Adriano et al. (1974b, 1975) measured N fertilization and utilization at two sites in lower Michigan. At one site vegetables, fruit, and occasionally meats were processed for 20 years before the study was initiated and the waste water was applied to land that grew

quackgrass (*Agropyron repens* (L.) Beauv.) that was clipped but not harvested. Average annual N application was 365 kg/ha. At the other site, dairy products were processed and the waste also applied to a field that grew quackgrass. Annual N application was 359 kg/ha. These application rates were not excessive if the grass had been harvested and removed from the fields.

While N loading should be a primary consideration in designing food processing waste water irrigation systems, the question of organic loading should also be considered. If the limits of N application are the amount of N that can be utilized by crops, then organic matter application rates will seldom be excessive. Organic matter applications in waste water should be limited to the amount that will decompose during the time between applications. The exception to this is wintertime irrigation with warm water when the organic matter accumulates. Jewell (1976) and Jewell and Loehr (1975) in field and laboratory experiments showed that under favorable conditions, soils that are conditioned to receive waste water containing organic matter can utilize high loading rates. In laboratory experiments, they found that at 26°C removal efficiency was nearly 100% at a vegetable processing waste water application rate of 19,000 kg COD/ha per day. Field sampling of waste water at two vegetable processing plant spray irrigation fields indicated that loadings up to 9,000 kg COD/ha per day were removed with >99% efficiency. While these removal rates may not be widely obtainable, the reports showed that food processing wastes are readily treated in the field and that the capacity of the soil to assimilate these types of organic wastes is very large.

The data reported earlier showed that most food processing waste water contains highly variable concentrations of NH_4^+ and organic N. During waste treatment or decomposition in soil, the organic N is converted to NH_4^+. The capacity of soils to absorb NH_4^+ is generally considered to be quite high. However, in cases where large amounts of NH_4^+ or readily mineralized organic wastes are applied to soil, the exchange capacity of the soil may be exceeded and NH_4^+ may migrate. Lance (1972) developed a method for calculating potential NH_4^+ migration in soil and related it to the NH_4^+ adsorption ratio (AAR). This relationship is similar to the Na adsorption ratio that is used extensively in evaluating Na movement in soils. The AAR is calculated as:

$$AAR = \frac{NH_4^+}{[\frac{1}{2}(Ca^{2+}) + \frac{1}{2}(Mg^{2+})]^{1/2}}$$

where concentrations are expressed in meq/liter. As this ratio increases the exchangeable NH_4^+ percentage increases. In high infiltration rate soils, the adsorption of NH_4^+ is not a very important N removal factor because the soils are soon saturated with NH_4^+. But in agricultural soils that have a high CEC, most of the NH_4^+-N can be removed from the waste water used for irrigating crops. The NH_4^+ will subsequently be converted to NO_3^- in the aerated soil between waste water irrigations.

D. Nitrification and Denitrification

The organic materials contained in the waste water from vegetable and fruit processing are mostly water soluble, readily decomposable, and have a relatively low molecular weight. Before application, much of the waste water is filtered leaving only the soluble and reflocculated organic materials. This is evident in the extremely rapid decomposition rates observed by Jewell (1976) with vegetable processing waste water organic materials. As these organic materials rapidly decompose, the organic N is also rapidly converted to NH_4^+ and then to NO_3^-. Ammonification is seldom a rate-limiting step and crops grown on the waste water treated fields will usually have adequate or excessive N. The waste water organic matter is usually adequate to high in N and decomposition is seldom slowed by N deficiency.

Excessive amounts of N are often added in food processing waste water, indicating that denitrification must also be considered. Smith et al. (1976) determined the potential for denitrification in a field irrigated with potato processing waste water. About 2,400 kg N/ha was applied during 2 years of irrigation. The saturated zone in the soil rose from below 140 cm to the 65-cm depth. Anaerobic conditions were measured at the 65-cm depth by platinum electrodes. The soluble organic material that leached to 65 cm provided energy for the denitrifying microorganisms, and denitrification removed nearly all of the NO_3^-. In another experiment with a deep water table (>25 m), Smith et al. (1978) showed that denitrification could be enhanced by irrigating with high organic waste water at strategic times during the warm season. This lowered redox potentials and promoted denitrification. All irrigations lowered redox potentials, but only irrigations with high organic waste water during warm weather lowered the redox potential sufficiently that denitrification occurred. These studies showed that denitrification can be managed and used as a tool to regulate leaching and loss of N to the ground water in waste water treatment and disposal fields.

