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R.J. Wright, W.D. Kemper,
P.D. Millner, J.F. Power, and
R.F. Korcak, Managing Editors

Abstract

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America's cities, farms, and industries are producing increasing amounts of byproducts. Biosolids and solid residues from our cities, animal manures from our farms, and coal combustion residues and other byproducts from industries require environmentally safe and cost-effective methods of disposal. The byproduct use problem presents a challenge and an opportunity for U.S. agriculture. Animal manures and many municipal and industrial byproducts may have substantial value if properly used in agriculture. The development of methods to optimally integrate byproduct use into sustainable agricultural practices could provide a partial solution to byproduct disposal problems.

This report emphasizes potential agricultural uses for major byproduct sources, including municipal byproducts (biosolids and solid residues), industrial byproducts (coal combustion residues and other selected byproducts), and animal manures. Individual chapters address each major byproduct source by providing information about amount produced, composition of the waste, current uses, problems and opportunities associated with agricultural and horticultural uses of the byproduct, and research needs. An executive summary provides an overview of the issues involved in using byproducts in agriculture and describes the research needed to transform municipal, animal, and industrial byproducts into an environmentally safe agricultural resource.

The audience for this publication includes scientists and administrators in research, education, and industry, and policymakers.

Keywords: aggregate fines, air quality, beef cattle manure, best management practices, biosolids, calcium silicate slag, coal combustion byproducts, composting, dairy cattle manure, flue gas desulfurization, fluidized bed combustion, gypsum, horticultural uses, incineration ash, land application, municipal solid residues, nitrogen, phosphogypsum, phosphorus, poultry manure, recycling, swine manure, water quality, wood ash

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Contributors

M.C. Brumm, professor of animal science, University of Nebraska, Northeast Research and Extension Center, Concord, NE 68728

T.C. Daniel, professor of soil and water chemistry, University of Arkansas, Department of Agronomy, Fayetteville, AR 72701

Bahman Eghball, research soil scientist, University of Nebraska, Department of Agronomy, Lincoln, NE 68583

Jerry L. Hatfield, laboratory director, USDA–ARS, National Soil Tilth Laboratory, Ames, IA 50011

Robert K. Hubbard, research soil scientist, USDA–ARS, Southeast Watershed Research Laboratory, Tifton, GA 31793

Donald D. Kaufman, research collaborator, USDA–ARS, Soil Microbial Systems Laboratory, Beltsville, MD 20705

W. Doral Kemper, research collaborator, USDA–ARS, National Program Staff, Beltsville, MD 20705

Ronald F. Korcak, associate director, USDA–ARS, Plant Sciences Institute, Beltsville, MD 20705

Robert R. Lowrance, research ecologist, USDA–ARS, Southeast Watershed Research Laboratory, Tifton, GA 31793

S.W. Melvin, professor of soil water waste management, Iowa State University, Agricultural and Biosystems Engineering Department, Ames, IA 50011

Patricia D. Millner, research microbiologist, USDA–ARS, Soil Microbial Systems Laboratory, Beltsville, MD 20705

P.A. Moore, Jr., research soil scientist, USDA–ARS, Poultry Production and Product Safety Laboratory, Fayetteville, AR 72701

James F. Power, research collaborator, USDA–ARS, Northern Plains Area Soil and Water Conservation Research Laboratory, Lincoln, NE 68583

Andrew N. Sharpley, research soil scientist, USDA–ARS, Northeast Watershed Research Laboratory, University Park, PA 16802

Lawrence J. Sikora, research microbiologist, USDA–ARS, Soil Microbial Systems Laboratory, Beltsville, MD 20705

Marion E. Simpson, research collaborator, USDA–ARS, Soil Microbial Systems Laboratory, Beltsville, MD 20705

C.W. Wood, professor of soil fertility, Auburn University, Department of Agronomy and Soils, Auburn, AL 36849

Robert J. Wright, national program leader—soil management, USDA–ARS, National Program Staff, Beltsville, MD 20705

Executive Summary

R.J. Wright

America's cities, farms, and industries are generating in excess of 1 billion tons of byproducts each year. Most of the 300 million tons of municipal byproducts (biosolids and solid residues) produced annually are placed in landfills, but the total number of landfills is decreasing. New, environmentally safe landfills that meet U.S. Environmental Protection Agency (EPA) standards are costly. While landfill disposal costs have stabilized in some areas of the country, the long-term trend is toward increasing tipping fees.

Many of our urban areas have an urgent need for long-term environmentally safe methods for recycling and disposal of byproducts. Our industries produce several hundred million tons of byproducts annually. Alternative uses have been found for a small fraction of these materials, but most byproducts are stockpiled at the site of generation or are taken to landfills. Most of our meat and other animal products are produced by large cost-effective operations in which livestock and poultry in confinement generate substantial quantities of manure annually. Accumulation of large amounts of animal, municipal, and industrial byproducts at the production site can result in degradation of soil, water, and air quality. Components of the byproducts and materials resulting from their degradation can cause odor problems. Also, greenhouse-effect gases such as carbon dioxide and methane can be released into the atmosphere, and nutrients, trace elements, and pathogens can contaminate surface water and groundwater. Currently available agronomic management practices need to be used and new management practices developed to protect environmental quality and to effectively use byproducts in agricultural production systems.

Byproduct use problems present a challenge and an opportunity for U.S. agriculture. We are currently confronted with the long-term goal of developing crop production practices that promote sustainability. Sustainable agriculture is characterized by plant and animal production practices that satisfy human food and fiber needs while enhancing environmental quality and the natural resource base. Efficient use of nonrenewable resources and on-farm resources is an important component of sustainable agriculture. Animal manure and many municipal and industrial byproducts have substantial value for agricultural use. Many of the byproducts contain essential nutrients that could meet crop requirements if applied to our land in the proper manner at the right time and in suitable amounts. Use of nutrients from byproducts could reduce dependence on fertilizers from our limited supply of mineral resources and thereby increase the sustainability of our agricultural systems. Organic byproducts can be valuable as soil conditioners and as a means of enhancing soil organic matter levels, which tend to decline with cultivation. Byproducts, byproduct compost, or mixtures of byproducts also may find specialty uses in the horticultural industry. The development of methods to optimally integrate byproduct use into sustainable agricultural practices could provide a major part of the solution to urban and industrial byproduct disposal problems.

Municipal Byproducts

The United States has reached a critical stage in the management of our two major municipal byproducts: municipal solid residues (MSW) and biosolids. Total generation of these two byproducts exceeded 325 million tons in 1995, with MSW accounting for approximately 95 percent of the total. MSW consists of a variety of components, including paper and cardboard products (approximately 35 percent by weight), yard wastes (approximately 20 percent by weight), and metals, plastic, glass, wood, and food wastes (each comprises approximately 6–9 percent by weight). The majority of MSW is placed in landfills, but the number of landfills has decreased from approximately 8,000 in 1988 to 3,200 in 1995. Thus recycling and agricultural uses of MSW are becoming preferable to landfilling.

Many of the components of MSW (paper, yard waste, food wastes, wood products) are biodegradable under proper conditions and may have potential to be used to improve agricultural and nonagricultural land. Because of limitations associated with odors, pathogens, and undesirable chemical and physical properties, new and unstable organic municipal byproducts cannot be added directly to the soil. However, composting of biosolids and selected MSW components is an effective waste management process. Composting is a self-heating microbiological process in which the decomposition of organic materials is accelerated by the growth and enzymatic activity of mixed populations of bacteria and fungi. Composting reduces the weight and volume of the byproduct while abating odors, destroying pathogens, and converting nutrients to forms that are more available to plants. As an alternative to composting, pathogen destruction and organic matter stabilization can be achieved by blending alkaline byproducts such as cement kiln dust, lime kiln dust, or coal combustion ash with biosolids. The alkaline stabilization process depends on exothermic chemical reactions to generate the high pH, heat, and drying effects needed to produce partial pathogen destruction and organic matter stabilization.

Biosolids and selected MSW components have value as biofertilizers and soil conditioners. However, these materials have relatively low nutrient contents and may need to be enriched with inorganic fertilizers to meet plant growth requirements. Municipal byproducts add organic matter to soils, thus maintaining and enhancing soil health and productivity. The EPA has developed standards, based on concentrations of trace elements and toxic organics, to regulate land application of biosolids. Currently the majority of wastewater biosolids are applied to land. Similar regulations are needed for land application of MSW components. Measures will need to be taken to segregate MSW components that may contain excess levels of toxic trace elements and hazardous organic chemicals. The fate and subsequent bioavailability of nutrients, metals, and synthetic organic residues during and after composting of MSW components will have to be determined.

Composted biosolids and MSW may have potential uses in the horticulture industry as growth media and for biocontrol of soilborne plant diseases. The safe and beneficial use of municipal byproducts in agriculture and horticulture will depend on our ability to develop products with a known and consistent range of physical and chemical characteristics. Compost quality and maturity criteria need to be developed to enhance horticultural uses of municipal byproducts. Methods need to be developed to dependably enhance the microbially mediated plant-disease suppression characteristics of compost and to reliably inoculate horticultural-grade compost with beneficial rhizosphere microbes that can biologically mediate plant nutrient uptake.

Agronomic management practices are needed to minimize the potential of land-applied municipal byproducts to degrade surface water and groundwater. Byproduct additions are generally based on nitrogen (N) requirements of the crop for the growing season. However, tests to estimate the fraction of organic N that will be converted to plant-available forms are not reliable, and N in excess of plant needs may be applied. This situation can result in leaching of nitrate to groundwater. New testing protocols are needed to allow effective management of N in municipal byproducts. Runoff of water from land amended with municipal byproducts can be a source of surface water contamination. A combination of restricted use of municipal byproducts on highly erodible land and soil conservation practices such as buffer strips and stiff grass hedges should prevent pollution of surface waters by nutrients and suspended solids from municipal byproducts.

Animal Manures

Cattle, poultry, and swine are the major sources of animal manure production in the United States. The manure (feces and urine) generated by animals raised in confinement (feedlots, dairy barns, poultry houses, and swine operations), if improperly managed, can result in significant degradation of soil, water, and air quality. There were approximately 99 million head of cattle and calves in the United States in 1990. At any one time, at least 10 million head of beef cattle are confined in feedlot operations. These animals generate 27 million tons of manure solids annually. The dairy cattle population in 1990 was 10.2 million. Dairy cattle in confinement are estimated to produce 21 million tons of solids annually. If improperly managed, the manure associated with beef feedlot and dairy operations can create significant environmental problems, including human health issues associated with contamination of surface water and groundwater.

Poultry operations in the United States produced 6.5 billion birds (chickens and turkeys) in 1990 and generated 14 million tons of litter and manure. About 90 percent of poultry manure is applied to agricultural land. Nonpoint source pollution of surface water and groundwater with N, phosphorus (P), and pathogenic microorganisms is becoming a major problem in states where the poultry industry is undergoing rapid and concentrated growth.

The farm inventory of swine tends to fluctuate between 50 and 70 million animals. Swine produce about 16 million tons of solid waste annually. Approximately 80 percent of the manure generated can be collected, stored, and spread on agricultural land. The major environmental concerns associated with storage or land application of swine manure are surface water and groundwater quality, gaseous emissions, and odors.

The animal manure generated annually in the United States contains about 8.3 million tons of N and 2.5 million tons of P. By way of comparison, about 10 million tons of N and 1.8 million tons of P are applied annually as commercial fertilizers. Animal manures are widely variable in chemical composition, physical properties, and moisture content. The nutrient content of manure varies with animal species, type of diet, growth stage and level of performance of the animal, production system used, amount of bedding material with the manure, and method of manure storage and handling. Average N, P, and potassium (K) contents in beef cattle feedlot manure are 1.9 percent, 0.65 percent, and 2.0 percent, respectively. Nutrient levels in swine manure vary with method of handling and storage but in general are lower than in cattle manure (average 0.5 percent N, 0.1 percent P, and 0.4 percent K). Poultry manure, with its relatively low moisture content and high nutrient content (4.6 percent N, 2.1 percent P, 2.1 percent K), is generally considered to be the most valuable animal manure for fertilizer purposes. The efficient conservation and use of nutrients contained in animal manures could protect environmental quality and greatly reduce the need for purchased fertilizers.

Animal manures, applied in solid, semisolid, and liquid forms, have traditionally been used as a source of nutrients for crop production. In addition, organic components of manure can build soil organic matter reserves, resulting in soils having increased water-holding capacity, increased water-infiltration rates, and improved structural stability. These changes can reduce soil loss by wind and water erosion. Soil-applied manures decrease the energy needed for tillage and reduce impedance to seedling emergence and root penetration. Manures can be used as an organic mulch when the previous crop does not produce sufficient crop residues to protect the soil surface. Manures stimulate the growth of beneficial soil microbial populations, increase microbial activity within the soil, and increase the population of beneficial mesofauna such as earthworms.

In 1990 there were approximately 330 million acres of cropland and 650 million acres of pasture and rangeland in the United States. Nationally, this provides an ample base for land application of animal manures. However, economic and environmental considerations place restrictions on the use of some land areas. Available farmland for application of manure generally exists in close proximity to most beef feedlot operations. Suitable land for safe and economical disposal of poultry manure is already a major problem in many areas of the United States. If the manure has to be transported a significant distance, transportation costs can easily exceed the fertilizer value of the manure. These economic restrictions may result in application of manure on inappropriate sites, namely those with elevated levels of N and P from previous application or those susceptible to runoff and leaching of manure nutrients and pathogens.

Environmental quality must be a major consideration when developing agronomic management practices to effectively use animal manures. Leaching and runoff of nutrients from manure at the production site and after land application can be detrimental to the quality of surface water and groundwater. Leaching of nitrate from animal manures to groundwater can be a health concern. Nitrate levels in excess of the EPA drinking water limit have been found in water wells in areas with high animal manure production and use. Runoff from production sites and fields receiving manure can pollute surface water with nutrients, pathogens, organic materials, and sediments. Phosphorus is the nutrient of primary concern from a surface water standpoint, since it is generally considered to be the limiting factor for eutrophication.

Pathogens in animal manures can be transmitted to other animals and to humans through food supplies and water. Use of animal manures in production of fresh vegetables and fruit could be a mechanism of pathogen transfer. Bacterial, fungal, and protozoan infections also have been related to manure contamination of surface water. Recent interest in this area has focused on *Cryptosporidium parvum*, a widespread protozoan parasite afflicting animals and humans. The dominant mode of transmission of *C. parvum* to humans is believed to be via contaminated drinking water and recreational waters. Although no clear-cut epidemiological cause and effect has been established, it is widely believed that farm animals are the predominant source of *C. parvum*. Dairy farms are particularly suspect as potential sources of *C. parvum* because newborn calves are readily infected and excrete large numbers of the infectious stage (oocyst) of this organism. Since many dairy operations are located near urban areas, careful management of dairy manure will be needed to prevent possible contamination of municipal water supplies. Development and use of agronomic management techniques to control runoff and erosion should enable farmers to safely use animal manures while reducing or eliminating movement of nutrients, organics, pathogens, and sediments to surface water. The Environmental Quality Incentives Program in the Conservation Title of the 1996 Farm Bill will provide cost shares and incentive payments to farmers to implement environmentally safe management of animal manures.

Animal diets are supplemented with a number of growth-promoting vitamins and minerals. The use of mineral supplements such as copper (Cu), zinc (Zn), arsenic (As), selenium (Se), and P result in elevated levels of these elements in the manure. Long-term soil application of these manures could lead to an undesirable buildup of trace elements and P in the soil. Management of soil pH will be needed to control plant uptake and food-chain transfer of Cu, Zn, Se, and As. High levels of trace elements, drug residues, and other contaminants also would limit refeeding of manure to other livestock.

Air quality has become a major environmental concern of the animal production industry. Odors generated at production and manure storage facilities constitute the most frequent source of complaints against animal producers. Uncontrolled decomposition of manure produces odorous gases, including amines, amides, mercaptans, sulfides, and disulfides. Ammonia volatilization from manure creates an odor problem and may contribute to acid rain. These noxious gases can cause animal respiratory diseases. Greenhouse-effect gases such as carbon dioxide, methane, and nitrous oxides also are released from manure handling and storage facilities. Improved manure handling and storage methods are needed to reduce emission of these gases.

Current manure management practices are not designed to effectively conserve and use animal manure nutrients. Nitrogen is the nutrient most susceptible to loss (primarily through ammonia volatilization) after manure excretion and during storage, transport, and land application. Because of these losses, less than 25 percent of the N originally present in the manure may be available for crop production. Swine manure can lose up to 90 percent of its N through ammonia volatilization in anaerobic storage lagoons. Because of these N losses, N:P ratios in manure are generally lower than crop needs, resulting in buildup of excess P in soil. Rapid tests to determine nutrient content and release rates from manure are needed so farmers can apply manure at rates that will supply crop needs. Improved handling, storage, application, and analysis methods need to be developed to effectively use animal manures for enhanced crop production while avoiding environmental degradation.

Industrial Byproducts

A number of industrial byproducts including residues from coal combustion, fertilizer production, the construction industry, and incineration are produced in substantial amounts throughout the country. Many of these byproducts have potential for use in agriculture, but they have not been thoroughly investigated or they have changed in composition as a result of new technology. Coal combustion byproducts are emphasized in this report because their production is rapidly increasing, coal combustion technology is changing, and disposal of byproducts from coal combustion is becoming a major issue.

Combustion of coal produces a variety of byproducts including fly ash, bottom ash, flue gas desulfurization (FGD) residue, fluidized bed combustion (FBC) residue, and coal gasification ash. Total production of coal combustion byproducts was 120 million tons in 1991, with production expected to increase to 170 million tons annually by the year 2000. Fly ash, the major byproduct from coal combustion, is the particulate residue that enters the flue gas stream and is either collected by emission control devices or released to the atmosphere. Bottom ash is the residue that remains in the boiler after coal combustion. FGD and FBC residues are byproducts from technologies used to reduce sulfur (S) emissions from coal combustion. FGD byproducts result from post-combustion treatment of flue gases with absorbent (calcium oxide, limestone, dolomite) to reduce SO_x discharge to the environment. FBC involves removal of SO_x during combustion of a finely divided mixture of coal and limestone on a fluidized bed created by injection of air. Production of FGD and FBC byproducts will increase to greater than 50 million tons annually by the year 2000 as a result of Clean Air Act requirements for reduced S emissions.

Currently about 20 percent of the ash byproducts are recycled. They are used in the construction industry as fill material and as components of other products such as concrete, cement, and asphalt. FGD byproducts are used to a limited extent in the production of sulfur-related chemicals and wallboard. Approximately 80 percent of coal combustion byproducts are retained at the power plant site. Storage techniques include surface impoundments, landfill placement, and placement in mines and quarries. Regulatory barriers to land application and relatively inexpensive on-site disposal have discouraged widespread agricultural use of coal combustion byproducts to date. Potential environmental problems associated with on-site storage and the ever-increasing amount of byproducts are making agricultural use more attractive.

Land application of selected coal combustion byproducts can bring about favorable changes in soil chemical and physical properties. Many of the byproducts are alkaline in nature and can be used as liming materials to increase soil pH. Although coal combustion byproducts are low in N and P, they can serve as a source of other plant essential nutrients, including calcium (Ca), boron (B), molybdenum (Mo), S, Cu, and Zn. Fly ash, alone or in combination with municipal biosolids, has been successfully used to reclaim acidic mined land. FBC byproducts and oxidized FGD materials contain significant amounts of gypsum. Surface applications of gypsum have been shown to be effective in ameliorating subsoil acidity. Byproducts high in gypsum also can improve soil structure, increase water infiltration, and reduce mechanical impedance to root growth.

Plant growth limitations, food chain transfer of trace elements, and water quality issues are among the problems that may be associated with agricultural use of coal combustion byproducts. Many coal combustion byproducts have a high pH and high levels of soluble salts that can severely inhibit plant growth. The most serious potential trace element hazards associated with agricultural use of coal combustion byproducts appear to be B, Se, As, and Mo accumulation in soils and plants. Excessive application rates of byproducts containing high levels of these elements could result in phytotoxic levels of B and elevated levels of As, Mo, and Se in plant tissue. Many FGD byproducts contain appreciable amounts of calcium sulfite. When initially added to soils, calcium sulfite can inhibit plant growth. Fortunately calcium sulfite is rapidly oxidized to calcium sulfate in soil systems. Therefore, if this oxidation can be completed prior to crop planting, it may be possible to add coal combustion byproducts that are high in calcium sulfite directly to soils.

Information is needed about the benefits and risks associated with agricultural use of the new-technology coal combustion byproducts. It is difficult to generalize about characteristics of a particular coal combustion byproduct because chemical and physical properties of these materials depend on a number of factors, including power plant design, operating parameters, sources and types of coal consumed, and types of reactive reagents used in the FGD process. Those coal combustion byproducts with significant amounts of gypsum seem to be most suitable for agricultural uses. More information is needed about the behavior in soils of FGD byproducts containing high levels of calcium sulfite.

In addition to coal combustion byproducts, three other broad classes of industrial byproducts were examined in this report: residues from the P fertilizer industry, residues from construction trades, and incineration ashes. Two byproducts from the phosphate fertilizer industry—phosphogypsum and calcium silicate slag—have some favorable properties for agricultural uses. Phosphogypsum is a byproduct from the production of phosphoric acid from phosphate rock. Approximately 40 million tons of this material are produced in the United States each year. Phosphogypsum has been used in agriculture as a Ca and S nutrient source and as an amendment to ameliorate subsoil acidity. The EPA has disallowed agricultural use of phosphogypsum sources with radium-226 radioactivity in excess of 10 pCi g^{-1}. This ruling has essentially eliminated the use of phosphogypsum from southern Florida but allows use of phosphogypsum produced from sources of phosphate rock with lower radium-226 activities. Calcium silicate slag is a byproduct of electric furnace production of phosphate fertilizer from phosphate rock. This byproduct has value as a liming material and contains plant nutrients. Silicon in the material has been shown to enhance sugarcane resistance to foliar ringspot disease.

The construction trades generate several byproducts with potential agricultural uses, including aggregate industry fines and concrete manufacturing residues. Annual production of sand, gravel, and crushed stone results in approximately 100 million tons of waste fines. Weathering of the mineral phases in the aggregate fines should serve as a slow-release source of plant nutrients. The aggregate industry fines could be mixed with organic residues to create an artificial soil. Approximately 10 million tons of concrete manufacturing residues are generated each year in the United

States. This material is alkaline and has a high content of calcium silicate. The material should be useful as a liming material and would be expected to have some common characteristics with calcium silicate slag. Additional research is needed to characterize benefits and risks associated with agricultural use of these two materials.

Production of ash from combustion of wood waste and the incineration of MSW is increasing. Most of the incineration ash currently is being placed in landfills. This material is likely to be quite variable in composition, particularly in regard to trace element levels. Potential problems with elevated trace element levels in incineration ash probably will preclude its use in agriculture. The combustion of wood waste to produce steam or electricity creates localized sources of ash. Between 1.5 and 3 million tons of wood ash are produced annually. Wood ash generally has value as a liming material and as a source of nutrients. Wood ash should not pose an environmental risk if it is applied to the soil at lime requirement rates.

Research Needs

Investment in research and education will be needed to increase and improve agricultural use of municipal, animal, and industrial byproducts. Additional research in the following areas will ensure efficient and environmentally safe uses for a variety of readily available byproduct materials:

1. Treatment methodologies and management practices need to be developed to minimize loss of nutrients from manure and other byproducts during handling and storage. A substantial portion of the N initially present in manure and in some municipal byproducts can be lost during aerobic composting and through ammonia volatilization, denitrification, leaching, and anaerobic decomposition in lagoons. Liquid wastes such as liquid manure and wastewater from animal operations require development of special treatment methodologies. Improved systems for nutrient capture, immobilization, or recovery should help conserve nutrients during handling and storage until byproducts can be beneficially applied to land.