E. Pollution Potential

Irrigating with food processing waste water in many cases applies N greatly in excess of that required for growing crops and management becomes the key to pollution control. Adriano et al. (1974b, 1975) measured N leaching in the study cited previously and found that when the grass grown on the fields treated with waste water was not harvested and removed, most of the applied N was leached. For fields receiving 365 kg and 359 kg N/ha per year, 76 and 69% of the added N was leached, respectively. Much of the leaching loss probably represented N that had been returned to the field in unharvested quackgrass.

In contrast to Adriano's experience, Smith et al. (1976) showed that organic matter from potato processing waste water applied to soil the previous winter decomposed as soil temperatures increased in the summer, releasing N that was utilized by growing grass. The excess N was denitrified as it

leached into the anaerobic zone near the water table. In this case, a large excess of N was disposed of without polluting the ground water. When irrigating with relatively high organic waste water, it is possible to manage the soil redox potentials to develop occasional low redox conditions that will denitrify any excess NO_3^- when needed, even in soil without a high water table. Temporary artificial water tables develop in the soil at soil particle size phase changes such as a change from silt loam to gravel, and these can be made anaerobic rather readily by irrigating with water with a high oxygen demand.

In a well-managed system in California, Meyer (1974) reported NO_3^- buildup in the soil at depths to 90 cm when irrigating with waste water from fruit and vegetable processing. He was able to grow a winter cereal crop in addition to the summer vegetation and remove most of the residual N from the soil profile that had accumulated during waste water irrigation. This decreased NO_3^- leaching and ground water pollution.

Timm et al. (see footnote Table 8) applied large quantities of tomato processing waste solids to fields at rates from 448 to 1,792 metric tons/ha. This applied 1,461–5,844 kg N/ha which created a lodging problem when growing barley. Excessive NO_3^--N accumulated (up to 8,700 μg/g in the growing crop), with severe potential for NO_3^- leaching through the soil and into the ground water.

IV. MUNICIPAL WASTES

A. Sewage Effluent

In an effort to provide clean rivers and lakes the U.S. Congress passed Public Law 92-500 in 1972 requiring fishable and swimmable waters by 1983. The cost of doing this with conventional waste water treatment methods will be very high. This law created a renewed interest in using soil to renovate waste water. Secondary benefits of this system include water and plant nutrient conservation. Basically there are three methods of using land for renovating waste water: (i) high-rate infiltration, (ii) overland flow, and (iii) crop irrigation.

1. HIGH-RATE INFILTRATION

High-rate infiltration relies on the physical, chemical, and biological properties of the soil profile to remove impurities from waste water. An intensive study of this method was started at Phoenix, Arizona in 1967 (Bouwer et al., 1974b). This site is in the Salt River Valley with a fine loamy sand (0–0.9 m) underlain with layers of sand, gravel, boulders, and traces of clay 75 m deep, where there was an impermeable clay layer. The static water table was at 3 m.

Secondary effluent was applied to the infiltration bed for 10–30 days followed by a 10- to 20-day drying period. The maximum hydraulic loading was 122 m/year using a 20- to 30-day effluent loading and a 10- and 20-day

drying period for summer and winter, respectively. However, Bouwer et al. (1974a) found that a loading of 91 m effluent/year resulted in removal of 30% of the applied N. The effluent contained 20–40 mg NH_4^+-N, 0–3 mg NO_2^--N, 0–1 mg NO_3^--N, and 1–6 mg organic N/liter which resulted in a N addition of about 28,000 kg/ha. The wet-dry cycle used was 10 days wet and 10–20 days of drying. Oxygen and organic C were the limiting factors for denitrification. The effluent from the infiltration basin was suitable for unrestricted crop irrigation and recreation in Arizona.

Satterwhite et al. (1976) reported on a year-round rapid infiltration system at Fort Deven, Mass. which has been receiving unchlorinated Imhoff effluent since 1942. The infiltration beds were underlain with silty sand to sandy gravel with 10 to 15% silt and clay. The annual effluent application was 27.1 m with a 2-day application and a 14-day drying period. A total-N balance showed a 60–80% reduction in total N, primarily by denitrification. The ground water in the immediate area contained from 10 to 20 mg NO_3^--N/liter.

In comparing the warm-arid and the cold-humid locations from the previously noted studies, the cold-humid site had a greater N reduction. This may possibly have been caused by higher organic C in the Imhoff effluent, lower loading rate, and long drying cycles, which allowed more time for N mineralization at the cold-humid site. Bouwer et al. (1974a) showed 80–90% denitrification when glucose was added to the system to supply organic C. In both of these locations the ground water quantity and quality were affected. The NO_3^- concentration of the ground water was increased, but this impact was ameliorated by lateral NO_3^- movement at the Fort Deven site and pumping NO_3^- containing water to the surface for crop irrigation and nutrient utilization by growing plants at Phoenix.