2. A greater understanding of nutrient and trace element transformations and reactions in byproducts and in soil treated with byproducts is needed. This information will allow systems to be designed to conserve nutrients during byproduct handling and storage. This information also is necessary to predict the environmental fate and bioavailability of nutrients and trace elements in byproducts.

3. Improved nutrient management tools are needed to prevent excess nutrient and trace element buildup in soil from land application of byproducts. Rapid and effective analytical methods are needed to estimate concentrations of bioavailable nutrients and toxic components in byproducts and soils amended with byproducts. The question of basing byproduct application rates on either N or P needs to be resolved. Tests for soil threshold levels of N and P for the purpose of protecting water quality need to be developed, assessed, and used to identify areas at risk from further byproduct application. Improved nutrient management practices will help farmers select byproduct application rates that keep nutrient concentrations within beneficial ranges, avoid contamination of our waters, and facilitate loading rates that are cost effective and timely.

4. Management practices need to be developed to protect surface water and groundwater from nutrients, trace elements, sediments, and pathogens in byproducts applied to land. A variety of management techniques, including vegetative buffers, wetlands, riparian zones, drainage management, and cover crops need to be assessed over a range of conditions to determine where they can be effectively applied to protect water quality and where their use would be inappropriate. Effective methods to remediate sites with excess nutrients and trace elements, including removal with plants and deactivation with industrial and municipal byproducts, need to be developed.

5. Methods of incorporation and surface application of organic byproducts to increase organic matter levels in soils without degrading environmental quality should be evaluated. Many of our soils have lost organic matter through erosion of topsoil and tillage-accelerated biological oxidation. Benefits associated with enhanced soil organic matter levels include protection from erosion, increased water infiltration rates, higher available water holding capacity, increased plant rooting depth, and enhanced supply of nutrients.

6. A careful assessment must be performed to determine the risks associated with trace elements, synthetic organics, and pathogens in byproducts applied to land. A risk-assessment pathway approach similar to that used to develop regulations for land application of municipal biosolids is needed for animal manures and other byproducts. EPA has established regulations for levels of trace elements, synthetic organics, and pathogens in biosolids, including regulations for cumulative amounts of these components that can be applied to land. Appropriate regulations for land application of animal manures and other byproducts is needed to allay public concerns about the agricultural use of these materials.

7. Effective control of odors from manures and other byproducts at handling, storage, and processing facilities is needed. Odors generated at these facilities create air quality problems and public relations difficulties, especially in urban areas. The basic microbial processes involved in odor formation need to be understood and linked to management practices geared to preventing odor generation during handling, storage, and use of byproducts and to preventing escape of odors.

8. Animal use of nutrients and dietary supplements in animal feed must be improved. More efficient conversion of feed to meat and other animal products can reduce the amount of manure produced and lower nutrient levels in the manure. Methods are needed to improve the use of P in grain by poultry and swine, thereby reducing the need for supplemental P additions to feed. Careful use of animal drugs and dietary supplements such as Cu, Zn, As, and Se can limit concentrations of these components in the manure, therefore making it a more valuable product for soil application or animal refeeding.

9. Byproduct quality and consistency needs to be improved to increase the desirability of these materials for agricultural and horticultural uses. The composition of many byproducts is not consistent over time. The use of composting to stabilize organic byproducts and convert them into fertilizers and soil conditioners is likely to increase, but quality and maturity criteria need to be developed. A known and consistent range of allowable physical and chemical characteristics of compost and other byproducts is needed to develop safe and beneficial uses of these materials.

10. Methods for co-utilization of municipal, animal, and industrial byproducts need to be identified and developed. Products with enhanced value and utility for agricultural and horticultural uses can be developed by mixing, blending, or co-composting different byproducts. Byproducts may need to be co-utilized to overcome problems in the materials themselves or to overcome soil problems. This approach can be used to eliminate pathogens and toxins, reduce availability of toxic trace elements, and enhance nutrient availability. Examples of this approach include stabilization of biosolids using alkaline byproducts and the reclamation of mined land with fly ash and municipal biosolids.

11. Alternative uses of manures and other byproducts need to be investigated and developed, especially in areas where land application of these materials can cause significant environmental risk. Many byproducts have potential for energy production through burning, methane generation, or conversion to other fuels, but these processes need to be developed and optimized. Manures and other byproducts such as food industry wastes, when properly used, have greater value as an animal feed than as a soil amendment. Development of methods such as composting or pelletizing to reduce the weight, volume, or form of manure and other byproducts could reduce transportation costs and open up alternative uses.

12. A national database is needed on major municipal, animal, and industrial byproducts. This database should include such details as amounts produced, geographic distribution of production, and the physical and chemical properties of each material. It should be designed to facilitate Geographical Information System (GIS) layering of the various data sets. A range of values for agronomically important parameters such as pH, nutrients, and toxic trace elements will facilitate selection of byproducts that will benefit the soil-plant system and identify those that should have restricted use. This database also will contribute to the development of models and decision-support systems for effective and environmentally safe use of byproducts.

13. Economic analyses of costs and benefits of potential byproduct management practices are needed. Economic viability is an essential component of sustainable byproduct management systems. For certain byproducts, the producer and the public may need to share costs with the farmer. Tradeoffs between farm income and environmental impacts need to be assessed. This approach allows selection of the most profitable byproduct management system to meet state and Federal regulations.

14. Decision support systems need to be developed to optimize land application of byproducts. These systems will need to incorporate recent information about byproduct materials, most effective agronomic management practices, and economic considerations. Decision-support systems should provide farmers, extension agents, farm management consultants, and Natural Resources Conservation Service personnel with a powerful tool to ensure effective use of byproduct materials while preserving environmental quality.

15. Regulations must be developed that allow the use of beneficial byproduct materials and yet protect the environment and human health. Environmental regulations developed and interpreted by individual states currently constitute one of the main barriers to increased land application of these materials. Regulations developed using sound scientific data will help overcome current barriers to agricultural use of byproducts posed by state regulations that are excessively restrictive and that may not have a sound scientific basis.

As the real advantages associated with careful agricultural use of byproducts are determined and hazards are defined and controlled, efforts will be needed to convey this information to the agricultural community and the public. Successful handling of the byproduct disposal problem will require a partnership between the urban and agricultural sectors. The agricultural sector will need to know which byproduct materials can be applied to land, how much can be applied, and which methods of application are environmentally safe. The public will need to be convinced that agricultural use of byproducts is environmentally safe and cost effective and does not pose a human health risk. Byproduct producers and the public may have to pay additional fees to make the use of byproducts more attractive to farmers. Byproduct transportation expenses may have to be subsidized, and additional steps may be needed at the production site to make products more valuable for agricultural and horticultural uses. These expenditures, however, may be small compared to increasing costs of current byproduct management practices and the potential benefits to be gained through environmentally safe use of byproducts in agricultural operations.

Chapter 1

Agricultural Uses of Biosolids and Other Recyclable Municipal Residues

P.D. Millner, L.J. Sikora, D.D. Kaufman, and M.E. Simpson

In the United States, the traditional management practices used for biosolids (also known as sewage sludge) and municipal solid residues (MSW) were driven by the "disposal" concept rather than the resource recovery concept. These traditional practices are now recognized as environmentally, ecologically, and economically inadequate. As a nation we are generating more organic, recyclable municipal residues than ever before, and many areas are rapidly exhausting their standard options for the safe, effective management of these materials. Handling capacity of landfills and some older incinerators continues to decrease. Regulatory guidelines and limits to traditional biosolids and residue management practices also are being implemented at national, state, and local levels. Federal Sub-title D regulations imposing strict liner and management requirements were fully effective in 1994. Thus, it is increasingly difficult to keep existing landfills open or to establish new facilities. In 1995, a total of 3,197 landfills were in operation, but this total was 361 fewer than in 1994 and 4,803 fewer than in 1988 (Steuteville 1996). By 1997, 20 percent of states could exceed their landfill capacity (Repa and Sheets 1992).

Communities face hard choices when evaluating the array of management options available. New York City, for example, has paid premium prices to transport its residues long distances to sites willing to accept and use them. Other communities have encountered intense public debate when siting treatment facilities close to collection sites. Not all communities, however, face such problems. Some have found creative solutions through source reduction and recycling programs and have been able to site new, environmentally acceptable facilities. Still, for much of the Nation, innovative solutions for residue management are much needed.

Identification and characterization of the constituents are an essential first step in developing suitable plans and strategies to deal with the problems associated with biosolids and recyclable organic residues management. For example, characterization of municipal solids involves estimating how much of each component in the mixture is generated, recycled, incinerated, and disposed of in landfills. The data are used to establish management goals and plans at the national, state, and local levels. Characterization of the materials can reveal opportunities for source reduction and recycling and provide data on unique management and application issues (Cook et al. 1994).

Quantities and Management of Municipal Solid Residues (MSW) and Biosolids

MSW

In 1990, the U.S. Environmental Protection Agency (1990a) estimated 177.7 million Mg (1 Mg = 1 metric ton = 1.10 U.S. tons) of MSW were generated in the United States (table 1). This is equivalent to 1.95 kg per person per day (Finstein 1992). After materials recovery for recycling and composting, discards were 1.63 kg per person per day. Virtually all of these discards were incinerated or landfilled. In 1995, 297 million Mg of MSW were generated (fig. 1), with 27 percent recycled, 10 percent incinerated, and 63 percent landfilled (Steuteville 1996). The amount of MSW produced is higher than the figures reported by the U.S. Environmental Protection Agency, which based estimates on per capita rates and the use of estimated disposal by states and did not include biosolids, yard trimmings, and recycling projects (Glenn 1990).

Both volume and weight of MSW are used to evaluate the scope of the recycling problem; volume is used to estimate how quickly landfills will reach capacity and the rate of change of various materials in the residue stream. A breakdown of the 1990 MSW by weight and volume is shown in table 1. Paper and paperboard products are the largest component of MSW by weight (37 percent) and by volume (32 percent). Yard trimmings are the second largest component (18 percent by weight). Glass, metals, plastics, wood, and food residues range from 6 to 9 percent each by weight. Rubber and leather, textiles, and miscellaneous organic solids comprised less than 4 percent each of MSW. Paper and plastics (combined) accounted for over half of the volume of MSW discarded in 1990. The three methods of disposal of MSW in 1990 were landfilling, recycling, and incineration—118, 30, and 21 million Mg of residues were disposed by these methods, respectively (U.S. Environmental Protection Agency 1990a). The composition of MSW indicates

Table 1. Weight, volume, and recovery of municipal solid residues (MSW)* in 1990

	Weight generated (million Mg)	Weight recovered (million Mg)	Percent recovery of material generated	Material discarded (million Mg)	Weight of MSW (% of total)	Volume of MSW (% of total)	Ratio† of vol % to weight %
Paper and paperboard	66.6	19.0	28.6	47.6	37.5	31.9	0.9
Glass	12.0	2.4	19.9	9.6	6.7	2.2	0.3
Metals							
Ferrous	11.2	1.7	15.4	9.4	6.3	8.9	1.4
Aluminum	2.5	0.9	38.1	1.5	1.4	2.2	1.6
Nonferrous	1.5	0.7	67.7				
Plastics	14.7	0.4	2.2	14.4	8.3	21.1	2.5
Rubber and leather	4.2	0.2	4.4	4.0	2.4	6.1	2.5
Textiles	5.1	0.2	4.3	4.8	2.9	6.4	2.2
Wood	11.2	0.4	3.2	10.8	6.3	6.8	1.0
Food scraps	12.0	‡	‡	12.0	6.7	3.2	0.5
Yard trimmings	31.8	3.8	12.0	28.0	17.9	9.8	0.5
Miscellaneous inorganics	2.6	‡	‡	1.4			
Other	2.9	0.7	23.8	5.2	1.6	1.4	0.4
Total	177.7	30.3	17.1	147.4	100	100	1.0

* MSW includes durable and nondurable goods, containers and packaging material, food scraps, yard trimmings, and miscellaneous inorganics from residential, institutional, and industrial sources. Residues include appliances, newspapers, clothing, boxes, disposable tableware, office and classroom paper, wood pallets, and cafeteria scraps. Specifically excluded from MSW are construction and demolition materials, municipal sludge, combustion ash, and industrial process residues and byproducts.

† Ratios greater than 1.0 indicate that the materials are less dense and occupy a greater proportion of landfill space by volume than by weight; the converse is true for materials with ratios of 0.5 or less.

‡ Negligible.

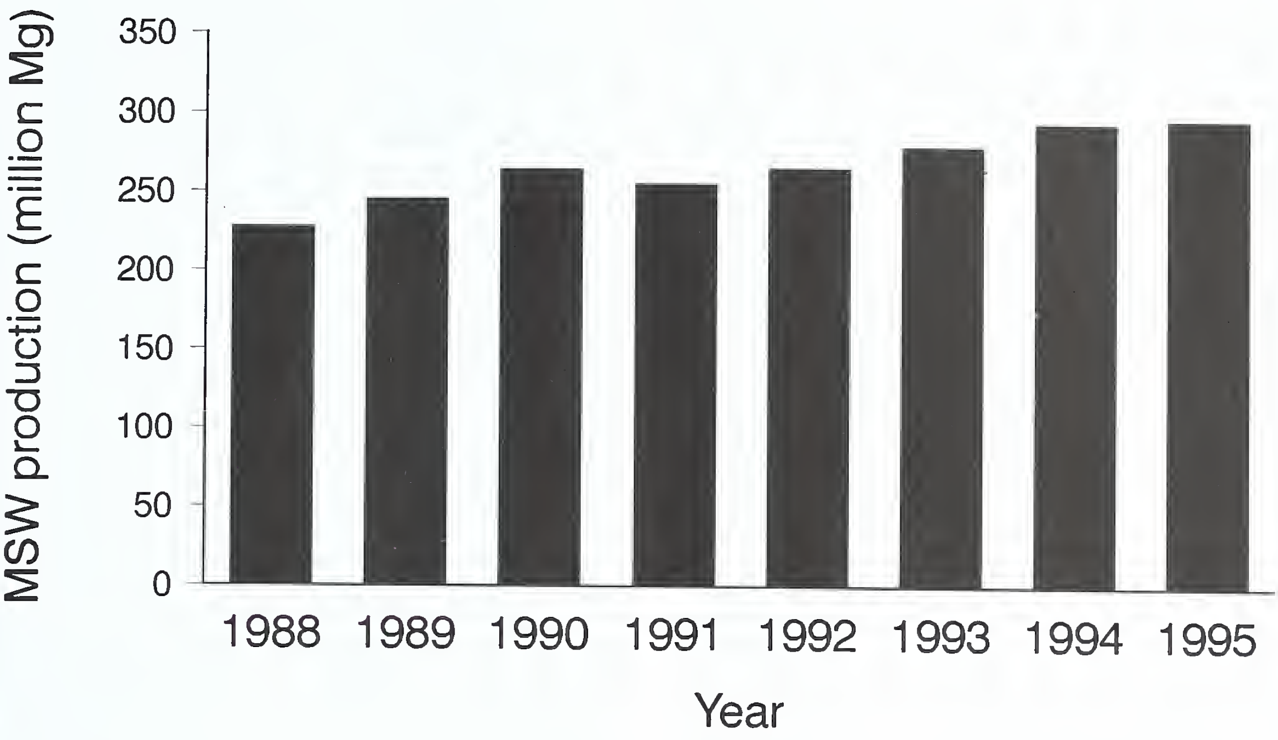

Figure 1. Municipal solid residue (MSW) production in the United States

that 70–80 percent of the residue stream is combustible, recyclable, or compostable (Clarke 1992).

The strong upward trend in recycling is evident in the 82 percent increase in the number of materials-recovery facilities operating in the United States—up from 177 in 1992 to 322 in 1995 (Berenyi 1995). Nationally, the recycling rate in 1995 reached 27 percent for MSW (Steuteville 1996), with 7,375 curbside programs serving 121 million U.S. citizens and 8,773 dropoff sites distributed among 35 states. Of the 53 million Mg of recyclables recovered (from 28 states reporting), about 14 percent (7.1 million Mg) were yard trimmings, which were primarily composted. Separated paper products, plastics, glass, metals, and commingled materials account for the majority of the remaining mass. In Florida, where segregated curbside collection is provided, 60 to 80 percent of all yard trimmings generated are collected. However, only 10 to 25 percent of the total generated volumes of yard trimmings are routed to dropoff sites when an alternative collection program is used. It is estimated that in 1997, about 35.5 million Mg of yard trimmings will be generated in the United States and about 9 million Mg will be composted (Kashmanian, 1993). Clearly, the United States is on a path of increasing recycling.

Biosolids

The U.S. Environmental Protection Agency (1990b) estimated that about 7.7 million Mg (29.1 kg per capita) of biosolids are generated in the United States annually; the projection for the year 2000 was 15.4 million Mg (U.S. Environmental Protection Agency 1989b). With the expected growth in population, technological improvements in biosolids treatment plant operations, the Federal restrictions on ocean dumping, and the 40 CFR 503 rules for land application of biosolids containing high concentrations of metals and toxic organic chemicals (U.S. Environmental Protection Agency 1993), biosolids production and concurrent disposal needs will also increase.

Landfills in the United States simply will not accommodate the expected high-volume input (U.S. Environmental Protection Agency 1990b). Unless the United States develops and implements the necessary technology to reduce its heavy reliance on landfilling and incineration, the Nation will not meet the U.S. Environmental Protection Agency's outlined goals and objectives for improving environmental (soil, water, and air) quality (U.S. Environmental Protection Agency 1989a). Equally significant to these goals is the urgent need to implement programs based on the sustainable growth and development vision that involves all sectors of the Nation's economy. One of the clearly critical elements of this vision requires development of economic recovery methods and appropriate reuse of resources through recycling. Our national obligation to conserve and protect our natural resources can be partially fulfilled through biosolids and organics recycling.

Improving management and beneficial use of municipal solids

The national recycling goal, which encompasses resource recovery, reuse, recycling, and reduction of landfill volume, called for an increase in recycling of MSW by 25 percent by 1992 (U.S. Environmental Protection Agency 1989c). Parr and Hornick (1992) estimated that meeting this goal would result in a 55-percent decrease in the amount of MSW placed in landfills and a 20-percent increase in the amount incinerated. To achieve this goal, some municipalities have implemented source separation and collection programs to recycle paper, metals, glass, and plastics and to collect and compost yard trimmings. Jurisdictional prohibitions against landfilling yard materials have helped create municipally operated or contracted collection and composting operations, which produce compost that is used in a wide variety of horticultural and landscape situations. In addition some localities have encouraged backyard composting of yard trimmings and some other residues and have sought guidance from private foundations experienced with small-scale composting techniques (Rodale Press 1982). Other communities have selected the alkaline stabilization process to transform biosolids into a low-analysis mixture of organic fertilizer and agricultural limestone (Logan and Burnham 1995).

Composting

Composting is a time-honored practice used to convert organic residues into useful soil conditioners and biofertilizers. The practice is viewed as a viable and important means of stabilizing and transforming municipal solid residues for safe and beneficial use in agricultural, horticultural, and forestry operations (U.S. Environmental Protection Agency 1989a–c, U.S. House of Representatives 1990, Parr and Hornick 1992). Several of the problems (for example, malodors, human pathogens, and undesirable chemical and physical properties) that occur when raw and unstable organic materials are directly applied to soil can be resolved by composting.

Composting is a self-heating microbiological process in which the decomposition of organic materials is accelerated by the growth and enzymatic activity of mixed populations of bacteria and fungi (Miller 1991). Composting can occur aerobically or anaerobically (Gotaas 1956), but the aerobic mode is preferable because it minimizes the production of malodors, speeds decomposition, and produces high temperatures necessary to thoroughly and rapidly destroy pathogens and to dry the mixture.

One of three possible process styles—windrowing, static aerated piles, or in-vessel (Haug 1980)—are typically used at composting facilities concerned with high throughput of feedstocks. Both the static aerated pile and the in-vessel approaches offer more control over critical process parameters than does windrowing. The latter is often established by communities because it is far simpler to operate and has lower capital requirements than either of the other two (Reinhart et al. 1993). Such features are particularly appealing to farmers who are interested in starting on-farm operations using existing equipment with few additional capital expenditures. Windrow composting was the second most common style of composting reported in a recent survey, with 78 projects of 281 using this type of composting (Goldstein and Steuteville 1995). In-vessel composting was operating at 66 of the 281 projects in that same survey.

The Beltsville Aerated Pile Method (static aerated pile) was developed to rapidly compost biosolids and has been readily adopted by more than 111 U.S. cities and municipalities (Parr and Willson 1980, Willson et al. 1980, Goldstein and Steuteville 1995). When static aerated pile composting is operated using a temperature feedback control system, the composting materials are rapidly dried though the process of evaporative cooling.

The U.S. Environmental Protection Agency (1993a) established rules based on cumulative pollutant loading rates for application of biosolids onto agricultural and nonagricultural land. These rules apply to any materials containing biosolids regardless of their treatment (composting, chemical fixation, or digestion) or blending with other substances after production.

Alkaline stabilization

Pathogen destruction and organic matter stabilization are critical outcomes that must be achieved by good management of biosolids. In the case of composting, the destructive heat, ammonia, and consequent drying are generated by biological (microbial) activities, whereas purely chemical, exothermic reactions generate the destructive pH, heat, and drying that occur with alkaline stabilization processes.

Appropriate technology for blending alkaline byproducts (ABs) with biosolids, or with mixtures of organic materials containing biosolids, was developed in the 1980's and has been used as an alternative to composting. Logan and Burnham (1995) have described the process principles and product uses for one of the commercial processes that has successfully applied the technology to biosolids. They noted that ABs that have a large content of free lime (such as cement kiln dust, lime kiln dust, and coal combustion ash), can be mixed at a rate of up to 25 to 50 percent (wet weight basis) of dewatered biosolids. Such mix ratios can raise the pH of the resulting product to 12 or greater. The fine particle size and low moisture status of ABs contribute significantly to successful stabilization of raw primary, waste-activated, or digested biosolids, with total solids ranging from 18 to 40 percent (wet weight dewatered biosolids basis). When the blending speeds (pug mills or screw presses) are adjusted to accommodate a selected mixing ratio, the resultant product is a soil-like, granular material that can be processed further in either of two ways to assure thorough destruction of pathogens and organic matter stabilization and to increase solids content to 65 percent by weight.

Value of Recovered Organics as Biofertilizers and Soil Conditioners

Parr and Hornick (1992) have delineated the essential factors involved in assessing the value of organic residues. They stated that evaluation can be approached in terms of fertilizer equivalency, capacity to alter soil physical properties, and agronomic impact on crop yield and quality. The most direct method of evaluating organic residues is to determine crop yield from those residues and the current economic (market) value of the plant nutrients found in the product, especially of N, P, and K (table 2 lists the value of some organic residues). In some cropping situations, the secondary plant nutrients (S, Fe, Mg), micronutrients (Cu, B, Zn, Mn, Mo), and lime equivalency values also need to be assessed. The value of some materials (such as soil conditioners and biofertilizers) could be negative if they contain high amounts of soluble salts, heavy metals, or hazardous organic chemicals or have high C:N ratios or extreme pH values (Parr et al. 1983).