High-rate infiltration is best suited to areas where water conservation is essential and the soil is deep and permeable. Control of the ground water at the site is necessary and this can be done using tile drains or recovery pumps (Reed, 1972). This high-rate infiltration system may also be suited to some seasonal operations such as canners or summer camps. With proper management, this system can be very successful. The N in this treated water is readily available to growing crops.

2. OVERLAND FLOW

Overland flow systems rely on the controlled release of waste water effluent onto sloping land. The water ideally should flow in sheets through a grass cover. This system is suitable for very slowly permeable soils. Precision grading is advisable to optimize the renovation process. Hunt (1972) stated that there are five primary mechanisms of stripping N from waste water during overland flow: (i) removal by plants, (ii) immobilization in the cells of the expanding aerobic heterotrophic microbial population, (iii) gaseous loss of NH_3 on alkaline soils, (iv) micosite-slime layer denitrification, and (v) the conversion of N into stable organic matter.

The N removal by plant harvesting can be quite significant. At Paris, Texas, 42% of the applied N was removed with the harvested forage crop

(Hunt, 1972). However, in a sparse pine forest little N was removed by the vegetation and no improvement was realized on frozen ground (Sopper, 1968). The immobilization of N by the expanding microbial population and the conversion of N into stable organic matter may be significant during the startup of an overland flow site, but when the system reaches steady-state these two mechanisms are of minor significance.

Slime layer denitrification tends to occur on wet areas where the waste water is repeatedly applied. This process is similar to a trickling filter with its slime-layer on the media.

Raw waste water from Melbourne, Australia is treated in the winter by overland flow through Italian ryegrass (*Lolium perenne* L.). The retention time of the field is about 2 days and the ryegrass is grazed during the summer when the site is not being treated. Seabrook (1975) estimated 60% total N removal with this system. The raw waste water contained 14.3 mg organic N, 35 mg NH_4^+-N, 0.75 mg NO_2^--N, and 0 NO_3^--N/liter. Daily application during the winter was about 1.9 cm.

The overland flow system can handle more effluent than the crop irrigation system, and the treated water goes to surface streams with some evaporation and percolation. The N removal is primarily by plant uptake and denitrification. Overland flow can be a viable means of waste water renovation if the system is managed properly and the waste water application rate does not exceed the removal capacity of the soil and crop produced.

3. CROP IRRIGATION

Crop irrigation maximizes both water renovation and plant nutrient conservation. The effluent is generally applied to crops by overhead sprinkler or flooding at rates of 0.6–2.4 m/year.

The success of waste water renovation will be dependent on the crop selected. Long season high-yielding sod crops utilize more N than annual crops. Hook and Kardos (1977) reported reed canarygrass (*Phalaris arundinacea* L.) receiving effluent daily at a rate of 5 cm/week provided 57–71% removal by forage harvest and the concentration of NO_3^--N in the ground water was generally below 10 mg/liter. An effluent ratio of NH_4^+-N/NO_3^--N of 2:1 resulted in less NO_3^--N loss to recharge water than when the sewage treatment plant produced effluent having a NH_4^+-N/NO_3^--N ratio from 1:2 to 1:5. At Hanover, New Hampshire Iskandar et al. (1976) observed the NH_4^+-N remained in the top 45 cm of the soil profile during the winter. In the summer, almost all the NH_4^+ was nitrified to NO_3^-.

In a study by Clapp et al. (1977) in Minnesota, forages utilized N throughout the growing season better than did corn. Over the entire growing season, less NO_3^- and NH_4^+-N were present in the soil water (60- and 125-cm depth) when 415 kg N/ha from effluent than when 341 kg N/ha from inorganic fertilizer was applied.

Irrigation of trees with sewage effluent appears most promising on a young, fast growing forest because N uptake is greatest during this growth time. Sopper (1968) reported the renovation efficiency of the forest de-

creases with time because there is no annual harvest. During the early stages of effluent application, the N content of the tree leaves, the soil biomass, and the soil organic matter increases; however, this condition eventually reaches a new steady state, and the N utilization and water renovation decreases unless the trees are harvested. Sopper (1968) reported a mixed hardwood forest site in Pennsylvania which was capable of renovating sewage effluent for ground water recharge. Hook and Kardos (1977) reported no N renovation was observed on this same site during 1973 after 8 years of year-round operation.