Table 2. Value of some organic residues based on their macronutrient content

Organic residue	Nutrients (%)			Value*
	N	P	K	($ Mg^{-1})
Cattle manure	4.4	1.1	2.4	23.47
Crop residues	1.1	0.2	2.0	8.44
Biosolids	4.0	2.0	0.4	21.40
Municipal solids	0.7	0.2	0.3	3.66

*Value Mg^{-1} of N, P, and K was set at $0.30, $0.37, and $0.20, respectively, based on average dealer prices of fertilizers at midwest terminal locations in December 1990.

Sources: Parr and Colacicco (1987), U.S. Department of Agriculture (1978), Parr and Hornick (1992). Reprinted from Parr and Hornick (1992, p. 552) courtesy of Marcel Dekker.

The soil conditioning value of organic wastes is profound on marginal or severely eroded lands that were reclaimed through the application of composted biosolids and feedlot manure (Hornick 1982, Hornick and Parr 1987). The economic value (that is, the fertilizer, lime, and secondary and micronutrient equivalency values) of organic residues are more easily assessed than the soil conditioning value.

In addition to the nutrient equivalency and soil conditioning values, the agronomic value (crop yield or crop quality) is used to determine the benefit of adding a particular material (Parr and Hornick 1992). Whereas there is substantial evidence of a positive effect on crop yield, there are very few reliable experimental evaluations of the effects on crop quality. The yield response to organic wastes is generally nonlinear and, at present, unpredictable because the interactions and interdependency of crop, soil type, climatic factors, soil and crop management practices, and properties of the residue material are incompletely understood. Crop yields tend to follow the law of diminishing returns—the greatest yields result from application of the first several increments of material, and gradually the yield increases level off with subsequent additions (table 3). Thus, the highest agronomic value per unit of organic material occurs at lower application rates.

Because the nutrient content of most municipal, industrial, or rural organic residues is generally low (Parr and Colacicco 1987), adding small amounts of synthetic fertilizers to them may increase their agronomic value. Generally, the net profit attributable to the use of organic amendments will depend on the properties of the material, the cost of transportation

Table 3. Effects of single applications of biosolids applied to soil in 1972 and an annual application of chemical fertilizer on corn grain yields

	Corn grain yields (hundreds of kg ha^{-1})					
Treatment*	**1972**	**1973**	**1974**	**1975**	**1976**	**Average**
Biosolids						
0 dry Mg ha^{-1}	25.1	21.1	6.8	16.2	0.91	4.0
56 dry Mg ha^{-1}	55.4	68.7	64.6	54.4	38.5	56.3
112 dry Mg ha^{-1}	66.5	102.8	68.7	66.0	67.1	74.4
224 dry Mg ha^{-1}	61.2	108.4	76.5	69.4	69.2	76.9
N–P–K fertilizer†	34.3	51.0	57.6	50.1	47.7	48.2

* Biosolids were applied at the rates indicated (Mg ha^{-1}, dry weight basis) in 1972 only. No inorganic fertilizer was applied.

† Fertilizer plots received 180 kg ha^{-1} of N, 40 kg ha^{-1} of P, and 75 kg ha^{-1} of K each year of the study; no biosolids were applied.

Sources: Barbarika et al. (1980), Decker et al. (1977). Reprinted from Parr and Hornick (1992, p. 553) courtesy of Marcel Dekker.

and application, and the market value of the crop (Parr and Hornick 1992). Parr and Hornick (1992) noted that studies by Barbarika et al. (1980) and Decker et al. (1977) on corn show that the grain yields (and therefore financial gains) of a crop in subsequent years could be as high or higher than yields in the first year (table 3). This phenomenon is linked to the slow release of N and P from the decomposing organic amendments that become mineralized and available for plant uptake and growth over the long term.

Parr and Hornick (1992) calculated the value of biosolids at $10.28 per Mg, based on the fertilizer equivalency values in table 2 and a corn grain yield of 4,800 kg ha^{-1} but excluding hauling and spreading costs. Hyatt (1995) discussed the need for a “net present value” method for calculating the financial advantage of long-term compost application versus several other alternatives. He presented a model (Hyatt 1995) designed to consider the economic value of compost’s residual N (that is, N available after the year in which it was applied). The model showed that compost gave a total net return of $14.60 ha^{-1} greater than chemical N, given the carefully considered input values used for calculations in the model.

Soil productivity is affected by various factors that may degrade or improve soil properties (fig. 2). Regular recycling of organic materials on the farm such as animal manures and crop residues will improve the tilth, fertility, and productivity of agricultural soils by protecting them from wind and water erosion and preventing nutrient losses from runoff and leaching. In some agricultural situations, limited supplies of good-quality organic materials are available on the farm to provide adequate soil and water conservation (U.S. Department of Agriculture 1978). In such cases, composts produced from biosolids and biodegradable fractions of MSW or alkaline stabilized biosolids or manures could be used to improve soil productivity.

Composts are more stable and easier to handle, store, transport, and apply than noncomposted organic residues. Parr and Hornick (1992) characterized noncomposted biosolids as organic material that has a high nutrient availability index (NAI) and that decomposes and mineralizes rapidly in soils. This decomposition releases significant amounts of N and P for plant uptake. In contrast, they characterized composted biosolids as an organic material with a high organic stability index (OSI), that is, a slow rate of decomposition in soil and a slow rate of nutrient release. In general the NAI value of an organic material is inversely related to its OSI value. Thus, composted or co-composted biosolids and MSW generally can be expected to have a greater inherent value as soil conditioners than as rapidly available sources of plant nutrients. Such materials may serve as fertilizer supplements but not usually as sole nutrient sources.

Parr and Hornick (1992) noted that a major part of municipal organic residues could be used beneficially

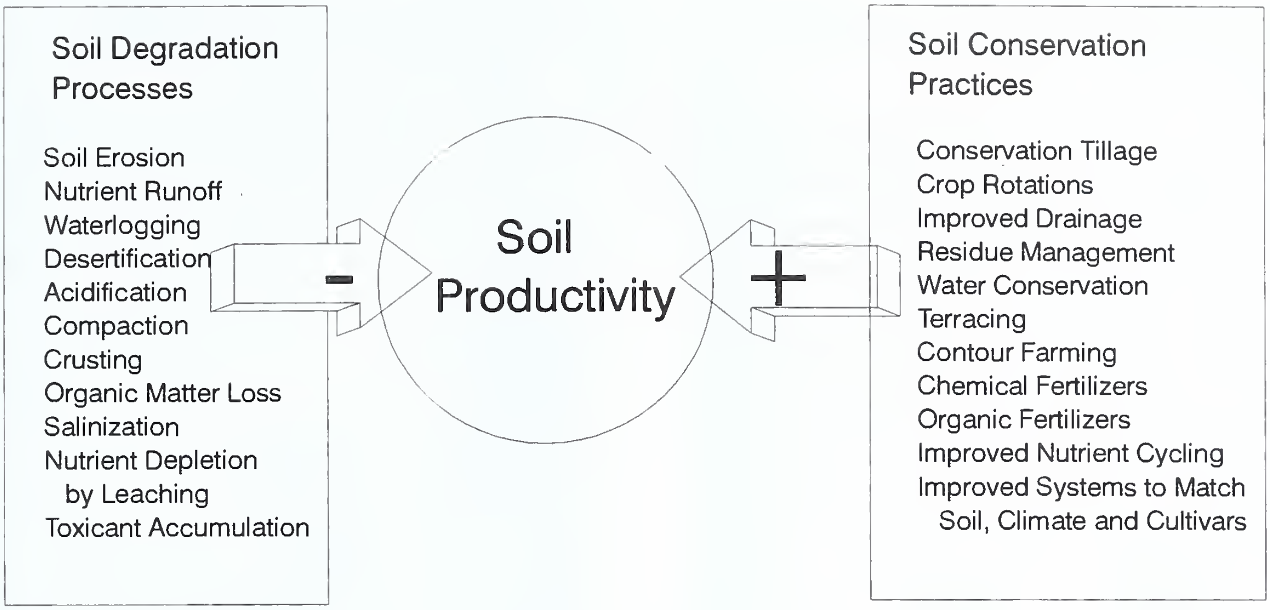

Figure 2. Positive and negative factors influencing soil productivity

on agricultural lands, especially on highly erodible land such as that in the Conservation Reserve Program (CRP). They estimated that about half of the 14 million ha in the CRP that have been set aside and planted primarily with perennial grasses could be returned to crop production. They proposed that waste application integrated with crop rotation systems be developed to prevent soil movement on such highly erodible lands. Research supporting this use of organic residues would be highly beneficial to a vast amount of agricultural soils that are at risk of degradation.

Although the potential for agricultural use of composted, co-composted, alkaline-stabilized, or uncomposted municipal materials is very large, currently they have not been used on most agricultural soils because the quality of the finished products is unpredictable (though in only a few cases have they been shown to be detrimental to plant establishment and growth). Future development and marketability of the products will require appropriate technology for reliable production and quality-assurance testing. Soil conditioners and biofertilizers produced must be of high quality and must be economical to distribute and apply. Development of such products will require some cooperation among urban, rural, scientific, and economic communities to solve problems associated with the costs of processing, transporting, and applying municipal residues. Researchers (Kashmanian et al. 1990, Parr and Hornick 1992) agree that very large quantities of municipal residues can and should be used on agricultural soils in the future in an effort to improve their productivity and quality. Research and solid interpretive analyses are needed to ensure that municipal residue recycling on land is safe, reliable, and beneficial, as well as economical to both the urban and agricultural sectors.

Costs of Collecting, Processing, Transporting, and Applying MSW

Continually increasing production of MSW in the United States and concurrent reduction in the number of landfills in operation, coupled with environmental pressures for a 25-percent reduction in landfilling (Gibson 1991), have strengthened the need for alternative methods for handling MSW. Costs of handling MSW have skyrocketed, controls have tightened, and technologies are slow and costly to develop. Pressures to meet Federal regulation of MSW facilities (pursuant to Subtitle D of the 1976 Resource Conservation and Recovery Act) through state and local government implementation are compounded by increased tipping fees at reprocessing and treatment facilities and public opposition to siting these facilities. Local governments historically have expended more on solid waste than have Federal or state governments because local governments bear primary responsibility for provision of services to households. In 1972 and 1987, local solid waste costs were \$3.5 and \$6.3 billion, respectively. Total costs are projected to reach about \$9.5 billion by the year 2000 (U.S. Environmental Protection Agency 1991). These problems and the cost of solving these problems encourage the creation of competitive uses for residues.

Tipping fees at United States landfills have increased to an average of about $29.50 per Mg, and the more densely populated areas have much higher fees (fig. 3). For example, in the Northeast, the average tipping fee is $51.51 per Mg. Connecticut has the most expensive fee at $71.50 per Mg. The Rocky Mountain states and the Midwest have the lowest fees. Compounding the problem of higher tipping fees is a reduction in usable landfill space. While some facilities have been closed because they failed to meet regulations, others are closing because they are full. In the United States, there were 7,924 operating landfills in 1988 but only 3,197 by 1995 (fig.4).

Costs for all types of processing systems are rising. In 1988, there were 115 U.S. composting facilities processing approximately 1,450 Mg of dry biosolids daily. These facilities were constructed at a capital expenditure in excess of $1 billion and have operating costs estimated at $100 million (Finstein 1989). A study of eight in-vessel composting systems revealed that each of the eight was using sawdust as an amendment. The cost of the sawdust was $3.92 to $18.64 per m^3 of sawdust. Of these eight systems, the cost per dry Mg of biosolids processed ranged from $110 to $418 (Johnston et al. 1989). In another study, 10 facilities were evaluated for capital and operational costs. Capital costs ranged from $250,000 to $78 million, and operational costs from $240,000 to $30 million (Curtis et al. 1992).

Yard trimmings have been identified in a majority of jurisdictions as the most readily and obviously compostable materials that need to be diverted from landfills to the recyclables market. Many communities now have or are planning to establish facilities for leaf and grass composting. Operating and capital costs of composting sites are important considerations for communities planning such recycling programs.

In an effort to aid planners, a comparative costs study was conducted at several composting facilities that represented a broad range of technological requirements and three levels of handling capacities (Renkow et al. 1994). Results showed that facilities handling 10 to 25 Mg yr^{-1} and turning piles only 2 to 3 times per year with a front-end loader on a packed clay surface are clearly the least costly to establish ($183,000 to $254,000) and operate ($49,000 to $66,000 yr^{-1}). However, the product from these facilities will likely be unscreened and of poor quality unless extra measures are taken to refine the material after composting. Furthermore, it was estimated that the process would take 36 mo, and the volume of the materials processed would be reduced only 30 percent. In comparison, facilities that are paved with asphalt or concrete and on which piles are turned once a month will have startup costs ranging from $280,000 to $440,000 when they process 10,000 Mg yr^{-1} and will have startup costs of $446,000 to $646,000 when they process 25,000 Mg yr^{-1}. Operating costs for these low-technology, paved sites range from $77,000 to $98,000 when they process 10,000 Mg yr^{-1} and $138,000 to $190,000 when they process 25,000 Mg yr^{-1}. With such increased costs, the product can be expected to be of low-to-moderate quality (unscreened), and the volume will be reduced 40 percent in 12 mo. Use of more sophisticated equipment, such as compost turners and screeners, along with a more frequent turning schedule will result in a consistently better quality product and a 50 to 55 percent reduction in volume within a shorter time (6 mo). From this comparative study, it is clear that lower cost methods of paving sites need to be explored so that startup and operating costs can be reduced without having to sacrifice the benefits of a stabilized work surface. Also, the use of alkaline byproducts as surface stabilizers should be investigated.

The city of Scranton, PA, built a biosolids composting facility with a processing capacity of 23 dry Mg of biosolids (aerated static pile method). The cost of construction was $3,338,000, and operational and maintenance costs were calculated to be $18.81 per wet Mg of biosolids processed (Elliott and Polidori 1988).

Agronomic Uses of Untreated and Treated Municipal Residues

Nutrient properties of treated and untreated residues

Municipal residues have variable nutrient values (or fertilizer values) depending on their source and treatment. Untreated residues such as raw biosolids may be similar to animal manures, which have a relatively high NAI and high N and P contents. MSW's are generally low in nutrients because they contain considerable paper and yard waste. Processes such as digestion or composting result in the loss of

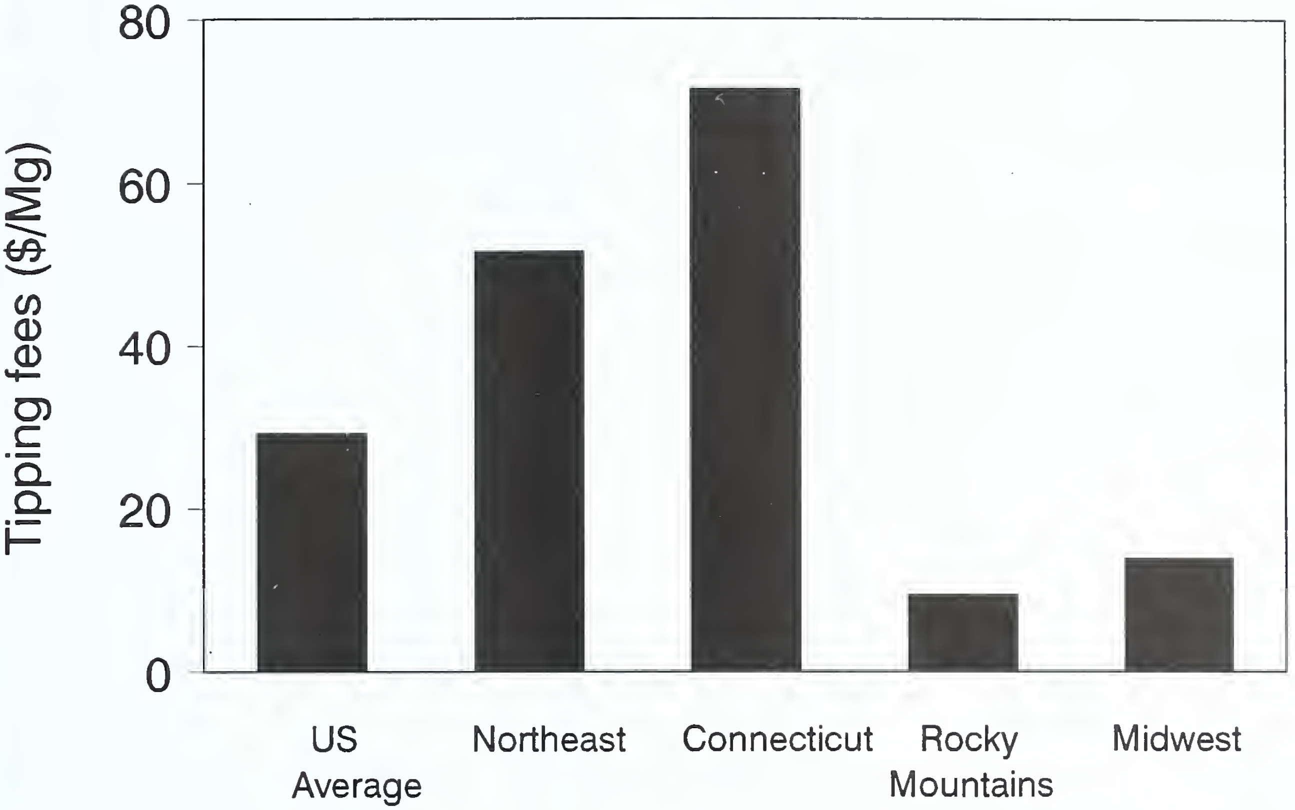

Figure 3. Average landfill tipping fees

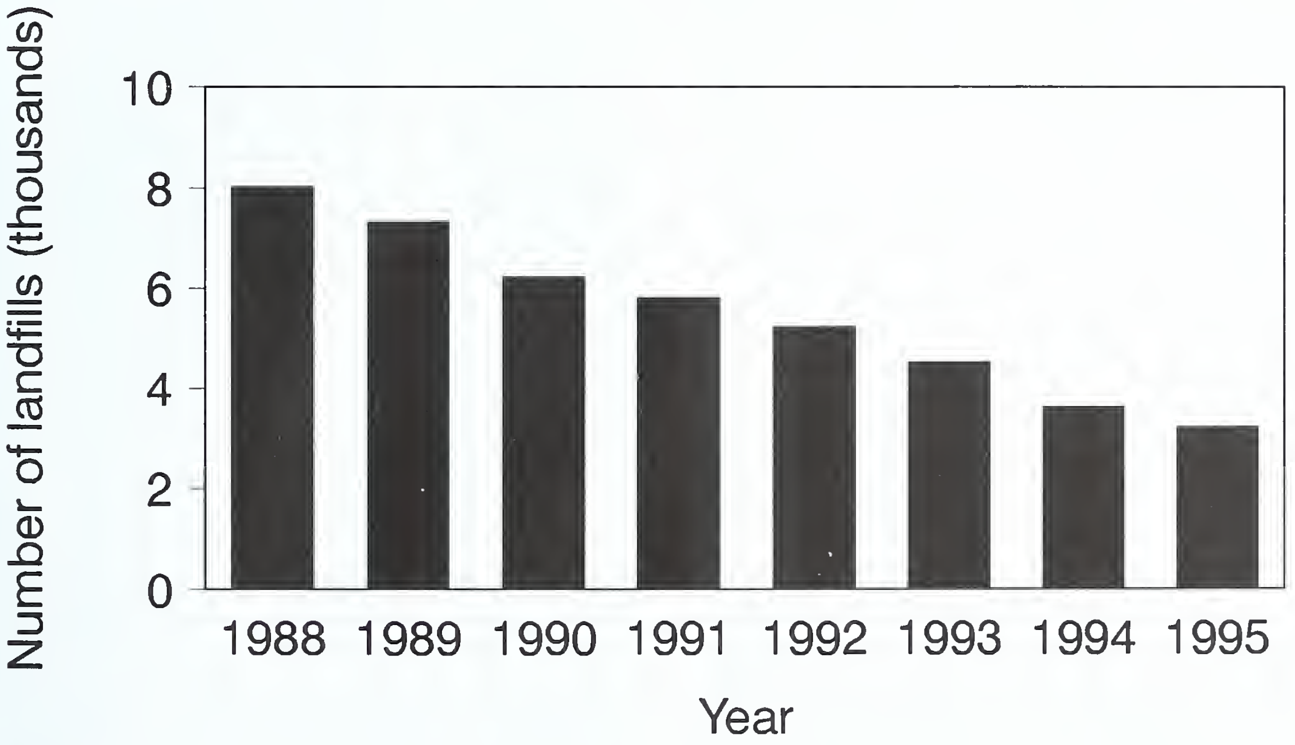

Figure 4. Number of landfills in operation in the United States

organic matter through decomposition and will (1) increase concentrations of "conserved" (slightly soluble and nonvolatile) nutrients such as P and trace metals, (2) decrease ammonia-N by volatilization, and (3) decrease K by leaching.

When biosolids are added to soil, the type of soil, the methods used to produce the biosolids, and the C:N ratios of the biosolids and the soil affect the mineralization of the biosolids N (Parker and Sommers 1983, Barbarika et al. 1985, Douglas and Magdoff 1991), P (Soon and Bates 1982, McLaughlin 1984, McCoy et al. 1986) and S (Taylor et al. 1978, Tabatabai and Chae 1991). O'Keefe et al. (1986) and Douglas and Magdoff (1991) demonstrated that mineralization rates ranged from nearly zero to 60 percent of the organic N added; in general, the more extensive the biological treatment or degradation of the residues, the lower the N-mineralization potential of the product. Composted biosolids usually have a mineralization rate of about 10 percent or less (Tester et al. 1977, Haan 1981, Douglas and Magdoff 1991).

The effect of a composted product on crop performance is determined by the product's maturity and mineralization rate in soil. A biogenic residue compost can provide 60 to 90 kg of available N ha^{-1} at an application rate of 54.5 Mg ha^{-1} (Vogtmann and Fricke 1989), and an MSW compost can provide 90 kg N ha^{-1} at a rate of 25.4 Mg ha^{-1} (Mays et al. 1973). Evaluation of the fertilizer value of a compost must include an analysis of the nutrient element and organic matter contents, mineralization rates, and C:N and C:P ratios.

Because composts have low levels of nutrients, they are often considered more valuable as sources of organic matter; but when used in large quantities, composts are sources of slow-release nutrients. Hortenstine and Rothwell (1972) showed that an application of 63.6 Mg ha^{-1} of MSW compost to phosphate-mining sand tailings increased the soil content of extractable K, Ca, and Mg. Bengtson and Cornette (1973) found that an application of 40 Mg ha^{-1} of MSW compost increased the concentration of exchangeable Ca in the soil after 28 mo. Comparisons of MSW compost with mineral fertilizer showed that compost additions did not maintain sufficient levels of available P in the soil (Cabrera et al. 1989), but these additions did increase the K level in the soil. Additions of an MSW compost on a sandy loam and a silty clay loam initially increased the percentage of organic C in the soils by 100 and 34 percent, respectively (Giusquiani et al. 1988).

Applications of 531 Mg ha^{-1} of a compost mixture of MSW and biosolids resulted in a 66-percent increase in the carbon content of the soil over 3 yr (Zan et al. 1987a). Soil N increased from an average concentration of 0.12 to 0.15 percent, but the C:N ratio of the soil did not change. P levels increased with the addition of compost. Applications of 9, 18, and 36 Mg ha^{-1} of the composted mixture of MSW and biosolids over a 24-yr period increased the total C and N content and the N-mineralization potential of the soil (Werner et al. 1988). The higher total N content was a result of higher hydrolyzable as well as nonhydrolyzable organic N compounds. The level of extractable P and S in a soil amended with composted biosolids is determined by the rate of P and S immobilization due to reactions with Fe and Al and by the rate of P and S mineralization from microbial action (Taylor et al. 1978). Phosphorus levels in soils can also be affected by organic residues containing septic tank effluent, which is high in P and results in high P loading (Sikora and Corey 1976).

Ticknor and Hemphill (1990) found that a wide range of herbaceous and woody plants can be grown in undiluted composted yard trimmings (N–P–K ratio of 0–0.1–0.25). Supplemental fertilizer was necessary and additions of bark or pumice were beneficial for optimum plant growth. Vegetable market refuse composted with slaughterhouse waste increased yields of sunflower as application rates were increased (Marchesini et al. 1988); the compost additions improved levels of several soil fertility factors. Biogenic composts, which are made from separately collected food and yard residues, have a relatively high fertilizer value. Vogtmann and Fricke (1989) showed that with additions of biogenic composts and N–P–K fertilizer to soils, crop yields of kohlrabi increased.

MSW compost (pH of 8, with a relatively low C:N ratio of 17, and N–P–K values of 1.67–0.55–0.40) produced a positive growth response in an acidic soil (Wong and Chu 1985). Peach trees treated with MSW compost applications showed greater growth than untreated controls but had lower average fruit weights (Strabbioli and Angeloni 1987). The average yield per tree, however, was not significantly different. The N–P–K ratio for the compost was 1.4–0.15–0.2. Heavy applications of MSW compost (102, 204, or 408 Mg ha^{-1}) on corn caused poor growth initially, but had a significantly delayed fertilizing effect (Stone and Wiles 1975). At 408 Mg ha^{-1}, normal growth was delayed 7 mo, but at the end of the experiment these treatments had favorable yields when compared to the yields from inorganic fertilizer treatments of 89.6 and 179.2 kg N ha^{-1}. An application of mature MSW compost at a much lower rate of 9 dry Mg ha^{-1} resulted in more growth of *Brassica rapa* var. *perviridis* compared to growth from an equal application of nutrients in the form of mineral fertilizer (Chanyasak et al. 1983).

Compost applied in conjunction with mineral fertilizers increased yields compared to those from fertilizer or compost alone. Yield of corn grain was increased from 3–4 Mg ha^{-1} for fertilizer alone to 10 Mg ha^{-1} for compost with fertilizer (Wang 1977). Grass mixture yields increased from 14.0 g pot^{-1} with N–P–K fertilizer to 24.8 g pot^{-1} with a mixture of Dano compost and fertilizer (Kropisz and Kalinska 1983). The dry weight of sorghum seed heads increased from 298 kg ha^{-1} with N–P–K applications and 206 kg ha^{-1} with compost applications to 618 kg ha^{-1} with applications of both (Hortenstine and Rothwell 1972). Oat forage yields (dry weight) increased from 2,144 kg ha^{-1} with N–P–K applications and 1,405 kg ha^{-1} with compost applications to 3,860 kg ha^{-1} with an application of a mixture (Hortenstine and Rothwell 1972). Corn treated with either 179.2 kg ha^{-1} of fertilizer N or 16 Mg ha^{-1} compost yielded 187.8 bushels ha^{-1} but yielded 222.4 bushels ha^{-1} when treated with both (Hortenstine and Rothwell 1972, Kropiscz and Kalinska 1983).

Hortenstine and Rothwell (1972) and Kropisz and Kalinska (1983) concluded from their results on grasses that physical changes (improvements in nutrient release, water, and soil aeration properties) in compost-amended soil allow more efficient use of mineral fertilizer. The increased yields when compost and inorganic N are combined are thought to result from a synergistic effect of the compost and inorganic N. They believe this to be true because, in the case of their corn studies, 16.3 Mg ha^{-1} of compost probably would not supply enough N, or the added organic matter does not increase the water storage capacity of soil enough to account for the yield increase obtained.

It is unclear which nutrient source—fertilizer or compost—generates more plant growth by itself. Fertilizer sometimes generates more growth than compost alone because compost is often comparatively low in nutrients. For example, fertilizer increased dry weights of *Brassica chinensis* by 190 to 1,000 percent and of *Lycopersicum esculentum* by 18 to 190 percent over dry weight produced at varying compost rates (Chu and Wong 1987). However, the fruit of *L. esculentum* showed variable growth responses to fertilizers and composts; some compost applications produced better yields than were obtained with fertilizer. Carrots amended with 45.4 to 113.5 Mg ha^{-1} of compost yielded larger roots and foliage than fertilized carrots, and sorghum amended with 58.1 Mg ha^{-1} of compost attained greater height than fertilized sorghum (Hortenstine and Rothwell 1973). Although N deficiencies were noted in corn treated with unscreened MSW compost applied at rates of 8, 18, 102, 204, and 408 Mg ha^{-1} without fertilizer, plots treated with 89.6 kg ha^{-1} of inorganic N or 204 or more Mg ha^{-1} of compost yielded more (188 bushels ha^{-1}) than untreated plots (136 bushels ha^{-1}) (Stone and Wiles 1975). Dry sorghum forage yields increased from 10 Mg ha^{-1} without compost or inorganic N to 15.4 Mg ha^{-1} with 173 Mg ha^{-1} compost and to 16.3 Mg ha^{-1} with 179 kg ha^{-1} inorganic N. Cabrera et al. (1989) found no noticeable growth differences from treatments consisting of a small application of MSW compost (12.7 Mg ha^{-1}), a fertilizer treatment (500 kg ha^{-1} of 15–15-15), or a mixture of the two. Highest yields of field and adzula beans were attained with compost-fertilizer combinations (Robinson 1983). The mixtures of compost and fertilizer, however, did not result in a positive synergistic response in all cases.

Applications of co-composted MSW and biosolids have been at least as successful as MSW applications alone. Co-compost applications resulted in a 1.7-fold increase in the total stem biomass of slash pine compared to the biomass yield of control plots (Jokela et al. 1990) and resulted in a greater yield of maize compared to the yield from MSW compost alone (Zan et al. 1987b). The yields of sorghum, common bermudagrass, and corn responded in a positive manner when the crops were amended with annual applications of 130, 72.6, and 101.7 Mg ha^{-1} of co-composted MSW/biosolids (N–P–K of 1.3–0.30–0.91), respectively. The yields were surpassed by applications of N at the rate of 180 kg ha^{-1} together with adequate P and K (Mays et al. 1973). Applications of co-composted MSW and biosolids produced higher forage yields of corn than were obtained with the untreated control (Giordano et al. 1975).

Songmuang et al. (1985) showed that long-term applications of compost (at 12 Mg ha^{-1}) made from rice hulls and manure eventually led to a buildup of organic matter in the soil and that this buildup and the subsequent compost applications could replace the fertilizer application. Similarly, Brinton (1985) hypothesized that long-term use of composted manure could eventually lead to a buildup of organic matter that would supply all of the N needs of the plant. In the 1st yr of Brinton's study, the mineralization rate of composted manure was 9 percent of the total N added.

Application of MSW results in accumulations of trace metals. Vineyards that received several applications of MSW had 2 to 20 times the amount of metals in the soils (Furrer and Gupta 1983). Jokela et al. (1990) studied a slash pine plantation that had been treated with three rates of garbage compost 16 yr earlier; significant but modest treatment effects were associated with increases in concentrations of N, P, B, Fe, Al, and Zn in pine tissues. Examples of successful uses of organic residues in soils are summarized in table 4.

Nonnutrient properties of treated and untreated residues

Addition of biosolids or compost to soil almost always improves the physical properties. However, with certain types of composts, such as biosolids, it may take much more of the compost to affect physical properties of the soil than it does to provide the necessary nutrients for plants. Chang et al. (1983) reported that more than 72.6 Mg ha^{-1} of biosolids compost was necessary to significantly affect the physical properties of soil, that is, aggregate stability, bulk density, porosity, organic matter content, and moisture holding capacity.

Soil aggregate size and stability are affected by the physical, chemical, and biological activities existing in the soil, especially the microbial decomposition of

Table 4. Examples of organic residue uses that result in adequate plant growth

Residue type	Application rate (Mg ha^{-1})	Crop	Source
Biosolids compost	69–122	Legumes and reed canary grass	Watkin and Winch (1974)
Biosolids compost	204	Spruce and pine trees	Gouin (1977)
Biosolids compost	134	Fescue	Tester (1989)
MSW compost	58	Sorghum	Hortenstine and Rothwell (1973)
MSW compost	45	Carrot	Chu and Wong (1987)
MSW compost	64	Bermudagrass	Wong and Chu (1985)
MSW/biosolids compost	224	Slash pine	Okela et al.(1990)
MSW compost	9	*B. rapa*	Chanyasak et al. (1983)
SW compost plus fertilizer	9	Corn	Wang (1977)
Biosolids plus fertilizer	4.5	Corn, winter wheat	White and Brown (1981)
Biosolids	134	Corn	King and Dunlop (1982)
MSW compost	91	Peas	Purves and Mackenzie (1973)

organic matter. Aggregate stability refers to the ability of soil aggregates to withstand disruptive influences such as water and pressure. Organic matter in the soil provides a substrate for microbial growth; the microorganisms in turn produce substances such as polysaccharides, which are necessary for aggregation. Biosolids increase stable aggregates (Epstein 1975) and water holding capacity (Khaleel et al. 1981) and decrease bulk density of soils (Khaleel et al. 1981). Although a wide range of application rates was used in these studies, no minimum application rate has been recommended for achieving specific minimum physical changes.

Many reports indicate that compost increases the aggregate stability of soil. MSW compost applied at 9 Mg ha^{-1} to a highly weathered clay-textured soil increased aggregate stability by 4 percent (Wang 1977). After 90 days, applications of 27.2 to 54.4 Mg ha^{-1} of MSW compost to a soil with low organic matter and high base saturation increased aggregate stability by 11 and 18 percent, respectively (Hernando et al. 1989). Soil carbon content after 90 and 180 days was correlated to the aggregate stability. MSW compost applied at 36.3 Mg ha^{-1} increased the water stability index in an alluvial soil after 3 yr, but had no effect on calcium-rich soil (Guidi and Poggio 1987).

MSW compost applied at 9 Mg ha^{-1} to a clay-textured latosol did not change the bulk density of the soil significantly (Wang 1977). An MSW compost applied at several rates caused no change in the total porosity or pore size distribution in either a Calcic Cambisol or a Eutric Fluvisol after 3 yr (Guidi and Poggio 1987). After 5 yr of amendments with a biosolids compost, a loamy sand soil had reduced penetration resistance and bulk density, increased soil water content and specific surface area, and a modified pH below the tillage depth (Tester 1990). Applications of mixed MSW and biosolids compost at rates of 45.4 and 136.2 Mg ha^{-1} increased the porosity of a sandy loam soil (Guidi et al. 1981, Pagliai et al. 1981) and increased total porosity significantly and soil aggregate stability slightly (Pagliai et al. 1981). An application of 63.6 Mg ha^{-1} of MSW compost on phosphate-mine sand tailings increased the organic matter content from 0.39 to 1.05 percent after one cycle of sorghum and oats (Hortenstine and Rothwell 1972). Applications of 27.2 Mg ha^{-1} of compost increased the soil organic matter content from its initial range of 1.6 to 2.1 to about 3.3 percent (Gabriels 1988). Duggan and Wiles (1976) showed similar results with compost additions of up to 181.6 Mg ha^{-1}. Elemental compositions, functional group contents, E_4/E_6 (absorbance at 400 nanometers to absorbance at 600 nm) ratios, and spectral characteristics were not useful for detecting differences in the structure of humic acids isolated from the soil before and after the addition of compost (Gonzalez-Vila and Martin 1985).

The term "moisture holding capacity" indicates the amount of water a soil can hold, while the term "moisture retention capacity" refers to the length of time a soil can retain water (Epstein et al. 1976). Both properties are greater in soils with large amounts of organic matter or clay particles (Einspahr and Fiscus 1984). Although these two factors together indicate how much water is in the soil, they do not necessarily indicate the availability of that water for plant use (Chang et al. 1983). Heavy applications of MSW resulted in decreased bulk density and compression strength and increased soil moisture content and moisture holding capacity (Mays et al. 1973). Addition of nonsegregated MSW increased aggregate stability, but the availability of moisture, as determined by relative soil moisture curves at 0.33 and 15 bar, was not increased (Webber 1978). Additions of composted and vermi-composted (worm-worked) biosolids increased the available soil moisture of a sandy soil from 10.6 to 54.4 and 31.6 percent, respectively (Einspahr and Fiscus 1984).

Applications of 9 Mg ha^{-1} of MSW compost increased the moisture holding capacity slightly at .33 bar (Wang 1977); the moisture content during a dry period was higher in the soil amended with compost. Hortenstine and Rothwell (1972) found that applications of 63.6 Mg ha^{-1} of MSW compost increased the moisture holding capacity after a 1-year rotation of sorghum and oats. Applications of 13.6, 27.2, and 54.4 Mg ha^{-1} of MSW compost increased the water holding capacity slightly throughout the 180-day monitoring process (Hernando et al. 1989), and the highest application rate produced the largest increase. An MSW compost application of 40 Mg ha^{-1} in a young slash pine plantation increased the soil moisture retention ability, especially during the first months after the application (Bengtson and Cornette 1973). In summary, depending on the type of residue and the application rate, improvement in several soil properties were generally recorded.

Spreadability of materials

Residue products applied to soils are most beneficial if they are uniform in composition and texture. MSW usually is highly variable in organics, glass, plastic, and metals; increased uniformity results from segregation of the inert materials. Grinding or composting of the organic fraction generally improves the uniformity and particle size distribution. Liquid or semisolid (<10 percent solids) residues are more uniform and can be sprayed onto the surface or injected below the surface of the soil. Below-surface applications greatly reduce the loss of nutrients by volatilization.

Solids can be (1) surface applied with little or no tillage, (2) surface applied and incorporated into the soil, (3) applied as a mulch, or (4) applied in a furrow or trough. The choice of application method depends on several factors, including the size and uniformity of the material, its nutrient content, the amount available, the crop being grown, and the application and tillage equipment available. Large-particle solid residues with low nutrient content may be best applied as a mulch where revegetation is necessary or prevention of wind or water erosion is important. Small-particle (<1 cm) materials are easily and uniformly spread with commercial fertilizer and manure spreaders. Experience with manures and biosolids indicates that nonuniform applications result in variable plant stands across the field. Preservation of nutrients is best accomplished by mixing materials with soil after application. Small particles are more easily mixed with soils, especially in areas where minimal disturbance is desired. Furrow or slot application results in less uniform mixing.

Application of solids in narrow trenches of less than 60 cm wide and 60 to 120 cm deep is generally considered a disposal operation rather than a treatment operation. This is because large amounts of material can be applied to small land areas. However, agronomic benefits can be obtained in areas where impervious boundaries exist in soil profiles. Deep trenches may penetrate the barriers and allow roots to penetrate into lower soil layers. If the trenches are filled with organic material, significant nutrient benefits will be obtained by roots.

Product quality standards

Product quality standards are related to the proposed use of the solid materials (table 5).

Techniques to reduce loss of nutrients

Losses of nutrients by volatilization or leaching of soluble components can reduce the value of a recycled material for both agricultural and horticultural uses. Losses of key soluble components such as K and nitrate can occur by leaching. Volatilization of ammonia from manures and biosolids is a critical problem. Prevention of losses by leaching during storage simply requires that materials be kept relatively unexposed to rainfall, that liquids not enter the material, and that excessive drainage be collected and recycled. Proper storage of products is required in lieu of soil application.

Volatilization of ammonia occurs when the pH of the medium is alkaline (8.0 or higher) or when the solubility of ammonia in the solution is exceeded. Ammonia is a product of N mineralization in organic residues. Volatilization of ammonia can be prevented if the pH is reduced below 7.0 and the accumulation of ammonia is reduced. Microorganisms use ammonia as a preferential N source, and, if microbial activity is maintained at a relatively high level, ammonia will be transformed into organic N as microbial biomass. Sikora and Sowers (1985) found that ammonia volatilization occurred during the first 10 days in the composting of lime-stabilized biosolids, but that only 10 percent of the total N was lost during this time. If microbial activity is inhibited (as when temperatures exceed 70 °C), more N would be lost. Addition of P to organic residues also reduces N loss possibly by stimulating decomposition of the materials.

Research needs

Determination of the heterogeneity and range in chemical and physical characteristics of the components in compostable residues is necessary so that maximal benefits can be achieved. The application of biosolids or composts to soil as a complete fertilizer is not a viable option. Many factors are involved in this conclusion, namely the amount of biosolids or compost necessary (demand) versus the amount available (supply), the restrictions on adding excess nutrients other than the macronutrients N and P, and concerns about accumulation of non-nutrient chemicals such as heavy metals. Because of these concerns, research on amendment combinations with mineral fertilizer is needed to address environmental, agronomic, and economic factors associated with the use of recyclable residues.

Table 5. Product quality standards for solid residues

Quality	Size uniformity	Analysis needed	Toxicity to plants	Inert materials allowed	Probable uses
Low	Uniformity and size analysis not necessary	Approximate analysis of nutrients	Nontoxic to hardy plants	2–5% inert material	Revegetation of barren or disturbed land
Medium	Particle size within range for uniform application	Sufficient analysis to make application to non-food-chain crop	Nontoxic to moderately sensitive crops	Percentage below that considered injurious or unsightly	Agriculture, silviculture, turf amendments
High	Particle size within guaranteed range of sieve size	Guaranteed analysis	Nontoxic to sensitive plants	Percentage below that considered injurious or unsightly	Potting mix, home gardens, all food-chain crops

The role of organic matter from municipal or industrial residues in maintaining or improving soil physical properties needs further clarification. Specifically, proper amendment rates need to be determined for increasing soil organic matter by 10 percent in 5 yr and for maintaining present organic matter levels in agricultural soils in which adequate amounts currently exist.

Agricultural and municipal residues can be mixed, but little data are available on the techniques, mixing ratios, benefits, and potential uses of mixtures. Procedures must be developed for blending organics and industrial byproducts to generate materials that are tailored to specific applications. The bioavailability of metal and organic contaminants in some of these byproducts needs to be reduced for the transformed products to be desirable.

Horticultural Uses of Untreated and Treated Residues

Products from the horticultural industry in the United States have an estimated value of approximately $9 billion annually; this figure includes the value of fruits, vegetables, flowers, ornamentals, and landscaping. In 1991 containerized plant production accounted for a $4.7-billion segment of the horticultural industry. Containerized growth media typically contain 60 to 70 percent organic substrata. The "ball and burlap" technique of harvesting trees and shrubs from nurseries results in the removal of approximately 227 Mg of topsoil per year. Together the containerized and field sectors of the horticultural industry have a continuing need to replenish the organic matter lost through normal production and sales activities. Presently, the industry relies on imported Canadian peat, domestic milled pine bark, and shredded hardwood bark. Familiarity with the performance characteristics and nutrient composition of these growth media encourages growers to continue using the same media unless it can be shown that new or alternative products have equivalent or superior properties.

Materials-reclamation managers need to be informed that recycled residues and byproducts represent a possible low-cost substitute for peat and bark. In order for managers to consider using these recycled materials, reclaimed organics need to have a dependable standard of quality in terms of pH, soluble salts, particle size, macroelement and microelement content, moisture, and pathogens. There has been much interest in the United States and Western Europe in finding an alternative organic substitute for peat and bark. Materials such as manure, MSW, biosolids, and leaves may all be acceptable substitutes after appropriate treatment and transformation.

Table 6. Compostable organic residues from various economic sectors

Economic sector	Residue type or source
Agriculture	Coco fibers, cork, cotton seed hulls, poultry carcasses, rice husks, rice straw
Aquaculture	Shellfish
Food	Chaff, coffee, tea, food flavor production, fruit culls and processing residues, hop processing
Paper industry	Bark, sludge
Pharmaceutical	Spent fermentation liquids or biomass
Textile	Cotton wastes, flax residues, wool residues
Town residues	Biosolids, garbage, gardens, grass clippings, leaves, night soil, restaurant and supermarket discards, wood chips
Wood industry	Bark, sawdust

Research has shown that untreated byproduct materials are often unsuitable for horticultural applications because of phytotoxicity, N immobilization, high salt content, or structural incompatibility (Verdonck 1988). A variety of compostable organic residues and byproducts from different industries have potential uses in horticultural media (table 6). Composts often compare favorably with peat as a major component of horticultural potting media (Bugbee and Frink 1989) (table 7), assuming that the quality control of the compost product is high enough to meet the required horticultural-grade criteria. Proper composting stabilizes organic residuals, reduces their water content for transport and storage, improves structural characteristics of the product compared to the raw stocks, and eliminates certain phytotoxicity problems (Hornick et al. 1984).

Of the organic materials used in potting media, at least 40 percent could be supplied from sources other than peat or pine bark (Sanderson 1980, Conover and Poole 1983, Fitzpatrick and Carter 1983, Marcotrigiano et al. 1985). Composted organic residues, such as sawdust, biosolids, and residues from fruit processing and pharmaceutical and food flavor production are all acceptable in some form for incorporation into potting media for the culture of nursery and greenhouse plants (Bugbee and Frink 1989). Properly managed composts can reduce the need for fertilizer (Falahi-Ardakani et al. 1987). Experience has shown that containerized plants grow as well in mixes containing up to 50 percent composted MSW, 30 percent composted biosolids, or 50 percent each of composted biosolids and MSW as in standard mixes, provided that the composts are mature and stable.

Phytotoxicity and growth-rate suppression, however, result when the MSW content exceeds 50 percent. These problems have been attributed to high content of soluble salts (Conover and Joiner 1966, Sanderson 1980, Lumis and Johnson 1982), boron toxicity (Lumis and Johnson 1982), poor aeration (Sanderson 1980), and heavy metal toxicity (Chu and Wong 1987). Phytotoxicity is associated with the presence of volatile fatty acids such as formate, butyrate, and propionate, and phenolic compounds, but is also dependent on the species and age of plant. For example, MSW levels above 50 percent inhibit early root growth in cress and tomato but not in ryegrass, pansies, salvia, or wallflowers (Keeling et al. 1991).

An active area of composting research has been in using wood-processing wastes in horticultural mixtures. Uncomposted bark, sawdust, and shavings of coniferous trees can be used to improve the physical qualities of horticultural mixtures. Up to 80 percent by volume of bark composts can be used in potting mixes or as substitutes for peat in growing many vegetable and ornamental plants (Pudelski 1983 and 1985). Short-term composting (for example, 3 wk to 3 mo) with the addition of nitrogenous materials (for example, sludge from soy scraps) (1) inactivates phytotoxic substances that may be present in the raw material, (2) corrects the C:N ratio and thus counteracts sorption of mineral N, and (3) initiates humification—a process that aids in the water holding capacity of the mixture. The mixtures require the addition of slow-release fertilizers (for example, sulfur-coated urea) containing necessary microelements and macroelements.