The selection of crops to be irrigated with sewage effluent is limited. Baier and Fryer (1973) reported stone fruits, citrus, grapes, sugar beets, potatoes, avocados, apples, melons, and squash were of lower quality when fertilized with sewage effluent, as compared with chemical fertilization and irrigation. They attributed this, in part, to the continuous addition of N throughout the growing season. Since the N is only needed for vegetative plant growth, this problem could have been reduced if an alternate water source were used during the period of fruit set. Day and Tucker (1959) reported that sewage effluent–irrigated wheat and oats produced 263 and 249% more pasture forage, respectively, than did check plots that received only pump water. Barley was more sensitive than wheat to the detrimental effects of sewage effluent. Corn yields may be reduced from effluent irrigation because the effluent N concentration is not high enough during the critical period of corn growth, unless supplemented by N fertilizer. Hook and Kardos (1977) observed the greater N leaching with corn because the highest rainfall occurs in the spring and fall when the cropped land is not irrigated. They also found that double-cropping or no-till planting can reduce the movement of NO_3^- to the ground water. This system of using the plant to hold the NO_3^- in the surface soil has the added benefit of reducing soil erosion. The management of a no-till system is more difficult than conventional tillage and with some soils periodic deep tillage is needed to maintain the infiltration capacity.

Treated sewage effluents can be effectively used as a source of N for growing crops if the crop N requirements are considered and N is either supplemented or withheld as the need develops. In addition, sewage effluent will be further purified, ground water can be recharged, and excess waste water disposed by this efficient method.

B. Sewage Sludge

1. SLUDGE PROPERTIES

Sewage sludge is obtained from waste water treatment. The N content of the sewage sludge depends on the nature of the waste water, type and extent of waste water treatment, type of sludge stabilization, and age of the sludge. Table 13 gives N concentrations of the anaerobically digested sludge from three sewage treatment plants with primary and waste-activated secondary waste water treatment.

Table 13—Total and NH_3-N concentrations in anaerobically digested sludge from three waste water treatment plants (Peterson et al., 1973; McCalla et al., 1977).

Water treatment plant	Capacity	Total N	NH_3-N
	m^3/day	% dry wt basis	
Hanover Park	2.3×10^4	5.2	2.4
Calumet	9×10^5	5.6	3.6
West-Southwest	3×10^6	6.9	3.3

The Hanover Park treatment plant is small with primarily domestic sewage while the Calumet and West-Southwest treatment plants are large with about a 3:2 ratio of domestic to industrial waste water. The total N concentration of the digested sludge at the West-Southwest Plant is higher because approximately two-thirds of the primary sludge is not anaerobically digested, but is diverted to drying beds.

King (1977) reported a threefold total N change in sludge composition over a 6-month period at Chatham, Ontario. McCalla et al. (1977) observed no significant seasonal differences in the N concentration of the digested sludge at the Metropolitan Sanitary District of Greater Chicago (MSDGC) West-Southwest waste water treatment plant. The greater diversity of industries using the larger plant may explain the lack of a significant seasonal N flux in this sludge.

Sommers (1977) reported on the chemical composition of sewage sludge from 150 treatment plants located in six states (Table 14). These data were skewed, therefore, the discussion will be based on the range and

Table 14—Concentrations of organic C, total N, NH_3-N, and NO_3^--N in sewage sludge (Sommers, 1977).

Component	Sample Type†	Sample Number	Range	Median	Mean
			%		
Organic C	Anaerobic	31	18–39	26.8	27.6
	Aerobic	10	27–37	29.5	31.7
	Other	60	6.5–48	32.5	32.6
	All	101	6.5–48	30.4	31.0
Total N	Anaerobic	85	0.5–17.6	4.2	5.0
	Aerobic	38	0.5–7.6	4.8	4.9
	Other	68	0.1–10.0	1.8	1.9
	All	191	0.1–17.6	3.3	3.9
			mg/kg		
NH_4^+-N	Anaerobic	67	120–67,600	1,600	9,400
	Aerobic	33	30–11,300	400	950
	Other	3	5–12,500	80	4,200
	All	103	5–67,600	920	6,540
NO_3^--N	Anaerobic	35	2–4,900	79	520
	Aerobic	8	7–830	180	300
	Other	--	--	--	--
	All	43	2–4,900	140	490

† Type of sludge treatment.

median values only. The NH_4^+-N constituted 28% of the total N present in anaerobically digested sludges. King (1977) reported 29% of the total N in anaerobic sludge was NH_4^+-N, ranging from 15 to 59%.

The organic C concentration in the sludges of unspecified treatments was the highest and the N concentration was the lowest. The resulting C/N ratio was 18. The C/N ratios of the anaerobic and aerobic sludges were 6.4 and 6.1, respectively. Peterson et al. (1973) reported 21% of the volatile solids were hexane-soluble materials. Varanka et al. (1976) reported that 19% of the total organic fraction of anaerobically digested sludge was fats, oils, and waxes. The other organic C materials include polysaccharides and proteinaceous constituents (Sommers, 1977).

The presence of NO_3^--N in sewage sludge is a function of sludge age and moisture content. Nitrate is not present in anaerobic sludge freshly drawn from a digester. However, during storage in an open lagoon the solid fraction settles, and the liquid fraction, which contains most of the NH_4^+, is oxygenated by wave action and nitrification can occur.

2. SLUDGE NITROGEN REACTIONS IN SOIL

When sewage sludge is applied to land, the following physical and microbiological processes occur: decomposition, ammonification, nitrification, immobilization, volatilization, and denitrification. Additions of sludge to the soil cause a rapid growth of zymogenous microorganisms that thrive on the sludge organic fraction.

The biological activity of a sludge-amended Blount silt loam soil (fine, illitic, mesic Aeric Ochraqualfs) was studied by Varanka et al. (1976). The soil had been receiving liquid digested sewage sludge for 6 years and was in continuous corn production. The total accumulative sludge application (dry wt) ranged from 0 to 369 metric tons/ha. Table 15 presents the concentrations of the major organic matter fractions found in the soil during the sixth year of sludge application and in the sludge used in 1973 and 1974.

The soil concentration of fats, waxes, and oils increased most with these sludge additons. This fraction is rather stable in soils. The total organic C nearly doubled with the addition of 180 metric tons organic matter from the sludge. About 50% of this C was evolved during the 6 years.

Soils were collected on three dates in 1974 for bioassays. Liquid sludge was first applied (15 July) after the second sampling (Varanka et al., 1976). Their data showed no clear proof of restrictions in microbial populations and enzyme activities. The percentage of denitrifiers and the protease and amylase activities increased as a result of sludge additions and invertase and urease activities were unaffected by sludge treatments. The actinomycetes population increased with rate of sludge additions after the 15 July sludge application. Also, the soil was drier at the last sampling (29 August). This favored actinomycetes growth over fungi and bacteria. The CO_2 evolution was always positively correlated with the amount of sludge applied.

The decomposition rate of digested sludge in soil was studied by Miller (1974) at 1/3 bar and saturated soil water contents and at application rates of 90 and 224 metric tons dry sludge/ha for 1, 3, and 6 months. Incubation

Table 15—Major organic matter fractions in percentage of total organic matter in the surface (0–15 cm depth) of Blount silt loam, the sludge-mended soil, and two samples of digested sludge collected in 1973 and 1974 (Varanka et al., 1976).

Fractions	Soil		Sludge	
	Nonamended	Amended with 369 metric tons/ha sludge	1973	1974
	% of total organic matter			
Fats, waxes, and oils	1.48	8.68	19.75	19.07
Resins	4.03	5.30	3.82	8.20
Water-soluble polysaccharides	7.79	5.47	3.22	14.38
Hemmicellulose	15.45	13.15	4.04	5.97
Cellulose	4.98	3.86	3.45	3.22
Lignin-humus	29.58	31.11	16.78	14.51
Protein	32.41	29.81	24.11	39.60
Total recovered	95.72	97.38	75.17	104.95
Total organic matter†	2.48	5.43	44.76	38.53

† Organic C determined by Walkley-Black method; Total organic matter (% of dry soil) = % C × 1.724.

temperatures were programmed for normal Ohio diurnal and seasonal variations. The sludge used had a C/N ratio of 8. The soils used were Paulding clay (very-fine, illitic, nonacid, mesic Typic Haplaquepts), Celina silt loam (fine, mixed, mesic Aquic Hapludalfs), and Ottokee sand (mixed, mesic Aquic Udipsamments). Miller used the rate of CO_2 evolution to measure the decomposition rate of sludge. At the high application rate, decomposition depended largely on soil texture or chemical properties. On the sandy soil, moisture did not affect the rate of sludge decomposition, but on the clay soil decomposition stopped when the soil was saturated. Soil temperature had a major influence on the rate of sludge decomposition.