Bark, sawdust, and wood shavings from hardwood species require longer composting times (for example, 6 to 9 mo) because they contain greater amounts of phytotoxic substances requiring degradation. Beech

Table 7. Comparison of the important horticultural properties of peat, composts (prepared from source-separated organic residues such as wood, leaves, and grass clippings), and the ideal horticultural growing media

	Properties of:		
Quality criteria	**Peat**	**Composts**	**Ideal growing media**
Impurities (plastic, rubber, glass, stone, etc.)	None	Low (very low if source-separated and screened)	No sharp pieces or articles >2 mm
Growth-inhibiting substances	None	None if composted properly	None
Plant pathogens, viable seeds, plant parts	None	Virtually none if composted properly	None
Heavy metals	None	Low if from organic household residues; very low if from composted green residues	As low as possible
Volume weight ($g\ L^{-1}$ dry matter)	40–80 for white peat; 120–250 for frozen black peat	300–700 (max. 1,300)	200–400
Porosity (% volume)	Very high (85–98)	Lower than peat in all composts (50–80)	As high as possible (75)
Water capacity (% volume)	Very high (40–87)	45–65	As high as possible (60)
pH ($CaCl_2$)	2.5–3.5	6.5–8.5	5.5–6.6
Salinity	Very low <0.5 $g\ L^{-1}$	Average to high (max. 6.5 $g\ L^{-1}$ for organic household compost; max. 3.0 $g\ L^{-1}$ for green matter compost	As low as possible (max. 3.0 $g\ L^{-1}$)
N ($mg\ L^{-1}$) available	Very low (0–80)	50–500	200 avg.; 100–300
P_2O_5 ($mg\ L^{-1}$) available	None	High to very high	150 avg.; 100–200
K_2O ($mg\ L^{-1}$) available	Very low (0–20)	Extremely high (max 6,000 $mg\ L^{-1}$)	300 avg.; 200–400
Mg ($mg\ L^{-1}$) available	Present (20–200)	High to very high	100 avg.
Trace elements	Very low	Unknown	Level varies by plant type
Organic matter (% volume)	100	99	60–70

bark compost is well recognized as a media component for growing vegetables in the field and in greenhouses (Pudelski 1980, Baumann and Schmidt 1981).

Sims and Pill (1987) found that tomato seedlings grow well in sphagnum peat amended with $\leq$30 percent (by volume) biosolids or poultry manure ($\leq$1 kg N m^{-3}). Incorporation of composted biosolids or poultry manure into growth media can eliminate the need for preplant fertilization. Gouin (1985) reported that a single N supplement of 600 mg L^{-1}, 2 to 3 wk after transplanting was the only fertilization needed for bedding plants grown in potting media containing composted biosolids.

The capacity for organic materials in municipal residues to absorb significant amounts of heavy metal cations, thereby reducing the availability and toxicity of the metals to plants, animals, and humans, suggests a potential for using residues in horticultural mixes to make use of this notable "sink" feature (Jones et al. 1978). Addition of a compost made from municipal leaves, sand, and biosolids to horticultural growing media did not increase the heavy metal content of container leachate (Bugbee et al. 1991).

Compost is being used successfully as a growing medium for sod production. As much as 189 m^3 ha^{-1} of biosolids compost, leaves, or other yard trimmings are used when sod is grown on a plastic liner; these compost additions lower the water requirement of the sod and reduce the number of natural weeds (Anonymous 1991b).

Plant disease suppression

Organic matter (for example, green and animal manures and composts) is well known for affecting crop production in agricultural soil. Among the many benefits described for the use of organic matter (U.S. Department of Agriculture 1978), the effects on plant disease control and suppression are the least thoroughly understood. Soilborne diseases result in losses of more than $4 billion annually to U.S. agriculture (James 1981). Compost made from a variety of organic materials reduces plant diseases caused by such soilborne pathogens as *Phytophthora*, *Pythium*, *Rhizoctonia,* and *Sclerotinia*. Currently, however, from the standpoint of using composts in container media in the nursery industry, the plant disease suppression afforded by compost is offset by inconsistency in maturity and horticultural and microbial quality of composts. Methods for predicting compost maturity (Zucconi et al. 1981, Hirai et al. 1983, Garcia et al. 1991 and 1992, Grebus et al. 1994) and stability (Iannotti et al. 1993) are available, and more are being developed by the Agricultural Research Service and their cooperators; these methods are also being used to measure and enhance the quality and disease suppressiveness of composts (Inbar et al. 1989).

Certain precautions must be followed when composts are added to horticultural growing media. A 4-mo curing period for the compost is required before it is mixed with other container media components. Once the mixture is made, it should be allowed to stand for 3 to 4 wk prior to use (Kuter et al. 1988). This is sufficient time for the saprophytic microorganisms responsible for biocontrol of specific plant pathogens, such as *Pythium* and *Rhizoctonia*, to become established (Kwok et al. 1987). Because of differences in the amounts of microbial biomass present in composts as well as differences in bulk density, up to 55 percent (by volume, v/v) composted pine bark can be used in potting mixes (Hoitink et al. 1991), but only 15 to 30 percent (v/v) composted biosolids can be used (Gouin 1985, Sims and Pill 1987). Similar information about the use of MSW compost needs to be obtained and related to standard measurements for stability and maturity.

Research needs for developing horticultural uses of recyclable residues

There are several areas in which research is needed in order for compost to have more uses in the horticultural industry. One such area is in defining quality and maturity criteria for composts that will be used in the industry. These criteria are critical for developing the market and achieving grower acceptance for composted products.

Research is also needed to develop process technology for co-composting biosolids and the biodegradable fractions of MSW to enhance product quality and acceptability for agricultural and horticultural use. Two important tools useful in developing co-composting operations have been developed: (1) a program, suitable for hand-held programmable calculators, for computing blend ratios for two feedstocks based on the C:N ratio (Fitzpatrick 1993), and (2) a program, suitable for laptop or desktop computers, for determining mix ratios of multifeedstock composts (Brodie 1994). The feasibility and practicality of improving the agronomic value of MSW as biofertilizers need to be determined. One method for improving the agronomic value might be "spiking," a process in which the MSW is enriched with chemical fertilizers. In view of the increasing rate of landfill closures and the increasing costs of landfilling, enhancing the safe and beneficial use of co-composted MSW and biosolids in the United States may provide the lowest cost option for recycling and byproduct transformation.

Methods need to be developed to dependably enhance the microbially mediated plant disease suppression characteristics of compost. This would significantly lessen the need for biocides in the horticultural industry and would reduce multipoint sources of pollutants in runoff water.

Methods also are needed to reliably inoculate horticultural grade composts with beneficial rhizosphere microbes that can biologically mediate nutrient uptake by the plant or aid root conditioning to lessen transplant shock. These microbes help reduce the plant's dependence on synthetic fertilizers and reduce the resultant nutrient leachate from the compost.

Other Uses for Treated and Untreated Organic Residues and Byproducts

Incineration, composting, and recycling are increasingly significant alternatives to landfilling (Clarke 1992). Ideally, residuals management begins with source reduction, followed by recycling, then composting, with the remainder incinerated or landfilled. During 1988 through 1991, recycling in the United States increased and landfilling decreased; but incineration increased (fig. 5).

Yard-waste composting facilities have increased by a factor of four (fig. 6), and curbside recycling programs (fig. 7) have increased by a factor of seven between 1988 and 1995. The number of operating material-recovery facilities (MRF's) has increased from 17 to 322 between 1990 and 1995 (Berenyi 1995). These figures are based on the following definition of MRF (defined by Governmental Advisory Associates): "a facility receiving multimaterial or multigrade recyclables for processing and marketing; with some sorting of commingled recyclables, and a substantial portion collected in a municipal program" (Berenyi 1995). The recent dramatic increases in operating MRF's in the United States have contributed to increased employment at these facilities; about 7,350 people were employed nationwide in MRF's in 1995, up from about 1,500 in 1990 and 3,300 in 1992.

Paper is one of the more routinely recycled products. By late 1990, used corrugated paper sold for $27.50 to $30.29 per Mg and used computer paper sold for $154 to $198 per Mg. Recycled newsprint has experienced a decline in production because of the insufficient number of processing facilities available (Anonymous 1991c). However, by 1990 there were 25 newsprint mills in the United States, of which nine used secondary fiber to recycle newsprint. Of the 5.0 million Mg of newsprint produced yearly, 1.2 million Mg (fig. 8) are recycled to make more newsprint (Sparks 1990). Some of the remaining newsprint is used as animal bedding or as bulking agents for composting, and much of the remainder is either incinerated or landfilled. Five towns in eastern Long Island have formed an alliance under a New York Department of Environmental Conservation grant and have more than doubled newsprint recycling rates from 4 to 10 percent; the newsprint is recycled into insulation, animal bedding, and soil additives (Arner 1991).

Annually, about 544,800 dry Mg of biosolids are generated by approximately 11,500 publicly owned wastewater treatment facilities (Finstein 1989). The number of biosolids composting facilities (fig. 9) increased from 67 to 338 operating plants between 1985 to 1996 (Goldstein 1988, Goldstein and Riggle 1990, Steuteville 1996). Mixed-residue composting is increasing; 18 plants are now operating in the United States and at least 7 more are being constructed.

MSW composting is not a simple solution to the reduction in the number of landfills. Only 10 to 20 percent of MSW is actually composted. After composting, screening removes reject materials ranging from 10 to 30 percent by volume of the mixture. The reject material is placed in landfills, and normal tipping fees are assessed by weight (Goldstein and Spencer 1990). As of mid-1991, the vast majority of MSW compost was landfilled as either cover or fill. This resulted from the fact that many compost production facilities regarded composting as a means of volume and weight reduction rather than as a means of producing a highly desirable horticultural or landscaping material. In the future, MSW composting facilities must view their processes as manufacturing operations that produce an end product to vend rather than to dump (Richard 1992).

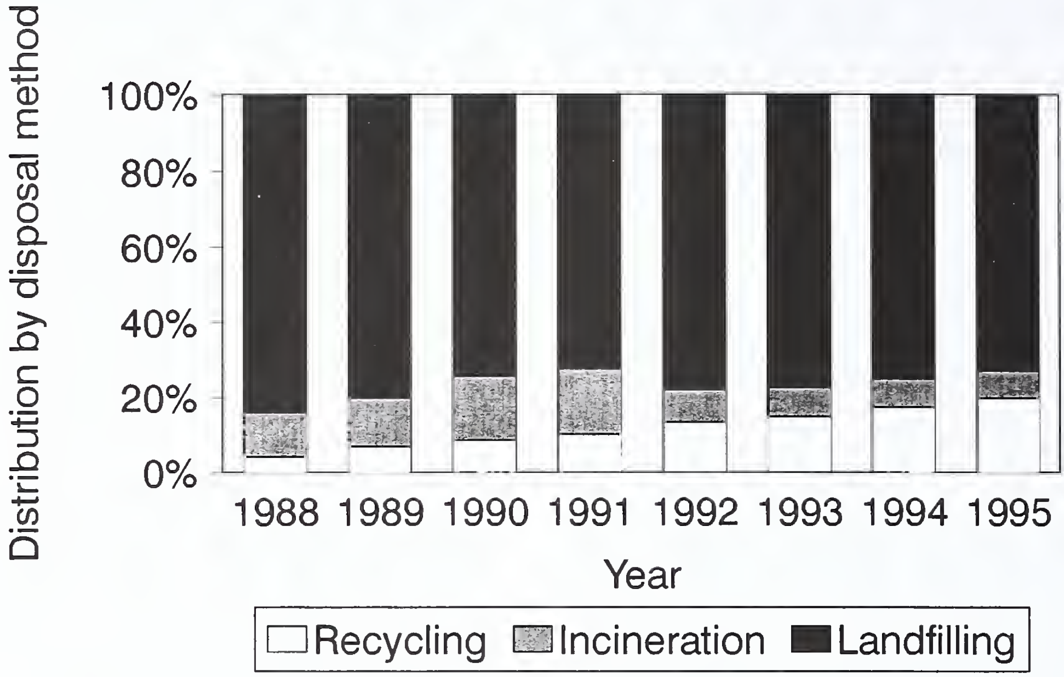

Figure 5. Percentage of U.S. municipal residues and byproducts processed by various methods

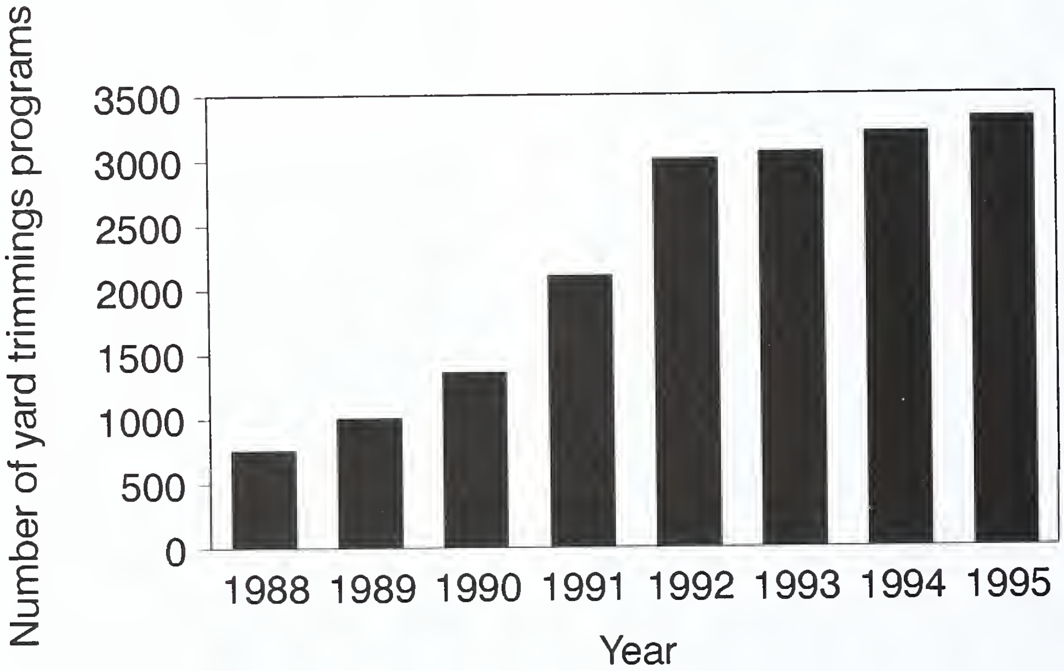

Figure 6. Number of yard trimmings programs in the United States

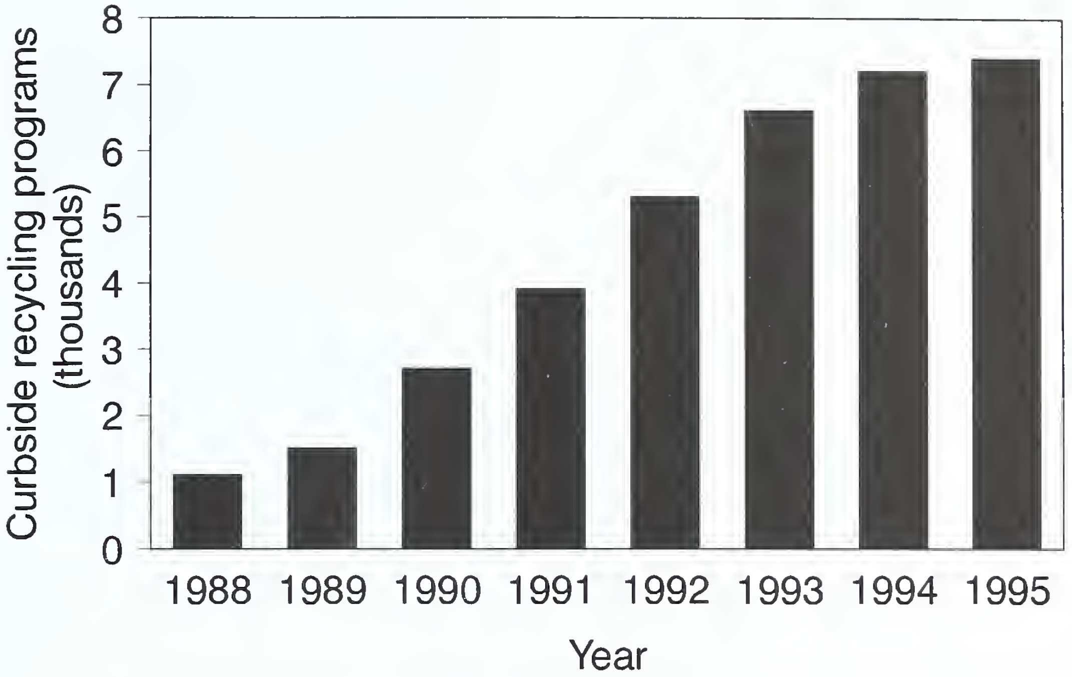

Figure 7. Number of curbside recycling programs in operation in the United States

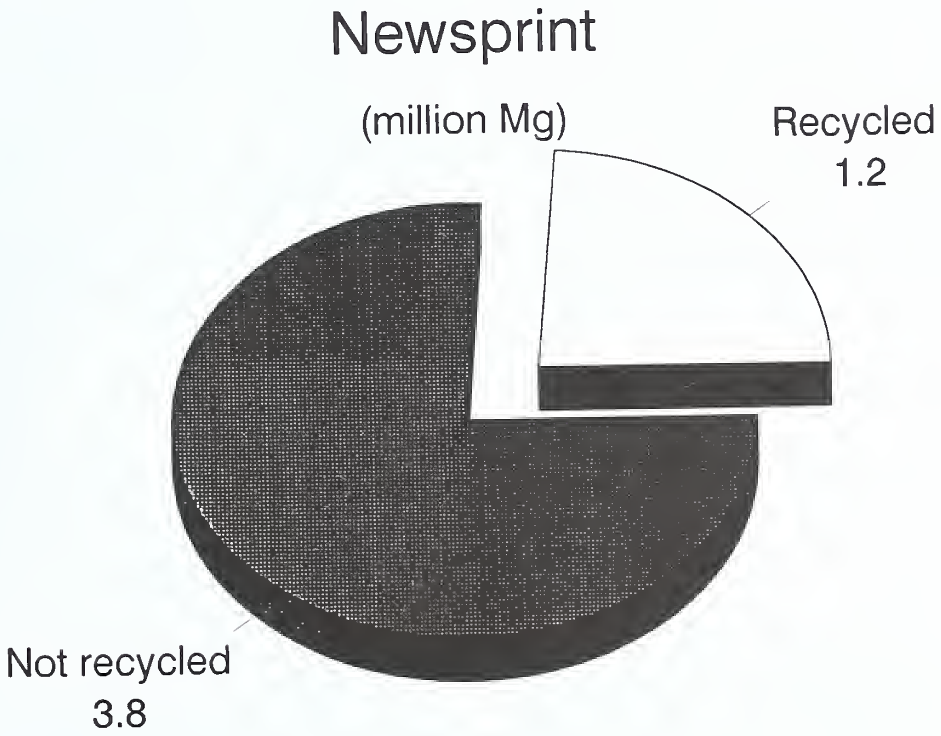

Figure 8. Recycling of newsprint in the United States

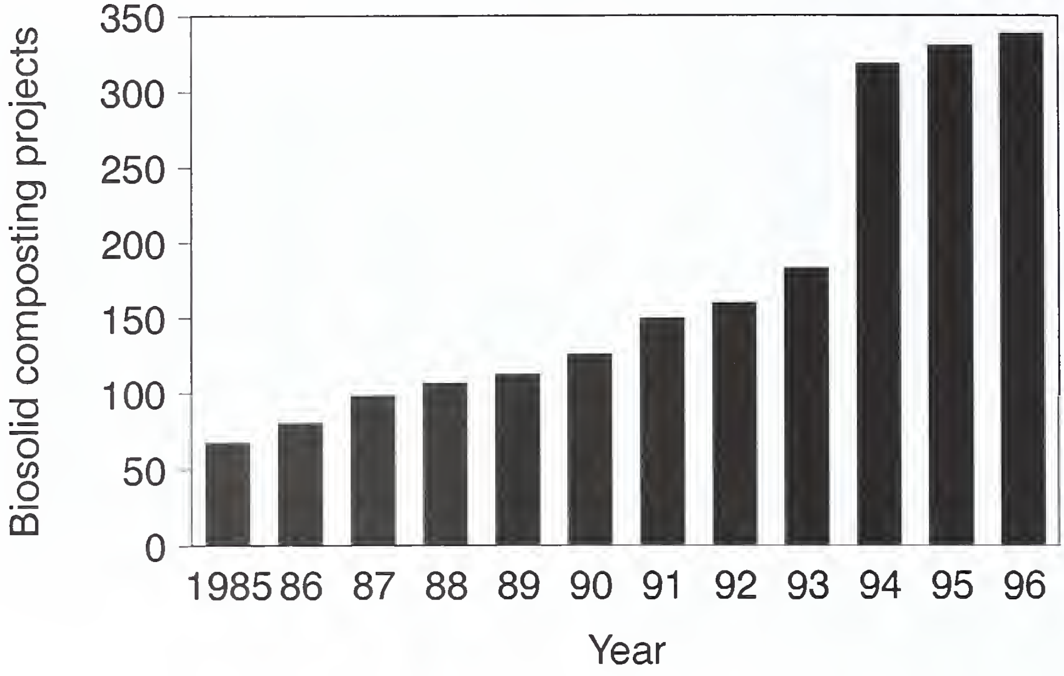

Figure 9. Number of biosolids composting projects in operation in the United States

Compost is not being marketed well so far, and substantive efforts in market development for compost will be required in the future. In a November 1990 survey of four facilities marketing MSW compost in Minnesota, Delaware, and Florida, only the Delaware facility reported income from marketing the composting product (Spencer and Goldstein 1990). In Memphis, TN, a large facility known as The Earth Complex combines biosolids composting and yard-trimmings composting with landfilling and a proposed methane-recovery system (Riggle 1991). In Austin, TX, the Hornsby Bend Waste Water Treatment Facility processes about 227 million liters of wastewater daily from which it extracts 40.9 to 45.4 dry Mg of biosolids. The majority of the material is windrow composted and then distributed and marketed under the trade name Dillo Dirt® (this trade name was registered in Texas). Dillo Dirt® is available to registered vendors at \$6.50/m^3 and to all city departments without cost. During most of the year, demand far exceeds supply. Texas and the U.S. Environmental Protection Agency released Dillo Dirt® for general use, but recommended that it not be used for growing crops for human consumption (Doersam and Armstrong 1992).

New applications for composted solid waste are currently being developed. Studies show potential uses outweighing current production by a margin of 47 to 1, with the largest potential use in agriculture (Slivka et al. 1991). Cooperative research projects on new uses are under way between the University of Maryland and the Composting Council, and these projects also involve private companies in New Jersey, Rutgers University, and the New Jersey Departments of Environmental Protection and Energy, Commerce, and Agriculture. The projects are designed to determine the potential agricultural uses for composted solids (Anonymous 1992).

Constructed wetlands are being designed and implemented to treat wastewater. By late 1990 there were 154 systems in operation or in design or construction, including systems that can treat as much as 454 million L per day (Reed 1991).

Some progress is also being made to reduce the production of toxic or noncompostable materials. Many printers are testing and using soy-based inks. Packers are using biodegradable materials with less bulk and fewer toxins.

The list of uses for recycled or composted materials expands daily. Processed garbage has been tested as a filler in concrete with good results in non-load-bearing situations. The processed garbage in the mix results in lower thermal conductivity, lower capillary suction, and lighter weight (Zhang and Whittmann 1991). Coal fly ash, the residue of coal-fired electrical generators, has been added to cement, sand, and water to produce Lytag-concrete, also known as power concrete. This mixture is almost as strong as and can be used as a general substitute for gravel concrete. Lytag concrete weighs 20 percent less than gravel concrete and has less potential to crack during hardening (Faase et al. 1991).

Composts made with coal fly ash have an increased availability of nutrients. This increased availability indicates that one or more chemical reactions and mineralization of N are occurring during composting. When compost amended with 20 to 40 percent coal fly ash was applied to soil, plant use of nutrients was more efficient (Menon et al. 1992). Furthermore, co-utilization of ash and biosolids ameliorated the reduced soil microbial activity that resulted from ash applications alone (Pichtel & Hayes 1990). Although considerable information is available on the use of fly ash and other combustion residues on land (Adriano et al. 1980, Clark et al. 1995, Korcak 1995, Norton 1995), specific research is needed to evaluate the effects of co-utilization of ash byproducts with organic materials.

Used tires, if recycled rather than being placed in landfills, can be used for several purposes. More than 200 million tires are disposed of annually. Some are reused after retreading or recapping; others find new life in the creation of playgrounds or reefs (Sienkiewicz 1990). Tires can be ground and mixed with asphalt to make an asphalt rubber, or they can be used as a rubber-modified asphalt concrete for road paving (Anonymous 1991c). Also, tires can be processed into chips that can be burned as a substitute for high-grade bituminous coal (Sienkiewicz 1990) or used as a bulking agent during composting. In the latter use, they can be recovered and reused in the process numerous times when finished compost is screened before release and delivery to the user.

Most resources used as substitutes for traditional materials are currently more expensive than the traditional ones. However, when the benefits of residuals reduction are taken into account, the cost of using recycled materials is closer to the price of using traditional materials. Lack of regulations and lack of time and funding for testing and classifying various composts have caused delays in the development of marketing programs for compost products. Development of methods for classifying and grading MSW and biosolids and composts for specific uses will help spur the development of marketing programs.

Potential Barriers and Constraints to Residuals Composting

Public perceptions and sociopolitical issues

Residuals composting has a consistent appeal to the public because it is viewed as an environmentally beneficial and economical way to return nutrients to soils and to transform unusable material into a valuable commodity. Several states presently permit MSW composting and co-composting as part of local solids-recovery plans. In most areas land application of municipal byproducts and residuals is regulated by Federal, state, and county agencies. Large, centralized municipal resource recovery facilities can be supplanted by smaller and simpler community-based composting facilities. Many local areas are now expected to compost their own MSW; this way the burden of processing is not unfairly put on a neighboring community. However, odor management remains a concern for citizens adjacent to smaller facilities.

The public's perception of composting facilities can affect whether or not a facility is built, where it is sited, and how it is operated. Public opinion is not necessarily the major factor presently influencing the future of MSW composting, but it is likely to carry more weight than in the recent past (Miller and Golden 1992).

Projects currently in place for screening and separating harmful components from MSW require expansion to provide the quality MSW needed to produce marketable compost. This accomplishment will require a coordinated effort at the local level and involve education of the public.

Odors

Odor management is a necessity for successful composting of biosolids, yard trimmings, manures, food market residuals, MSW, and other organic substances. Although composting can transform very odorous mixtures into useful soil conditioning and low-analysis fertilizer products, it can also generate odors if the process is improperly managed. Considerable information is available for characterizing odors from composting materials; new methods are being developed to abate or control the production or escape of odors.

Odors may be controlled by optimizing the composting process to minimize anaerobiosis; maximizing microbial metabolism of odorous substrates; and collecting, treating, and dispersing odors that are formed. A crucial part of planning for treatment is identifying and characterizing odors and predicting their path of transport by use of appropriate models. Ideally, these planning and modeling activities should precede the design and operation of the composting facility. More often than not, however, control efforts are initiated only after community opposition has threatened to shut down the facility (Libby 1991, Anonymous 1992, Goldstein 1994). Before new facilities are opened in the future, more proactive approaches to odor management should be applied to the various situations projected.

Control of odors should focus not only on treatment of odors, but also on preventing their generation by applying knowledge about the environmental conditions that favor and interfere with odor production. Since the conditions for producing many different odors are similar, several odors can be controlled with just a few strategies. Odor control in composting is therefore much simpler than chemical control, which might require a number of different processes to handle different chemical classes.

Odors from composts can be analyzed by gas chromatography and mass spectrometry (GC/MS). These procedures are being used more than ever because they are now more affordable. Hundreds of odorous components can be found in almost any odorous compost sample, and any of the odors produced can be at an intolerable level (that is, above their "threshold level"). Hentz et al. (1992) and Van Durme et al. (1992) have published GC/MS analyses of odorous air from various sources around wastewater and biosolids processing facilities, including around composting plants. Kissel et al. (1992) described the odorous compounds potentially emitted from MSW composting facilities. Hydrogen sulfide seems to cause odor problems in mainly acidic oxygen-starved points in wastewater treatment or anaerobic windrows. Odorants from properly operated biosolids composting plants typically arise from more alkaline and oxidizing environments, which are more likely to occur in wastewater treatment plant emissions. Alkaline and oxidizing environments also typically produce detectable odor levels of ammonia, dimethyl disulfide, and terpenes such as limonene in excess of threshold levels. Relatively less odorous compounds such as methylethyl ketone, terpene alcohols, and alkylbenzenes may produce odor levels above their threshold in alkaline or oxidizing environments.

Odorous compounds are grouped according to the method used for their removal. These groups are as follows:

Type of compound (example in parentheses)	Method for removal
Semivolatile (isoborneal)	Condensation
Volatile (limonene)	Sorption
Alkaline (ammonia)	Acidification
Acidic (butyric)	Alkalization
Reducing (dimethyl sulfide)	Oxidation
Oxidizing (dichloramine)	Reduction

There are three main treatment processes for removing odors from composting facilities: condensation—thermal or steam swept, or activated sorption; neutralization—with organic or inorganic acids or bases; oxidation and reduction—with air, organic or inorganic reagents.

A phase diagram can be used to predict the type of odor likely to be emitted from a compost. A phase diagram is based on the sulfur/carbon ratio, nitrogen/carbon ratio, and available oxygen/oxygen demand ratio of the contents of the compost. Once the likely odors to be emitted are identified, rational pretreatment strategies and odorant treatment processes can be compared in terms of their implementation requirements, such as capital, operating costs, manual labor, operator training, and maintenance training. Generally, options that decrease the air volume that needs to be treated generate large savings in compost treatment costs. Proper chemical pretreatment, compost operations, gas cooling, and air auto-oxidation reactions are cheaper than chemically treating the odors in stacks.

For several decades biofilters have been used as a means to treat odorous compounds and potential air pollutants in gas streams from wastewater facilities, MSW processing facilities, rendering plants, chemical manufacturing facilities, and composting facilities. In recent years the interest and use of biofilters for odor control in composting facilities has increased dramatically. Organic and inorganic compounds that are biodegradable and in the gas phase can be treated effectively (that is, greater than 95 percent can be removed) by biofilters.

More information is needed on the benefits of various media used for biofilters. These media must provide optimal absorption capacity, minimize system head loss, and provide an environment suitable for the proliferation of microorganisms that oxidize odorous compounds.

Pathogen and parasite destruction

Research on the infectious disease hazard of spreading biosolids on land was reviewed by Burge and Enkiri (1978); they noted that most sewage-related disease outbreaks resulted from (1) use of raw sewage wastewater, raw biosolids, or night soils on food crops that were consumed raw; (2) contamination of drinking water from septic tanks; or (3) consumption of raw shellfish from waters polluted with biosolids.

After raw biosolids are applied to the soil surface, surface movement of water could increase the hazard for pathogen distribution. Lime addition or composting, however, stabilize biosolids and reduce the hazard significantly by inactivating pathogens. Other methods for processing biosolids can also destroy pathogens but also lead to putrifaction and persistent instability. Regrowth of human pathogens, fecal coliforms, and *Salmonella* spp., has occurred even in biosolids treated by a process to further reduce pathogens, for example, heat drying, high-temperature composting, irradiation, and pasteurization (Yanko 1987). Other studies have shown that regrowth is negligible or nonexistent in well-managed biosolids composting operations (Hussong et al. 1985, Burge et al. 1987, Millner et al. 1987). In addition to human pathogens, certain plant pathogens can colonize in compost if the indigenous microflora of the compost are not too competitive (Hoitink and Poole 1980, Hoitink et al. 1991). Other plant pathogens are destroyed by heat produced during composting if temperatures are high enough for a long enough period (Hoitink and Fahy 1986, Bollen 1993).

Time and temperature criteria for destroying human and plant pathogens in biosolids are shown in figure 10. These criteria should also be applicable for destroying pathogens in composts containing MSW, manure, leaf or yard trimmings, and food-processing residuals or any of these materials co-composted with

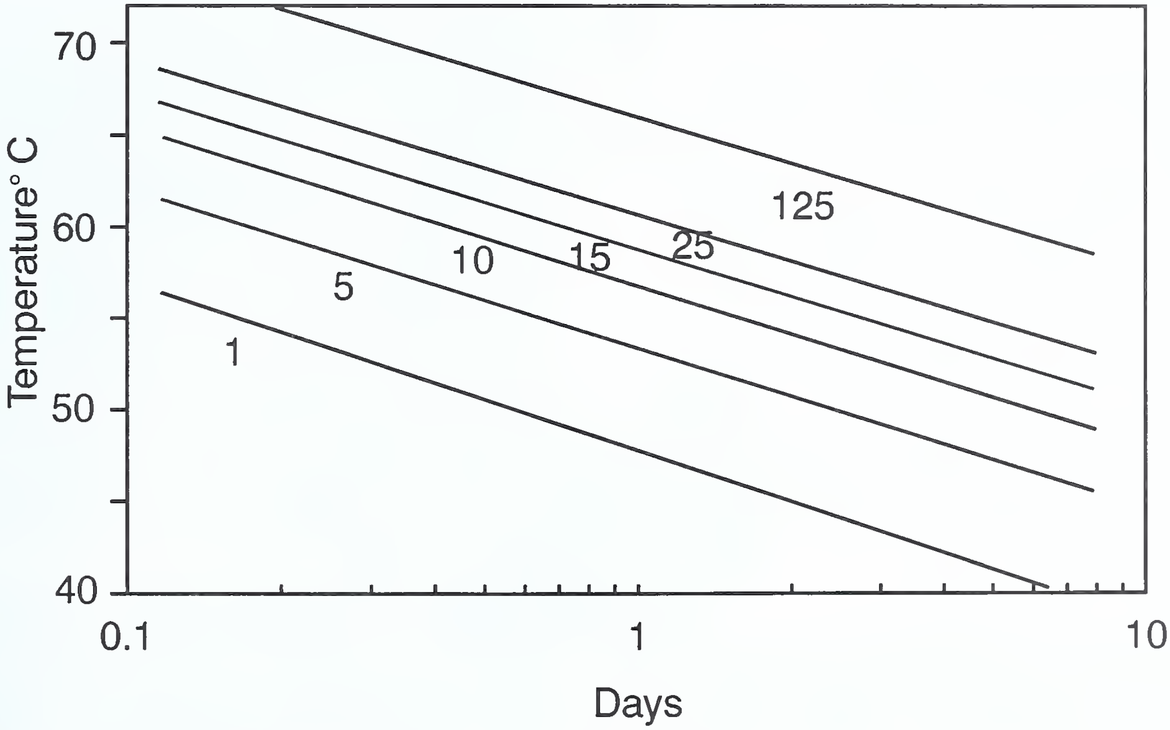

Figure 10. Curves showing the time-by-temperature regimes necessary to inactivate a desired number of logs of f_2 bacteriophage. Number of logs for each respective curve is shown below each curve.

biosolids. However, only composts that include biosolids are required to meet specific Federal regulations (EPA 40 CFR Part 503—see U.S. Environmental Protection Agency 1993) for pathogen destruction. Time and temperature criteria studies for pathogen destruction in nonbiosolids-based composts still need to be conducted.

The methods for detecting *Salmonella* spp. in biosolids, that is, the methods specified in EPA 40 CFR Part 503 regulations (U.S. Environmental Protection Agency 1993), in Hussong et al. (1984), and in Walter and Yanko (1984), were compared by Yanko et al. (1995). Ten samples of activated biosolids, anaerobically digested biosolids, and compost were included in the study. The study showed that the methods mentioned in the Part 503 regulations detected significantly fewer *Salmonella* than the two specially designed methods, which were equivalent. One of the Part 503 methods was regarded as acceptable, whereas one failed to detect the pathogen in 43 percent of known positive samples.

Additional methods for pathogen detection are still being developed using molecular biology methods, for example, the amplification by polymerase chain reaction of *Escherichia coli* to the level of 1,000 cells per gram of compost (Pfaller 1994). The use of such methodology offers the potential of detecting specific strains of pathogens such as *E. coli* O157:H7.

Water-quality factors in relation to residual use on land

Reducing the impact of agricultural practices on groundwater and surface water has been a goal of farmers and government officials for decades. A number of key factors have evolved from research efforts in this area. These factors include adding residuals to soils only in amounts in which the nutrient supply of the residue equals the nutrient requirement of the crop for that season. Because N in its soluble form does not bind appreciably to soil particles, residue application rates are generally adjusted to the potentially mineralizeable N content of the residue to prevent N loss in leachate (Sowden and Hore 1976, Sikora et al. 1979).

The P supply in residues added to soil is sometimes in excess of plant needs, but the form of P in residues and the soil type greatly affect the movement of the nutrient through the soil profile (Sikora and Corey 1976); the abundant supply often results in more P being absorbed than predicted. Because soil has the ability to remove P from percolating leachate, the occurrence of P in groundwater is rare.

The enrichment of surface (runoff) water from organic materials or fertilizers placed on land is of genuine concern. Soil conservation practices such as contour tillage, use of buffer strips, and residue management help control losses from fields. The effectiveness of these practices was evaluated by measuring overland flow areas from which organic materials had been applied (Tedaldi and Loehr 1990). This measurement is made simply by applying liquid residues to the soil surface and allowing them to flow down slope. In general the slope of the land, the infiltration capacity of the soil, the vegetation present, and the volume of rainfall in any one event determines the probability of nutrient runoff and surface water pollution (Wendt and Corey 1980, Sharpley et al. 1981, Ahuja et al. 1982).

Soil conservation practices are usually sufficient for controlling losses from highly erodible soils amended with composts and fertilizers; therefore research aimed at further reducing losses is not of high priority. Compliance with suggested soil management and farming practices normally controls contamination of groundwater or surface water by nutrients or chemicals from all amendments.

Metals and organics

In an effort to reduce the volume of materials being sent to landfills, Federal and state agencies have promoted the composting of leaves and yard trimmings. However, the environmental impacts of the various types of compost applications have not been researched or evaluated specifically with regard to the environmental fate of nutrients, metals, and pesticide residues. Previously, most research focused on the mechanics of composting. Information is still needed on (1) the range and quantities of toxic residues in yard trimmings, (2) degradation of residues during composting, (3) leaching of residues into soils and water during and after composting, and (4) concentrations of toxic constituents in finished (marketable) products.

A survey of nine composting facilities in the United States showed that the mean levels of potentially toxic metals in MSW compost were about half the mean levels in biosolids compost (Walker and O'Donnell 1991). However, there still is a need to determine (1) whether MSW composts are similar to composted biosolids in terms of capacity to limit bioavailability of metals, (2) the extent of microbially mediated degradation and immobilization of potentially toxic metals and synthetic organics in MSW and biosolids, (3) safe and beneficial methods for decreasing the bioavailability of toxic metals and synthetic organic contaminants in MSW compost, and (4) reliable methods for achieving rapid stabilization of composts.

A few investigators have reported on the occurrence and fate of organic toxicants in yard trimmings or MSW composts (Sikora et al. 1982 and 1983, Petruska et al. 1985, Savage et al. 1985, Racke and Frink 1989, Fogarty and Tuovinen 1991, Shimp 1993, Williams and Keehan 1993, Cook et al. 1994). A very high temperature is needed to destroy some toxic compounds, for example, a temperature of 427 °C is needed to destroy dicamba and trifluralin (Kennedy and Stajanovic 1969)—an unachievable, undesirable temperature for composting. At 65 °C, 28 percent of 2,4–D in MSW compost was degraded (Snell 1982).

Much of the risk of contaminants in MSW is eliminated by up-front processing of the input materials to separate organics from glass, metals, plastic, and so forth. In addition to improving the compost quality, this procedure greatly increases materials recovery (Glenn 1991) and essentially rids the compost of especially hazardous materials such as polychlorinated biphenyls (PCB's) and cadmium.

Separating motor oil from MSW is still a problem. Of more than 2,250 million L of motor oil waste generated annually, 1/3 of the used oil (or 750 million L) is poured onto the ground or into storm drains or put into the trash (fig. 11). The used oil contains carcinogenic and other toxic substances, including large amounts of lead. If it is spread on land, motor oil will reduce soil productivity, contaminate groundwater, and possibly be directly consumed in contaminated water or in plants that can accumulate hazardous levels of toxic substances. A U.S. Environmental Protection Agency report shows that 1 L of oil will foul the taste of 1,000,000 L of water. Used motor oil does not need to become a contaminant or pollutant; it can be recycled as a valuable resource. It can also be re-refined for reuse as motor oil or as heating fuel (Sienkiewicz 1989).

Proper mixing of composts—a critical step

Composting is a heat-producing biological process that begins with the decomposition of organic material and

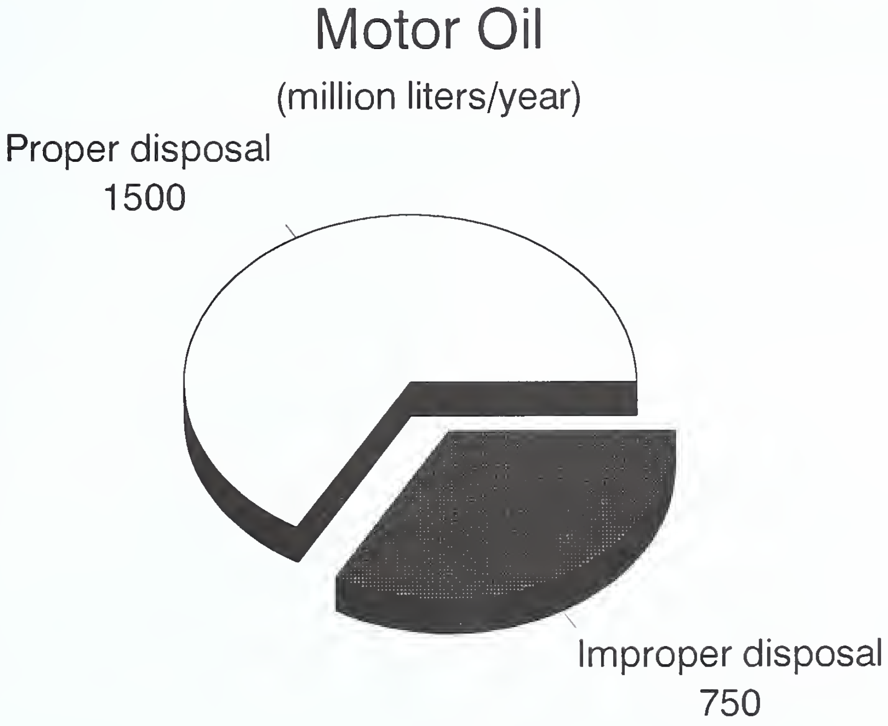

Figure 11. Disposal of motor oil in the United States

ends when energy, moisture, or oxygen become limiting. To achieve maximum degradation, it is critical that these limits not be imposed prematurely by improper combinations or improper mixing of ingredients. Premature halting of decomposition can be caused by material being too wet (reduces oxygen exchange), too dry (reduces biological activity), or poorly mixed (composting occurs in pockets rather than being uniform). For composting to be successful, mixing of the feedstocks is, therefore, the most critical step after ingredient quality.

Several types of mixers are appropriate for compost production. Batch mixers such as those used to mix livestock feed are successful for several ingredients (Rynk et al. 1992). Others include rotary drum mixers, pug mill mixers, and windrow machines (Willson et al. 1980). If materials of assorted sizes such as grass clippings and tree limbs are to be mixed, shredders, grinders, or hammer mills can be used to prepare uniform-size particles. For farm use, front-end loaders or manure spreaders have also been successful as mixers (Rynk et al. 1992). If mixtures are too wet, additional dry ingredients should be added. If wastes are too dry, additional water or aqueous residues should be sprayed over the contents during mixing.

Various manipulations of compost increase time and labor costs that must be considered in the final use of the product. The more uniform and stable the compost, the higher its value. Mixing is critical to achieve both uniformity and stability in the final product.

Limitations to on-farm composting of municipal organics

Interest in on-farm composting of municipal leaves, food, and other organic materials has been increasing. Some of the reasons for this interest include the potential for reduced disposal costs to municipalities, extra income to farmers, and a source of humus for farm soils.

Since the advantages of on-farm composting are so obvious, researchers have begun to look at the limitations to see how they can be overcome. Oshins and Kelvin (1992) studied opportunities for municipalities and farms to work together to manage rural and urban residues and byproducts. They identified six general limitations that farmers or municipalities experienced in working together (table 8). These limitations include transporting the material to the farm, farmer reliability, regulations, contamination, costs, and education. Key strategies to overcome these limitations include farmer compensation, resource coordination, and streamlining of regulations.

The study by Oshins and Kelvin (1992) was conducted in two phases. The first phase involved surveys of 60 selected municipalities and provided a cross section of demographic areas, which ranged from high-quality agricultural lands to rapidly urbanizing areas. The second phase of the study involved interviews of 71 municipal officials (elected officials, recycling coordinators, and public works directors). This phase also involved a series of focus groups, each comprised of mixed personnel (farmers, municipal officials, county personnel, and private interest groups). The function of these groups was to rank the limitations in order of importance and to propose solutions.

A primary conclusion from the study by Oshins and Kelvin (1992) was the need to develop methods for compensation of farmers. The value of the service that the farmer is providing to the municipality needs to be recognized and adequately compensated. Although some municipal residues have value to farmers, research showed that the costs of processing the material exceed the value. Tipping fees and contracts could be used to compensate the farmer for direct costs of using the materials. Contracts and compensation also help promote farmer reliability by giving the farmer incentive to stay with the program and providing the farmer with a sense of responsibility, obligation, and value.

A second conclusion of the study by Oshins and Kelvin (1992) was the recognition of the importance of an intermediary in coordinating the connection between municipalities and farmers. A broker can coordinate the delivery of materials from several municipalities to several farms more easily than individual farmers or municipalities. The highest rate of diversion of materials to farms occurs in areas having brokers (Anonymous 1991a).

A third conclusion from the study by Oshins and Kelvin (1992) identified government regulations as a problem. Existing guidelines were initially written for municipal audiences and are difficult for farmers to decipher. Co-composting municipal residues with farm manures has frequently not been addressed in either the municipal residues guidelines or manure management guidelines. Farmer-friendly guidelines for appropriate agricultural uses of municipal residues

Table 8. Limitations, factors to limitations, and strategies to working with farmers to manage municipal yard trimmings

Limitations*	Factors	Strategies
Transportation of materials to the farm	Coordination of collectors, haulers, and farmers; distance to sites; requirements for different materials	Have a broker (public or private) serve as coordinator for several municipalities.
Reliability of farmers	Availability of fields during inclement weather and during the growing season; dependability of farmers over time	Provide multiple dropoff storage sites during poor weather. Provide compensation to farmers and negotiate a contract with them.
PA–DER† management	Regulations are hard to understand, ambiguous, and inconsistent; farmers have antiregulation attitude; future regulation changes could create liability problems.	Rewrite guidelines for municipal yarn trimmings in farmer-friendly format; incorporate these guidelines into manure-management guidelines.
Contamination	Trash level	Design collection system to minimize trash. Tell residents where materials are going and provide them with compensation. Provide for disposal of trash in yard trimmings at no cost to farmer.
	Pesticide level	Preliminary data show that pesticides are not a problem, but more research is needed to allay fears. Reduce potential for contamination through a residential program that minimizes pesticide use and promotes grass recycling and not mowing sprayed lawns for several days.
Costs (other than for collection and transportation)	On-farm expenses for equipment, operations, site improvements, guideline compliance	Provide direct compensation to farmers. Costs can be shared by perhaps PA–DER† recycling grants, USDA's FSA, and EPA's Chesapeake Bay Program.
	Municipal expenses for administration and personnel	Compare cost reduction to cost of other available options.
Education	Promotion of awareness of on-farm composting of municipal residues	Included in PA–DER† yard-trimmings manual. Increase visibility of composting programs.
	Promotion of further participation by cooperating farmers	Host or attend field days and workshops on composting research.