Mann and Barnes (1956) observed mineralization of 38% of organic C from digested sludge applied at an annual rate of 18.8 metric tons/ha over 9 years. Premi and Cornfield (1969) found that at low sludge application rates (114 kg/ha of NH_4^+-N from sludge) rapid mineralization occurred, but with higher sludge application rates there was a lag phase before nitrification began. Also, mineralization of the soil organic N was stimulated at the lower application rates. In their 8-week study, mineralization of N from the solid portion of the sludge was negligible. Molina et al. (1971) noted that only the NH_4^+ fraction was immediately available to the plants. They observed inhibited germination of corn and soybeans if freshly digested, high NH_3 sludge was applied. Aging the sludge before application to the soil or avoiding immediate planting after a liquid sludge application eliminated this inhibition.

Addition of digested sludge with a C/N ratio of 8 to most soils will rapidly increase soil NO_3^- levels, and if very high sludge application rates are used, this excess NO_3^- may leach to ground water. To reduce the rate of sludge N oxidation, a high C material can be added to the sludge. Agbim et al. (1977) used wood and bark which they mixed at various ratios with

sludge to regulate the rate of CO_2 evolution. The greatest rate of CO_2 evolution was with sludge alone, and as the ratio of either wood or a wood-bark mix was increased, the rate of CO_2 evolution decreased. The net results are that by proper management of this sludge-wood mixture, sludge can be applied at higher application rates, the soil organic content will be increased, the crop will have a more uniform supply of NO_3^-, and the excess NO_3^- will be less apt to leach.

Composted sludge and wood chips were studied in soil by Tester et al. (1977) in a laboratory incubation experiment. The evolved CO_2 and NH_3 were measured, as well as the remaining C and N fractions in the soil. In 64 days of incubation, 16% of the sludge-compost C was evolved and NH_3 volatilization was minimal, regardless of soil texture. Approximately 6% of the compost N in the loam sand treatment had mineralized after 54 days. Evidence of interlattice NH_4^+ fixation was noted in the silt loam and silty clay soils. Ryan and Keeney (1975) reported that when waste water sludge was applied to the soil surface, 11–60% of the applied NH_3-N was lost by volatilization. These losses decreased as the clay content of the soil increased. Volatilization rate increased with increasing rate of sludge application and with repeated application of sludge.

Gilmour and Gilmour (1979) developed a computer simulation model of sludge decomposition. The model used first-order kinetics and considered temperature and water content changes during decomposition. Sludge was considered to have a rapid and a slow decomposing phase. Model output showed sludge half-life to be 13–51 months depending on temperature and moisture regimes. The month of sludge addition to the soil had little effect on sludge half-life. Monthly temperature and moisture data gave better values of half-life than annual average data. When "optimum" data were inserted into the model, half-life was reduced by a factor of 2 to 7 and they concluded that laboratory studies under "optimum" conditions would greatly underestimate half-life under field conditions.

Peterson et al. (1973)[2] studied NH_3 volatilization and NH_4^+ fixation by sludge fertilized calcareous strip-mined spoil material (pH 7.8). During this 2-week laboratory experiment, fixed, exchangeable, and water-soluble NH_4^+ were measured. The highest liquid sludge treatment was equivalent to 89.6 metric tons/ha dry solids (1,622 kg NH_4^+-N/ha). This spoil material had a 28% clay content. The dominant clay mineral was illite (54%). Kaolinite, chlorite, and vermiculite were estimated to be 28, 8, and 11% of the total clay fraction, respectively. The surface area of the clay fraction was 138 m^2/g which indicated a small quantity of expandable clay minerals. This experiment indicated that 35% of the added NH_4^+ was fixed by the clay minerals. After 2 weeks of incubation, up to 14.7% of the added NH_4^+ was water-soluble and 8.7% was exchangeable. By difference, the NH_3 volatilization was calculated to be about 50%, regardless of sludge application rate, with most of this loss occurring during the first week. No NO_2^- or

[2] J. R. Peterson, J. Gschwind, L. Papp-Vary, and R. L. Jones. 1973. Ammonia volatilization and ammonium fixation by sludge fertilized strip-mined spoil material. Agron. Abstr. p. 180.

NO_3^- was detected in the soil after the first or second week of the experiment.

Keeney et al. (1975) estimated that 15–20% of the sludge N is mineralized the first year and, after initial application, about 6, 4, and 2% of the remaining N is released for the 3 subsequent years.

King (1973) found that in 18 weeks 22% of the surface-applied and 38% of he incorporated sludge N was mineralized. Surface application of sludge resulted in 36% of the applied N being lost and incorporation resulted in a 22% loss in 18 weeks. Only a small part of this loss was NH_3 volatilization. King concluded that denitrification was the major pathway of N loss. The large addition of organic materials provides the energy source required for denitrification and causes an O_2 depletion with the rise in microbial activities. Using this concept, weekly or biweekly addition of liquid sludge on cropland should increase the denitrification rate just as Bouwer et al. (1974b) observed with waste water.