*Listed in order of most frequently cited.
† PA–DER = Pennsylvania Department of Environmental Resources.

alone or in combination with farm operation residues is urgently needed.

When developing residue reduction and recycling strategies, municipalities should consider involving local farmers and should compare disposal on farms with other available options for disposal. Farmers often have the land, equipment, and knowledge to compost more cheaply than a municipality. Disposal on farms reduces the output transportation costs to municipalities. Another advantage to farm use of municipal residues is that the farmer is the end user of the compost.

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Chapter 2

Management of Manure From Beef Cattle in Feedlots and From Minor Classes of Livestock

B. Eghball and J.F. Power

When animals graze in pastures and rangelands, manure is dispersed across a large area and little management is needed because the material is spread over a wide area and decomposes on the soil. However, when animals are concentrated in a small area, the quantity of manure and the need for management increases significantly. In the United States, beef cattle are raised mainly in the central and southern Great Plains. Leading states for cattle raising in 1993 were Nebraska, Texas, Kansas, Iowa, and Colorado. These five states accounted for two-thirds of the U.S. beef cattle. Approximately 84 percent of the cattle are fed in feedlots having a capacity of 1,000 or more head (Krause 1991). The handling and use of the manure produced in these large feedlots is a significant environmental problem that must be addressed.

Manure from feedlots is an important resource for crop production and soil sustainability because this manure is a potential source of macronutrients (N, P, and K) as well as secondary and micronutrients. Manure is also an excellent source of organic matter when added to soils. However, manure produced by beef cattle can potentially be a source of water, air, and land pollution. These products can pollute the surface water and groundwater with excess nitrates, salts, microorganisms, and pathogens. Production of greenhouse gases from the feedlots is another factor to consider when managing animal manure.

The purpose of this chapter is to review present practices and knowledge relating to beef cattle manure production and use. Emphasis is put on manure production in confined beef feedlots because, although this represents no more than one-third of the total beef cattle population in the United States, problems related to manure management are much greater for feedlots than for pastures and ranges.

In addition to beef cattle, several other types of livestock are often raised in confinement, including sheep, goats, horses, veal calves, and mink. The production and uses of manure from these livestock are also discussed briefly.

Manure Production and Composition

There were about 99 million head of cattle and calves in the United States in 1990 (table 9). This is a reduction from the 102 million head in 1987 and 132 million head in 1975 (U.S. Department of Agriculture 1990). About 84 million of these cattle and calves are grown for beef production. If each animal excretes

Table 9. Number of cattle and calves in the United States at different times

	Number of cattle and calves (millions)							
Year	Beef cows	Milk cows	Bulls	Calves >227 kg	Calves <227 kg	Total cattle & calves	Cattle slaughtered	Calves slaughtered
1975	45.7	11.2	3.0	35.8	36.3	132.0	41.5	5.4
1980	37.1	10.8	2.5	33.3	27.6	111.2	34.1	2.7
1985	35.4	10.8	2.4	34.7	26.4	109.7	36.6	3.4
1990	33.7	10.1	2.2	33.9	19.3	99.3	34.1	2.2

Source: U.S. Department of Agriculture (1990).

56.2 kg N and 16 kg P annually, total production of N and P are about 4.72 and 1.34 million Mg, respectively. This is about 61 and 64 percent of all N and P, respectively, excreted by all classes of livestock in the United States. However, about two-thirds of these cattle and calves are raised on pastures or ranges, and the manure from pastures and ranges cannot practically be collected and used elsewhere.

About 28 million head of cattle were fattened on grain and concentrates in the United States in 1987 (U.S. Department of Commerce 1987); 64 percent of these cattle were located in the Great Plains area, compared to 58 percent in 1982. At any one time, there are about 10 million head of beef cattle on feed (table 10), and each excretes approximately 145 g of N in fresh manure daily (Overcash et al. 1983a). After the manure is collected from the feedlot, however, the N collected per head per day is 124.9 g. Thus, approximately 457,900 Mg of N (505,000 tons) is collectible annually in the manure from these cattle (table 10). Comparable values for P and K in this manure—based on 42.7 g P and 131.5 g K excreted per head per day (Overcash et al. 1983a)—are 157,000 Mg P (173,000 tons) and 482,000 Mg K (531,000 tons). If purchased as fertilizer, the value of the N, P, and K in this manure would be approximately $111 million, $180 million, and $170 million, respectively—for a total value of $461 million annually. The total value does not include the value of the minor elements in beef feedlot manure. The N from manure on feedlots is sufficient to fertilize almost 4.6 million ha of grain crops or 8.4 percent of the corn and wheat acreage in the United States at a rate of 100 kg N ha^{-1}.

Table 10. Annual manure production from feedlot beef cattle in the major cattle-producing states and N, P, and K quantities in the manure

State or country	No. of animals (millions)	Manure production* (millions of Mg)	Manure content (thousands of Mg) N	P	K	N value of manure† (million $)
California	0.39	0.94	17.9	6.1	18.8	4.4
Colorado	0.90	2.16	41.0	14.0	43.2	10.0
Illinois	0.30	0.72	13.7	4.7	14.4	3.3
Iowa	1.02	2.45	46.6	15.9	49.0	11.3
Kansas	1.70	4.08	77.5	26.5	81.6	18.8
Minnesota	0.33	0.79	15.0	5.1	15.8	3.6
Nebraska	2.15	5.16	98.0	33.5	103.2	23.8
Oklahoma	0.32	0.77	14.6	5.0	15.4	3.6
S. Dakota	0.27	0.65	12.4	4.2	13.0	3.0
Texas	2.11	5.06	96.1	32.9	101.2	23.4
U.S.A.	10.06	24.1	457.9	156.7	482.0	111.4

* Based on 2.4 Mg of manure produced per animal per year and 1.9 percent N content.

† Based on $243.2 per Mg N.

Sources: Overcash et al. (1983a) and U.S. Department of Agriculture (1990).

For the approximately 54 million head of beef cattle on pastures and ranges, their manure is dispersed across a large area. This manure is not normally collected and usually does not constitute an animal waste management problem. The effect of this manure on the environment is also minimal since the dispersed manure is decomposed on the soil. Overgrazing of pasturelands and rangelands, however, can be a potential problem by creating soil erosion and loss of riparian vegetation and by causing surface water contamination by manure.

In addition to beef cattle, several other types of livestock are often raised in confinement, including sheep, goats, horses, veal calves, and mink. Manure production and use from these livestock is also discussed briefly. There are approximately 11 million sheep and 2.5 million horses in the United States (table 11). The N and P excreted by a 45-kg sheep is estimated to be about 10 percent of that excreted by a 450-kg beef cow. Calves raised for veal produce about 20 percent as much N and P in their manure as a beef cow. We estimate that about 50 percent of horse manure is produced in confinement. Manure produced by mink has an N and P content similar to that produced by broilers. Very few goats are kept in confinement, so the amount of manure recovered from them is negligible. The total N and P recoverable (produced in confinement) from horse manure is 75 and 16 thousand Mg per yr, respectively. The next largest producer of manure is sheep. Five million head of sheep are in confinement, and the manure N and P produced in the confined feedlots of these sheep is approximately 46 and 7 thousand Mg per year, respectively. Manure N production by confined veal calves and mink totals only a few thousand Mg per yr, and manure P production of these animals amounts to only hundreds of Mg. The total recoverable manure N and P from all five species shown in table 11 is only about 129 thousand Mg of N and about 25 thousand Mg of P. Although on the national scale, total quantity of N and P from manure of these five species is negligible, some is often produced on the fringe of urban areas where manure handling, use, and odors are more critical.

Several factors that may affect mineral composition of animal manure are animal size and species, housing and rearing management, ration fed, manure storage, and climate. Typical nutrient concentrations of manure from cattle raised in feedlots are given in table 12. Overcash et al. (1983a) found that N contents of cattle manure were 3.1, 4.2, 2.7, and 1.9 percent of total solids when collected from scrapings under slotted floors, in pits or tanks, in bedded units, and in feedlots, respectively. Westerman et al. (1985) found that fresh and scraped manure from beef cattle had P contents of 1.1 and 0.7 percent of dry weight and had K contents of 2.5 and 2.0 percent, respectively. Overcash et al. (1983b) indicated that N content of urine and feces increased with increasing N content of feed. Nitrogen is often lost by ammonia volatilization from stored

Table 11. Number of sheep, goats, horses, veal calves, and mink in the United States and N and P content of their manure

			N and P production per year in manure			
	Number of head		Per head		Total from all confined livestock	
Species	Total	Confined	N	P	N	P
	------------thousands------------		------------kg------------		-----thousands of Mg----	
Sheep	11,000	5,000	9.1	1.4	45.5	7.0
Goats	2,000	–	–	–	–	–
Horses	2,500	1,250	59.6	13.1	74.5	16.4
Veal	350	350	9.1	3.1	3.2	1.1
Mink	4,600	4,600	1.2	0.3	5.5	1.3

Sources: Overcash et al. (1983) and U.S. Department of Agriculture (1990).

Table 12. Nutrient and trace element concentrations in dry feedlot manure

	Concentration (percent)	
Nutrient	Range	Average
N	0.55–4.00	1.9
P	0.12–1.60	0.65
K	0.29–3.20	2.00
Ca	0.17–3.60	1.30
Mg	0.19–1.50	0.69
Na	0.10–2.80	0.74
Fe	0.12–1.25	0.56
Zn	0.001–0.014	0.008
Cu	0.0001–0.003	0.002
Mn	0.006–0.115	0.038
B	0.014	0.014
Cl	1.4	1.4
S	0.5	0.5
Cd	0.0002	0.0002
Al	0.52	0.52
Li	0.0009	0.0009
Pb	0.0002	0.0002

Source: Overcash et al. (1983a).

manure. However, losses are highly variable (0 to over 50 percent) and depend on a number of factors, including type of storage.

Nutrient and trace element concentrations of the dry solids collected from feedlots are given in table 12. Almost all of the P excreted by cattle (96 percent) is found in the feces, and only trace amounts are excreted in the urine; in contrast most of the K excreted by cattle (73 percent) is in the urine (Safley et al. 1985). About 58 percent of N is excreted in the urine, most of it as urea (Overcash et al. 1983a). The main form of N in fresh cattle feces is organically bound N. Fresh cattle manure also contains urea and small amounts of ammonium N (Kirchmann and Witter 1992). Fresh manure from a 454-kg beef animal contained 37 percent urine (Overcash et al. 1983b). Total P, K, Ca, Mg, and Na contents of fresh cattle manure are 1.1, 2.4, 1.5, 0.55, and 0.46 percent of total solids, respectively (Overcash et al. 1983a). In the feedlot, these percentages were 0.65, 2.0, 1.3, 0.69, and 0.74 percent, respectively (table 12).

Manure Management Systems

Approximately 84 percent of the beef cattle fattened in 1989 were fed in lots with a capacity of over 1,000 head, and 50 percent were fed in lots with a head capacity of greater than 16,000 (Krause 1991). Because so many cattle are raised in concentrated feedlots, manure management and available land for application are important factors to consider. Manure management guidelines must consider (1) the effects of different management systems on nutrient content of manure at the time of spreading, (2) various methods for spreading and incorporating manure and the effects of these methods on nutrient availability, (3) methods to assist farmers in determining application rates to achieve a desired crop yield, and (4) safe manure application rates that will not cause undue losses of N and other nutrients to surface and groundwater (Bulley et al. 1980).

Feedlot manure from cattle contains considerable amounts of nutrients that can be used for crop production. Nitrogen loss from manure during storage, handling, and after application is a major problem in effectively using this resource. Up to 50 percent or more of the N in fresh livestock manure may be in ammonium form (NH_4) or may be converted to ammonium form in a very short time following excretion and is therefore subject to volatilization loss (Vanderholm 1975). In a laboratory study simulating cattle feedlot surface conditions, Stewart (1970) found N losses from urine to be 25 to 90 percent, largely due to ammonia volatilization. Adriano et al. (1971) found that nearly 50 percent of total N was lost from manure on simulated feedlot surfaces, which was consistent with their 40-percent loss from corral surfaces in the field. In studying solid waste from feedlot surfaces, Gilbertson et al. (1971) recovered 42 to 55 percent of estimated excreted N, indicating that the rest was lost. Losses of N from the feedlot are primarily through runoff or gaseous emissions (NH_3 volatilization and denitrification).

Most cattle feeding occurs in confined open lots; only a small percentage occurs in closed housing. Manure normally accumulates in the pens of cattle feedlots until animals in a pen are marketed (usually 90–180 days). At a minimum pens are cleaned out once each year. Because a high percentage of beef cattle are fed in drier climates, the mechanisms by which nutrients are lost from open feedlots are much different than those from confined housing operations, especially confined operations in which water is used to flush manure into pits for storage. Typically in the central and southern Great Plains, cattle feedlot manure is scraped from feedlots. The manure collected from these scrapings can contain up to 50 percent soil. This manure is then stockpiled until fall. Spreading on cropland normally occurs after harvest in the fall or in early spring before crops are planted.

Much of the lamb fattening in the United States occurs in open pens (similar to how beef cattle are fed). Feedlots for lambs are cleaned with the same frequency as those for beef—usually once or twice per year. Lamb feeding, however, is more seasonal, and therefore feedlots for lambs may be idle part of the year. The idle period may lead to more mineralization and a greater possibility for nitrification, but no research has been done to prove this. Lamb feeding occurs primarily in western states, such as California, Texas, Wyoming, and Oregon. Numbers of lambs fattened declined from 7.8 million in 1975 to 5.4 million in 1989.

Horses are generally kept in pastures or in stables or corrals or a combination of the two. While very few draft horses exist in the United States, saddle horses are found throughout the nation. In western states, these horses are used primarily on cattle ranches for working and managing beef cattle. Few of these horses are kept in confinement for appreciable time periods. In the last several decades, the number of pleasure riding horses has increased greatly, mostly on small farms surrounding urban centers. Many of these horses are kept in pens or stables for considerable time periods, requiring the periodic removal of manure. Often only a few horses are kept on each farm, so there is usually sufficient pasture land on which to spread manure. There are a limited number of instances (such as racing stables and other large stables) where manure disposal from horses is a problem. Usually horse manure is applied to land for crop production, similar to beef cattle manure. There are some specialized uses of horse manure, such as for mushroom production in several northeastern states. The spent manure from these operations is then applied to land.

Mink are produced entirely in confinement, so eventually all manure produced from mink is collected and used. Approximately 1,000 mink ranches exist in the United States (about the same as in 1975). States with the largest number of mink operations are Wisconsin, Minnesota, Oregon, Idaho, and Utah. The total quantity of manure produced on these ranches is relatively small. Many mink operations are located near urban areas on limited land bases, so arrangements are often made with local farmers for the manure to be used.

If traditional methods for collecting manure from beef feedlots are used, often over 50 percent of the N in the manure is lost before the manure is removed from the feedlots (Gilbertson et al. 1979b, Overcash et al. 1983b). For example, in Nebraska, Gilbertson et al. (1971) found that 80 percent of the N fed to beef cattle was excreted in the manure, but only 48 percent of this N was in the manure collected from the feedlot (39 percent of the total N in the feed was in the manure collected). For dairy barnlots, Safley et al. (1986) measured N, P, and K losses of 23, 0, and 10 percent, respectively. Depending upon how the manure is handled and field applied, as much as 50 percent of the N remaining in the manure after removal from feeding pens may be lost by the time the manure is spread and incorporated. Thus, often only a fraction (about 25 percent) of the N excreted in feedlot manure is applied to the field for the growing crop.

There is a tremendous opportunity to improve the efficiency of recovery and use of the N in feedlot manure. Several practices might be used to reduce N losses from the manure deposited during the feeding operation. These might include more frequent cleaning, use of bedding, and use of additives to reduce volatilization and denitrification losses of N. Other options might include the use of such materials as nitrification inhibitors, urease inhibitors, acidifying materials (phosphoric acid, pyrite, ferrous sulfate, sulfur), and precipitants or stabilizers (alum, quick lime, or cement kiln dust). Although there have been a few studies using some of these materials on poultry or swine manure, essentially no comprehensive research has been conducted on their use for beef cattle feedlot manure.

Temperature, moisture, pH, and C/N ratio are among the important factors in determining the amount of N lost from manure. Muck and Richards (1983) concluded that little N is lost if daily temperatures are below 5 °C, but 40 to 60 percent of total manurial N is lost through ammonia volatilization at temperatures between 5 °C and 25 °C. Adriano et al. (1974) found that at 10 °C, average losses of N from cattle manure were 26 and 39 percent at 60 and 90 percent moisture levels, respectively. At 25 °C, 40 and 45 percent N losses were observed for the 60 and 90 percent moisture levels, respectively. Manure application rate did not have a significant effect on the percentage of N lost. When manure was mixed with soil and incubated in large soil columns, Peters and Reddell (1976) found a 10-percent loss of total N at a soil pH of 7.5 and a 20-percent loss at a soil pH of 12. Stevenson and Wagner (1970) stated that losses of N as free ammonia are particularly serious on calcareous soils. Webber and Lane (1969) reported that a soil pH > 8.0 is favorable for ammonia volatilization.

A large C/N ratio in manure may reduce volatilization loss of N. Bedding placed in the feedlot helps absorb urine and helps reduce volatilization losses of N during drying by immobilizing more N. However, Hensler et al. (1970) concluded that the use of bedding results in a reduction of manure N availability because it causes the manure to have a large C/N ratio. They also found that when manure was applied to corn, total dry matter of corn was not affected by manure containing up to 8 percent bedding; however, when the bedding was increased from 8 to 16 percent, yields were usually less than they were from manure having no bedding.

Composting manure is a useful method of producing a stabilized product that can be stored or spread with little odor or fly-breeding potential (Sweeten 1988). Composting also kills pathogens and weed seeds and improves handling characteristics of manure by reducing its volume and weight (Willson and Hummel 1975). Decomposition of manure occurs through biological action and spontaneous chemical reactions. The initial chemical and biological composition of manure is a function of ration fed, animal age, bedding used in feedlots, and other factors that can influence the decomposition process. In a constant temperature and humidity chamber, ammonia volatilization from beef cattle manure resulted in a 35-percent decrease in N content of the material during composting (Stone et al. 1975). In the compost, ammonia was only 3 to 4 percent of the total N and 0.05 to 0.1 percent of the dry manure. The greatest loss of ammonia occurred at 48.8 °C and 70 percent moisture.

Eghball and Power (1994) found up to 40 percent N loss during open composting of beef feedlot manure. The amount of N loss was proportional to the initial manure N content, and ammonia volatilization accounted for more than 92 percent of the N loss. They also found significant runoff losses of K and Na but minimal loss of P from composting windrows. Wells et al. (1969) also showed that N is lost as ammonia during composting. Martin et al. (1972) indicated that increasing the C/N ratio of the waste decreases the amount of N lost during composting. Loehr (1974), however, stated that composting conserves much of the nutrient content, including N. Compared to fresh manure, 3-mo-stabilized farmyard manure had significantly greater concentrations of total N, water-soluble substances, and lignin and had less organic C, lipids, and hemicellulose and a lower C/N ratio (Levi-Minzi et al. 1986).

Nitrogen loss during composting depends on the conditions under which the material is being decomposed. Willson and Hummel (1975) found that while moisture content, pH, and material bulk have little effect on N loss, periods of anaerobic activity during composting may increase N loss. Since N losses are more than offset by the reduction in volatile solids due to biooxidation, N concentration during composting usually increases. During composting, N can be lost from manure in runoff and by nitrate leaching. The quantity of N lost by these processes is affected mainly by site-specific conditions.

Composted manure can be applied to soil as an odorless and drier source of nutrients as compared to noncomposted manure. In addition, Kirchmann (1990) found that applying composted poultry manure to soil caused plants to take up more soil N than they did when fresh poultry manure was applied. Composted manure with low available energy caused a positive N interaction, while energy-rich fresh manure caused a negative N interaction and a subsequent lower soil N uptake.

The amount of manure to be applied to a particular soil depends on crop requirements for N and P, composition of the manure, and environmental conditions. Manure applications to provide adequate N for crops may result in soil buildup of salt, P, K, and other ions in areas where rainfall is limited. It is usually best to base manure application rates on the P needs of a crop and to supply additional N with fertilizer if needed. This method of supplying N and P helps to avoid adverse environmental effects, especially nitrate leaching, runoff losses, and high P levels in runoff.

After manure is applied to soil, nutrient loss depends on degree of incorporation and environmental conditions. In laboratory and field experiments, Steenhuis et al. (1981) showed that most N loss from manure spread on frozen soil was in water soluble forms (mainly nitrate and ammonium-N). The first meltwater contained the highest concentration of readily available N. Dairy manure applied at 35, 100, and 200 Mg

ha^{-1} in 1972 resulted in average runoff losses of 16, 1, and 0.2 kg inorganic N ha^{-1} in 1972, 1973, and 1974, respectively (Klausner et al. 1976). Phosphorus losses from the manure during these 3 yr were 3.5, 0.7, and 0.01 kg ha^{-1}, respectively.

Land Application of Manure

There were about 390 million ha of land in farms in the United States in 1993. Of this farmland, 133 million ha were cropland and 265 million ha (650 million acres) were pasture and rangeland (U.S. Department of Agriculture 1993). Nationally this provides an ample base for land application of animal manure. In only a limited number of counties nationwide, mostly in coastal states where few beef cattle are produced, does the supply of livestock manure greatly exceed the cropland available for manure use. Other potential uses of manure are landfilling, burning, converting to methane, and refeeding to other animals. However, for beef cattle feedlot manure, particularly that produced in the central and southern Great Plains, these other options offer limited opportunity for manure use. Factors to consider for land application of manure are transportation and spreading-equipment problems and related costs, land base available, problems in collecting a representative manure sample for nutrient analysis, and application rates that provide the crop with sufficient nutrients without having adverse effects on the environment.

Transport of animal manure to the application site is an important part of any management system. Manure can be in solid, slurry, or liquid (<5 percent dry matter) form, and each requires a different management practice. Beef cattle feedlot manure, valued for its N and P nutrient content, is an economical substitute for commercial fertilizer. Freeze and Sommerfeldt (1985) found that the cost of hauling manure from large feedlots (>500 head capacity) is justified up to about 15 km. If the manure is hauled a greater distance, the cost of hauling will exceed the value of nutrients in the manure. They also found that manure from small feedlots (<500 head capacity) can also be economically hauled up to 15 km if noncash costs and labor charges are disregarded. Fortunately, in the area of the United States where most beef cattle are fed (Iowa to Colorado to Texas), most of the land is under cultivation, so there is seldom a shortage of farmland for application of manure. If an animal excretes 145 g N per day, this would provide about 53 kg N per yr—about enough to fertilize 0.3 ha of irrigated corn, assuming the N is 100 percent effective and no N losses occur. Thus the manure from a 10,000-head feedlot should be spread on 3,000 ha or within a radius of about 3 km of the feedlot, and the manure from a 50,000-head feedlot should be spread on about 15,000 ha or within a radius of about 7 km of the feedlot.

Manure spreaders are the most typical device for transporting and spreading animal wastes with moisture contents <80 percent (Overcash et al. 1983b). These spreaders can be either box-type or of the open-tank design. Box spreaders can be pull-type or truck mounted. Slurries may be transported with either a mobile tank or by pipeline. Some agitation is necessary before removing liquid material from storage areas or pits. Liquid waste with hydraulic behavior like water is normally transported in tanks, although this is more expensive than using irrigation equipment for transport. The liquid wastes can be applied with surface (furrow, flooding, or border) irrigation, but better distribution can be obtained by using a sprinkler irrigation system, a traveling gun, or a center-pivot system.