The Illinois Environmental Protection Agency[3] has proposed NH_3 volatilization rates of 50% of the sludge NH_4^+ on sandy and nonsandy soils and a 75% volatilization rate on clay-type soil if the sludge is left on the soil surface. They estimated a 50% loss on sandy soils and a 20% loss on silt and clay-type soils if the sludge is applied by shallow incorporation, such as discing. If the sludge is applied by deep incorporation, such as plowing, the volatilization rate was estimated at 50% on sandy soils and zero on finer textured soils.

To conserve the sludge N, the liquid sludge should be promptly incorporated into the soil. Conversely, to apply the maximum amount of sludge without exceeding the capacity of the crop-soil system and losing NO_3^- to the aquifer, the sludge should be aged, dewatered, and surface applied. The NH_3 volatilization rate will approach 50% with NH_3 surface application and with sludge incorporation the gaseous NH_3 losses will decrease as the depth increases and the soil texture becomes finer. The mineralization rate of the organic N in the sludge is greater at warmer soil temperatures.

3. PLANT RESPONSES TO SLUDGE NITROGEN

To the agronomist, the use of sewage sludge on land must result in an increase in crop quality and quantity. To this end a great deal of research has been done. Kelling et al. (1977b) reported optimum rye and sorghum-sudan (*Sorghum bicolor* × *S. Sudanese*) forage yield with the application of 7.5 metric tons/ha sludge solids on Plano silt loam (fine-silty, mixed, mesic Typic Argiudolls) and with 15 metric tons/ha on Warsaw sandy loam (fine-loamy over sandy or sandy-skeletal, mixed, mesic Typic Argiudolls) during the year of application. Yield of the first crop following the application of 30- and 60-metric tons/ha sludge was depressed, possibly because of excess soluble salts present in the sludge. The residual effects of the sludge were

[3] Design criteria for municipal sludge utilization on agricultural land—draft copy—2nd ed. Tech. Policy WPC-3. Illinois EPA, Springfield. 1977. 35 p.

observed in the crop yield for at least 3 years. Over the 4-year period, 50% of the available N was recovered by the crops at the lower sludge treatment rates and 14% of the available N was recovered at the 60-metric ton/ha application rate.

Coker (1966) observed grass-clover yield increases of 15.2 kg/ha dry forage, for sludge applied at a low rate (70 kg N/ha), and 11 kg dry forage/kg N when applied at a high rate (132 kg N/ha). Stewart et al. (1975) recovered 12% of the sludge N in the corn crop with an application of 400 kg sludge N/ha. Higher rates of sludge application resulted in a less efficient utilization of the sludge N by the corn.

Hinesly et al. (1979) studied the residual effect of digested sludge on corn. Sludge had been applied for 5 years with a maximum total application of 61.1 metric tons/ha of sludge solids for an average of 12 metric tons/ha per year. From 1974 to 1977, corn was grown on these plots without adding more sludge. In 1976 and 1977 significantly higher corn yields were obtained with the highest sludge application. During the years of sludge application, corn yields were higher at all rates of sludge application compared to inorganic fertilizer. The total N and organic C concentration of the 0- to 15-cm soil depth increased with increasing sludge application and the residual total N and organic C remained stable 4 years after the sludge application stopped.

If high sludge application rates are necessary for the operation of a sludge disposal site and if the climate is warm and humid, coastal bermudagrass (*Cynodon dactylon* L. Pers.) with a winter rye (*Secale cereale* L.) cover crop may prove to be the best user of sludge N. King and Morris (1972) reported 63% of the sludge-applied N was used by this cropping combination. The sludge supplied 724 kg N/ha per year. The N concentration of the forage increased with increasing rates of sludge application. The longer the growing season of the crop, the less chance NO_3^- has to leach below the root zone.

The renewed interest in sludge use on land was not primarily because of its fertilizer potential, but rather as an alternative to other methods of sludge disposal. Ecologically, all methods have good and bad points. In the mid-1970's, because of the oil embargo and threats of future oil shortages, a great deal of the organic fertilizer stockpile was reduced. Depending on the quality of the waste, some sources, such as animal manures, were completely cleared. Sewage sludges in the smaller cities were often used for their fertilizer value. In the large cities this also occurred, but owing to the size of the stockpiles, the distance, and the nature of some of the sludges, complete removal of all sludges has not occurred.

4. POLLUTION

This discussion will be limited to N. The application of sludge on land may cause N enrichment of surface and ground waters. If rates and time of application of sludge on cropland are properly managed, the amount of N reaching surface and ground water can be held to a reasonable level.

At high sludge application rates, better management is necessary to hold surplus N in the soil and in the root zone. The practices of double cropping (King & Morris, 1972; Hook & Kardos, 1977) adding wood products (Agbim et al., 1977), composting with carbonaceous wastes (Epstein et al., 1978), or intermittent applications of liquid sewage sludge with dry cycles between application (Bouwer et al., 1974a) can effectively immobilize or volatilize a part of the sludge N added to the soil. The ideal situation is to retain N in the root zone.

Kelling et al. (1977a) reported improved soil aggregation and increased infiltration with sludge application to the soil. Runoff water from a sludge-treated soil slightly increased in organic N but organic N and sediment load were decreased. Peterson et al. (1979) observed no change in total N concentration of runoff water after 5 years of sludge application in two watersheds. A decrease of NH_4^+-N in one watershed and no change in the other was observed after 5 years. An increase from 0.09 to 1.61 mg (NO_2^- + NO_3^-)-N/liter in one watershed and from 0.04 to 0.88 mg (NO_2^- + NO_3^-)-N/liter in the other watershed was observed after 5 years of sludge application.

Atmospheric NH_3 concentration over a sludge-fertilized field was studied by Beauchamp et al. (1978). They observed the maximum NH_3 flux occurred about midday. The *half-life* or time necessary to lose 50% of the ammoniacal N (NH_3-N and exchangeable NH_4^+-N) in the sludge was 3.6 days in May and 5.0 days in October. During the 5-day experiment in May, 60% of the 150 kg/ha ammoniacal N applied in the sludge was volatilized, while during the 7-day experiment in October 56% of the 89 kg/ha ammoniacal N applied was volatilized. The gaseous NH_3 may be absorbed by the surrounding vegetation, surface water, or soil (Denmead et al., 1976).

V. SUMMARY

Crop residues produced annually in the USA amount to approximately 208 million metric tons, which contain 3.56 million metric tons of N. Most of the residues are returned to the soil to supply plant nutrients and organic matter. Turnover in the organic waste materials depends upon their original N percentage and a number of other factors with N release from low N residues being slow and from higher N residues relatively rapid. Rice residues produced outside the USA amount to 454 million metric tons annually; they contain 2.3 million metric tons N. In lowland rice culture, the rice residues containing approximately 0.5% N, have adequate N for optimum decomposition rate, and will supply some of the N needed for the succeeding crop of rice. With the exception of highly acid soils, most organic wastes decompose rapidly with 33% remaining after 1 year and 20% after 5 years.

Animal manure production in the USA is approximately 2 billion tons annually. Manure decomposes fairly rapidly in soil releasing some of the contained N at relatively predictable rates depending upon composition. Several models have been developed for manure decomposition and N re-

lease. Large amounts of manure may be applied for disposal or small amounts for growing crops, the latter is preferred. Pollution of soil and water will probably result from high manure applications, but is not likely if the rates are below 20 metric tons/ha.

A wide variety of vegetables, fruits, meats, and dairy products, amounting to an estimated 56 million metric tons/year are processed that produce large amounts of solid and liquid waste materials containing about 35,000 metric tons N. Most of these waste materials are suited for use on land for fertilization and irrigation of crops. The organic matter in the wastes decomposes rapidly in soil, releasing the N. Most food processing wastes contain high enough N concentrations that they readily serve as sources of fertilizer nutrients. Decomposition is seldom limited by N deficiency in food processing wastes. Overfertilization and potential for pollution of soil and ground water can be a problem with many food processing wastes. Dairy manufacturing and meat packing wastes may need pretreatment or special handling because of unusually high COD and N concentrations in the waste materials to avoid soil and ground-water pollution and other problems such as odors.

Municipal waste effluents are being used extensively for irrigation and fertilization of agricultural crops or for reclamation on land. Distribution methods include high-rate infiltration, overland flow, and crop irrigation. The choice of methods of water distribution is site dependent. Some of the methods are capable of recharging surface and ground water, and providing water and nutrients for growing plants. Nitrogen in these previously treated waste waters is usually NH_4^+ or NO_3^- and is readily available to growing plants. Nitrogen fertilization can be efficient and turnover rates rapid with irrigation using municipal waste water. Nitrogen mineralization in sewage sludge is relatively slow compared with municipal effluents and other waste materials.

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