Solids, slurries, and liquids can be applied to the surface or incorporated into the soil. Applying animal manure below the soil surface has advantages in that it prevents an unsightly appearance to the field, reduces odor and fly problems, reduces volatilization and runoff losses, and generally results in better conservation of nutrients for use by crops (Barlett and Marriott 1971). Large-bore irrigation nozzles can be used on sprinkler irrigation systems to handle slurries as well as liquid wastes.

Recent farm legislation in the United States requires producers to protect highly erodible soils from erosion. Therefore, when crop residues are sparse, it may not be possible to incorporate manure and still meet conservation compliance requirements. Unfortunately there have been few experiments conducted using beef cattle feedlot manure in no-tillage cropping systems, but considerable ammonia would probably be lost by volatilization because of lack of incorporation.

Effective use of manure and determining the best agronomic rates of application depend on proper sampling of the manure. Since animal manure is highly variable in nutrient content, collecting a representative sample for analysis is essential for determining proper application rates. Manure applied in excess of the crop needs for any nutrient can contribute to surface water and groundwater contamination. Soil sampling should be done prior to manure application to assess the nutrient additions needed for the crop and to determine the proper application rate based on manure nutrient content. Because plant availability and crop uptake of nutrients in manure are affected by many variables, it is usually desirable to adhere to Extension Service recommendations in each state to determine proper application rate. Gilbertson et al. (1979b) estimated that on the average, about 35 percent of the N and 20 percent of the P in cattle manure were utilized the first year after application to a corn crop but that these values can range widely depending on conditions.

Alternative Uses of Cattle Feedlot Manure

Cattle manure has been used for algae and fish production in lagoons, reclamation of sandy and mined soils, recovery of energy (collection of methane gas), and refeeding (Umstadter 1980). Anaerobic bacterial decomposition of cattle manure generates methane gas, which can be collected and used for various purposes. About one-third of the manure N can be refed to animals, depending on the type of manure and type of animal consuming the manure (Overcash et al. 1983b). Manure can also be used in pyrolysis, hydrogasification, oil conversion processes, composting, and fish farming. Pyrolysis is a process in which animal manure is pretreated by thermochemical processes in a closed system at elevated temperatures of 204 to 800 °C. This process results in the production of the following three fractions: a solid fraction termed ÒcharÓ, a gas fraction that when condensed is an oil or fuel, and a gas fraction that when condensed is aqueous in nature. Hydrogasification is a process in which cellulose in the presence of hydrogen is partially converted to a gas rich in methane. This process requires high pressure and temperature. A process similar to liquidification of coal can be used to convert manure to an oil-like product.

Composting is the aerobic treatment of manure in the thermophilic temperature range (40 to 65 °C). Compost is an odorless, fine-textured, low-moisture-content material that can be used in bulk as a fertilizer or bagged and sold for use in gardens, potting media, or nurseries. The heat generated during composting also can be harvested.

Although there are a number of potential uses for cattle feedlot manure, in practice only a small fraction of manure is used for purposes other than land application. Part of this probably results from the fact that most beef cattle manure is produced in agricultural regions where demand for other products (methane gas, energy, etc.) is better provided by other sources. Although this manure, when processed, can be fed to poultry, the distance between concentrated cattle feeding and poultry production centers is generally too great for this practice to be economical.

In the Far East, manure has been used in aquaculture for centuries. Wohlfarth and Schroeder (1979) found that maximum yields per unit area in aquaculture are higher when high-protein feeds are used instead of manure, but the high-protein feed costs more. Best results were obtained in fish ponds with frequent applications of manure. Incorporating manure into high-protein feeds for aquaculture resulted in reduced growth and failed to reduce feed cost per unit area.

Agronomic and Environmental Effects of Manure Application

Beef cattle manure is a valuable resource because of its nutrient and organic matter contents and can be effectively used for crop production and soil improvement. Manure contains N, P, K, and micronutrients that are necessary for plant growth. Organic matter content of soil can be increased by adding manure to the soil. Organic matter is an ion exchange material, a chelating agent, a buffering material, and an important agent in soil aggregation. Total organic C, Kjeldhal N, and potentially mineralizable N in manure-amended surface soils (0 to 7.5 cm) were 22 to 40 percent greater than in nonmanured soils receiving fertilizer or herbicide or both (Fraser et al. 1988). Application of cattle feedlot manure significantly increased soil organic matter and total N and lowered the C/N ratio in the top 30 cm of soil in (Sommerfeldt et al. 1988). Soil organic matter, available P, and exchangeable K, Ca, and Mg increased on a loam and a sandy loam soil with increasing rates of manure application (Vitosh et al. 1973).

Manure application can improve soil physical properties such as infiltration, aggregation, and bulk density, which in turn results in reduced runoff and reduced wind and water erosion. Manure also decreases energy needed for tillage and reduces impedance to seedling emergence and root penetration. Increased soil aggregation and subsequently better soil water infiltration also result =from manure application (Mielke and Mazurak 1976, Boyle et al. 1989). However, excess manure application may have adverse consequences. In addition to increased potential for surface water and groundwater pollution, excess manure application may increase soil electrical conductivity and the sodium adsorption ratio and may decrease soil pH (Chang et al. 1991). An increased sodium adsorption ratio may reduce soil water infiltration rates.

Conservation of nutrients in storage and during handling and more timely incorporation of manure to conserve N and other nutrients could reduce the cost of crop production. These practices offer the commercial crop and livestock producer an opportunity to achieve a greater degree of self-sufficiency in recycling nutrients and using energy efficiently (Stonehouse and Narayanan 1984). When farmyard manure was priced on the basis of its total N and P contents, net returns for applications of 11 and 22 Mg manure ha^{-1} averaged \$48 and \$100 ha^{-1}, respectively (Holt and Zentner 1985).

Beef cattle manure application can increase the yield of most crops. In several published results, the yield of corn silage, corn grain, grain sorghum, forage sorghum, and perennial forage crops were increased with applications of cattle manure or manure effluents (Sukovaty et al. 1974, Swanson et al. 1974, Mathers et al. 1975, Magdoff and Amadon 1980). Manures, if properly handled, are a good substitute for fertilizers as a source of nutrients and have the added benefit of improving soil physical characteristics.

Manure should be managed and applied at rates that do not adversely affect the environment. Manure applications supplying available N in excess of crop requirements can be a potential source of groundwater contamination. For grass swards grown on a deep and well-drained soil, manure supplying approximately double the crop's total N requirement contributed nitrate-N to the groundwater (Marriott and Bartlett 1975). Plots treated with 22 to 224 Mg of manure ha^{-1} had nitrate-N amounts ranging from 100 to 2,400 kg ha^{-1} in the top 1.8 m of soil (Mathers et al. 1975). Deep-rooted crops can be used to extract nitrate-N from soil depths greater than that of the root zone of most annual crops (usually 1 to 1.5 m). Alfalfa grown on heavily manured plots removed water and nitrate-N to a depth of 1.8 m the first year and to 3.6 m the second year. Schuman and Elliott (1978) also reported significant removal of nitrate-N by alfalfa from an abandoned feedlot area with elevated nitrate content (> 2,000 kg nitrate-N h^{-1}) to a soil depth of 4.6 m. Corn was not as effective as alfalfa in removing nitrate-N and contained too much nitrate in the forage to be safely used by livestock (Schuman and Elliott 1978).

High rates of manure application will cause a significant buildup of N, other nutrients, and salt in the soil. Large applications of manure (> 22.4 Mg ha^{-1}) can also cause a significant buildup of soil exchangeable K and extractable P (Vitosh et al. 1973). Bray and Kurtz No. 1 P soil-test values increased linearly from 45 to 391 mg kg^{-1} with manure applications of 0 to 361 Mg ha^{-1} (Vivekanandan and Fixen 1990). These high soil P levels could have adverse effects on the availability of some minor elements. In areas with heavy rainfall and natural leaching, salinity buildup from manure application is not a major problem; however, in irrigated and low-rainfall areas, application of materials containing salt must be limited to prevent salt accumulation (Gilbertson et al. 1979b). The amount of NaCl salt in the beef ration directly affects Na concentration in the manure, which in turn affects the exchangeable Na and sodium adsorption ratio in soil (Horton et al. 1975). Sodium accumulation results in soil dispersion and greatly reduces infiltration. The quantity of NaCl in rations today is considerably less than 20 yr ago, so the problem is less acute than it was when much of the reported research was conducted.

Manure in the feedlot can be a source of pollution. Nitrate-N in abandoned feedlots averaged 7,200 kg ha^{-1} in a 9.1-m soil profile, whereas adjacent cropland had only 570 kg ha^{-1} nitrate-N in the same soil depth (Mielke and Ellis 1976). Some abandoned feedlots had as much as 18,200 kg nitrate-N ha^{-1} in a 9.1-m soil core. However, Ellis et al. (1975) took soil cores from 15 active eastern Nebraska beef cattle feedlots and showed that most were not a nitrate pollution hazard to groundwater. In active feedlots compaction from hoof action coupled with NaCl in the manure resulted in essentially no water infiltration or leaching (Mielke and Mazurak 1976); hence little accumulation of nitrate occurred in the subsoil (Lorimor et al. 1972). Mechanical removal of manure from feedlots also reduced the opportunity for nitrate movement into the soil, helped to maintain the surface of the feedlot in an aerobic condition, and minimized odor (Lorimor et al. 1972).

Runoff from cattle feedlots can contaminate surface waters. Pollutants in this runoff include chemicals, microorganisms, organic materials, and soil sediments. Proper assessment of the pollution potential depends not only on the size, stocking rate, and other physical characteristics of the feedlot but also on the intensity, duration, and frequency of rainfall (Swanson et al. 1971). During a rainfall event, runoff will begin sooner from a feedlot than from adjacent cropland because of the lower infiltration rate in a feedlot. Ammonium and nitrate-N are transported in the initial runoff from the feedlot surface and add to the surface water pollution problem (Swanson et al. 1975). Under Nebraska conditions, typically only 3 to 6 percent of the manure deposited in a feedlot is removed in runoff (Gilbertson et al. 1979a). Erosion in the feedlot depends on the land slope, slope length, infiltration rate, and physical properties of the soil. Methods of surface water control have been developed for feedlots to reduce or collect the runoff water, such as the use of terracing, check dams or porous dams, settling basins, tiled infiltration beds, lagoons, and vegetative filters. Wetlands can be constructed that use vegetative filters to remove solids and some soluble nutrients from runoff water before it is impounded in a shallow basin.

Runoff loss also can occur from the fields receiving manure and can contribute to pollution of surface waters. The amount of runoff is influenced by time, rate, and method of application and by soil and cropping management practices (Khaleel et al. 1980). Application of manure to frozen soils often results in the loss of organically bound N and P with snowmelt runoff. High nutrient loss also may result from runoff events occurring shortly after application. Therefore, it is best to apply manure when runoff events are least likely. Incorporation of manure after application reduces runoff loss, conserves manure nutrients, and improves soil physical properties. The amount of runoff loss usually increases with increasing rate of application. Patni et al. (1985) found no consistent differences in bacterial quality of runoff from manured and nonmanured fields when the manure had been incorporated.

Manure can be a source of air pollution because several gases are formed and volatilized during decomposition. Considerable dust may also be added to the air. Gases such as carbon dioxide, methane, ammonia, nitrous oxides, and hydrogen sulfides may contribute to the greenhouse effects (warming of the atmosphere by trapping of heat). The magnitude of the contribution of these gases to global warming is not known. Ammonia is readily volatilized from the urea in urine and often increases atmospheric NH_3 concentrations severalfold near feedlots (Elliott et al. 1971). However, ammonia is readily washed back into the soil by precipitation, so air contamination is usually local. Nitrous oxides escape to the atmosphere when nitrates are denitrified, usually under wet conditions such as rain-soaked feedlots. Nitrous oxides can be a major contributor to the greenhouse gases. Unfortunately essentially no data are available to quantify the amount of nitrous oxides emitted from beef cattle feedlots annually.

Agronomic and environmental effects of the manure produced by the other five species of livestock discussed in this section do not differ greatly from those discussed above for beef cattle. Almost all the manure produced from these five species is used for land application (an exception being horse manure used for mushroom production). No recent data exist on the decomposition rate or nutrient availability of these manures, but they are presumed to perform similarly to manure from beef cattle.

Effective, Nonpolluting Uses of Cattle Feedlot Manure

Education is the key to a proper animal manure management system. Water quality protection, particularly from nonpoint sources or unregulated point sources, is one of the issues that needs to be addressed by increased research, technology transfer, public policy initiatives, and private action on the part of the producers (Sweeten 1992). Other issues include air quality protection, emissions of greenhouse gases, land and soil sustainability, animal welfare, water use, societal and producer's benefits from animal manure, energy recovery from animal manure, effects of pollution from animal manure on the animals themselves, and ability of livestock to convert nonedible plants into human food products (Sweeten 1992).

Point sources of water pollution from livestock can be minimized or eliminated by use of proper management systems that include selection of appropriate sites for concentrated animal-feeding operations, proper design of manure storage areas, wastewater collection and application to croplands, and application of nonexcessive rates of manure to croplands. Air-quality impacts of animal manure can be lessened by aeration, anaerobic digestion, composting, and capture of odorless and odorous gases. However, reducing the impacts of manure on water and air quality will require development of economically viable management alternatives for diverse feedlot settings.

Government regulations can greatly alter the management system used in a beef cattle production operation. The Federal regulatory approach to animal manure management emerged in the early 1970's as the U.S. Environmental Protection Agency initiated its regulatory programs to implement the goals of the Clean Water Act of 1972. U.S. Environmental Protection Agency regulatory efforts initially focused on point sources of pollution, which at the time were mainly effluents and solids from urban and industrial areas. Agriculture was largely seen as a nonpoint source of pollution. However, in 1973 the U.S. Environmental Protection Agency identified concentrated animal-feeding operations (CAFO's) as point sources of pollution and from that point on required the issuing of National Pollutant Discharge Elimination System (NPDES) permits. CAFO's included operations in which more than 1,000 animal units (cattle or equivalent for poultry and other animals) were confined and fed for at least 45 days or in which pollutants were discharged following storms smaller than a 25-yr, 24-hr storm event. Medium-sized feedlots with 300 to 1,000 animal units that discharge pollutants directly into navigable waters were also made subject to NPDES permits. Land application of animal manure was also considered a nonpoint source and was not subject to NPDES permits. Nonpoint sources of pollution became the target of U.S. Department of Agriculture and state voluntary programs for improved animal manure management. State regulatory approaches are basically consistent with NPDES requirements but vary from state to state depending on differences in climate, rainfall amounts, and the number and mix of livestock.

Best management practices are essential for the effective use of beef cattle manure for crop production and pollution prevention. Nutrient conservation is the first step toward a best management system. Nitrogen is the most susceptible nutrient to loss by volatilization and leaching and subsequently should be conserved as much as possible. Factors that affect N loss include temperature, moisture, pH, aeration status, rainfall, and C/N ratio. These factors should be considered when planning the uses of animal manure. Most other nutrients (for example P, K, and Ca) are lost only through runoff and erosion of organic material. Reducing erosion and controlling runoff will considerably reduce the loss of all nutrients.

Proper rate and method of manure application are vital to nutrient conservation and can greatly improve soil sustainability and crop production. Manure should be applied at a rate that provides adequate but not excessive nutrients to the crop. Incorporation of manure after application greatly reduces nutrient volatilization and runoff loss. If incorporation is not possible because of the increased soil erosion hazard from incorporation, ammonia volatilization will probably be greater, but there is essentially no long-term research evaluating these effects.

Beef cattle manure can be effectively and economically used by crops if a proper land base area is available to the cattle feeding operation. Manure can be an economical substitute for commercial fertilizers when it is transported no more than about 15 km from the source (Freeze and Sommerfeldt 1985). Because most of the major beef feeding operations in the United States are located in rural areas away from centers of population, there are relatively few problems with odors or fly populations.

Research Needs for Improved Management of Cattle Feedlot Manure

When beef cattle feedlot manure is considered as potentially a major source of N for the crops produced in the United States, several facts are apparent. Present feedlot management systems result in about a 50-percent loss of N from the manure before it is removed from the feedlot. In addition, another 25 percent of the N excreted in the feedlot may be lost as the manure is hauled, spread, and incorporated into the soil. Thus, often only about 25 percent of the manure N is actually applied to cropland. Consequently, considerable additional research is needed to develop practical feedlot and manure management practices that will reduce these losses of N to the environment. This approach would also reduce the magnitude of environmental damage that is now associated with beef feedlot operations.

We presently have some evidence that several changes in feedlot management may have some potential for reducing N losses from manure. These include such practices as frequent cleaning and the use of carbonaceous bedding (straw, cornstalks, paper), inhibitors (chemicals that decrease the rate of nitrification or urea hydrolysis), and various types of stabilizers (acids or acidic materials, quick lime, and alum). However, considerable research needs to be conducted to determine the benefits, costs, and practicality of these new methods.

There are many unanswered questions and problems in determining the proper rate of application of manure to land. Suitable methodology is lacking for making rapid and economically acceptable field determinations of the nutrient content of manure—a necessary step in calculating acceptable rates of application. Also we lack dependable and practical equipment to accurately spread manure on soils at the desired rates. We need considerably more research and new models to determine the best application rate for a given situation. Considerable basic research on the soil microbiology associated with manure decomposition is needed to accurately predict availability and release rates of nutrients in manure. We also need to know how decomposition processes are affected by climatic conditions at those times of the year when it is practical to apply manure. In addition, we need to evaluate the effects of manure on concentrations and availability of minor elements in different soils, and we need to define acceptable upper limits for enhanced soil-P availability resulting from repeated manure application.

It is known that manure application results in changes in soil aggregation and tilth, which in turn affect soil, water, and air relationships. It is also known that changes in soil, water, and air relationships affect microbial activity. However, in order to better define these changes and quantify relationships that exist between all these factors, we need greatly improved technology for characterizing these properties and parameters. Factors that affect soil properties and microbial activity also affect the potential losses of nutrients from the soil by leaching, runoff, volatilization, or denitrification. We have very little information for quantifying denitrification losses.

Several other problems associated with feedlot management require additional research. These include the management of understocked or abandoned feedlots where the potential for nitrate leaching is great. Additions of soluble C, such as alcohol, to these sites could possibly denitrify the nitrate present at the sites. Also, as was pointed out in an earlier discussion in this section, we need to develop technology whereby manure can be used with no-tillage systems to maintain residues on the soil surface for erosion control. Likewise, especially in drier regions, we still need to establish soil-loading rates that will prevent undesirable salt buildup. The circulation, amounts, and effects of ammonia gas in the atmosphere near feedlots also require more study.

One could continue for some length on this list of information needed for improved management of beef cattle feedlot manure. The paramount problem, as stated earlier, is to develop methodology whereby one can greatly reduce the loss of nutrients (especially N) from manure into the environment. If these losses are substantially reduced, many of the other factors listed above will be at least partially addressed.

Complementary to the research program outlined above, a corresponding technology transfer program is needed to get the information into the hands of the users. This will require some detailed economic analyses of different situations, which can probably be best addressed through the development of suitable computer models. It is disheartening to see how little use is presently being made of the information that is available, much of which was published 15 to 25 yr ago.

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Chapter 3

Poultry Manure Management

P.A. Moore, Jr., T.C. Daniel, A.N. Sharpley, and C.W. Wood

The recent demand for low-cholesterol meat products has led to tremendous expansion in the poultry industry. In several states this rapid and concentrated growth of the industry has caused increasing concern about the disposal of poultry wastes with respect to nonpoint source pollution. Although poultry litter is one of the best organic fertilizer sources available, excessive applications of litter (as with any fertilizer source) can cause environmental problems. Nitrate leaching into the groundwater, nonpoint source P runoff into surface water bodies, and release of pathogenic microorganisms are three of the main problems encountered with improper management of this resource. The objective of this chapter is to give an overview of the current state of knowledge on the agricultural use of poultry litter and the options available to integrate litter into economically and environmentally sound management systems.

Manure Production and Composition

Poultry production in the United States is concentrated in the midsouth region. Arkansas, Georgia, North Carolina, and Alabama account for over 40 percent of national cash receipts derived from the sale of poultry products; Arkansas leads all states in both quantity and cash value of poultry products. As midsouth states are crucial to national poultry production, levels of poultry production are similarly important to the economic well-being of these midsouth states. Cash receipts from poultry and eggs constituted 45 percent and 51 percent of total 1989 farm income for the states of Arkansas and Alabama, respectively.

Litter associated with broiler production, manure generated from laying operations (hens and pullets), and dead birds are the three wastes of primary concern in poultry production (Edwards and Daniel 1992). Approximately 13 million Mg of litter and manure were produced on U.S. poultry farms in 1990, much of which (45 percent) was generated in Arkansas, North Carolina, Georgia, and Alabama (table 13). Broiler litter accounted for 68 percent of the total fecal wastes generated by the poultry industry in 1990. Although data on amounts of dead birds generated in poultry production are scarce, a 4-percent mortality rate over a production cycle is considered normal for most poultry operations (Edwards and Daniel 1992). Using this rate combined with the data in table 13 and live weights of 0.9 kg bird^{-1} for broilers, 0.9 kg bird^{-1} for layers, 0.7 kg bird^{-1} for pullets, and 5.0 kg bird^{-1} for turkeys (one-half of the live market weights, Sims et al. 1989), we calculated the weight of dead birds requiring disposal on U.S. poultry farms in 1990 to be approximately 270,000 Mg. Commonly used, approved methods of dead-bird disposal include burying in pits, incinerating, and rendering. However, co-composting dead birds with poultry litter (Cummins et al. 1992), an acceptable and desirable disposal method that produces a material amenable to land application, has become popular.

Land application offers the best solution to management of the enormous amounts of manures generated on U.S. poultry farms each year. Depending on the composition of individual poultry manures, these materials can enhance crop production via their capacity to supply nutrients and increase soil quality. Broiler litter is a mixture of manure, bedding material, wasted feed, feathers, and soil (picked up during recovery). Bedding materials are used to absorb liquid fractions of excreta. The type of material used depends on locality, but typically includes wood chips, sawdust, wheat straw, peanut hulls, rice hulls, and recycled paper products. Owing to its relatively low moisture and high macronutrient content (table 14), broiler litter is generally considered to be the most valuable animal manure for fertilizer purposes (Wilkinson 1979). Broiler litter also contains significant amounts of secondary plant nutrients and micronutrients (table 14). Chicken manure without bedding typically has an N content similar to that of broiler litter, but has higher concentrations of water, P, Ca, Mg, and Zn (table 14). It also has a higher proportion of N as ammoniacal-N, which is subject to loss via ammonia volatilization. Turkey litter typically contains similar amounts of N and P compared to the amounts in chicken litter, but has lower concentrations of K (Sims et al. 1989). Dead-bird compost is similar to broiler litter in its nutrient composition, except for its lower N content; N losses are inherent to the composting process (table 14).

Manure Management Systems

Handling systems for poultry manures encompass operations for removing manure from poultry houses,

Table 13. Number of birds and quantity of manure (dry basis) generated from them on U.S. farms in 1990, ranked according to total amounts of manure generated

	Broilers		Layers[*]		Turkeys		Total	
	Number[†] (millions)	Manure[‡] (thousands of Mg)	Number[†] (millions)	Manure[§] (thousands of Mg)	Number[†] (millions)	Manure[‖] (thousands of Mg)	Number (millions)	Manure (thousands of Mg)
Arkansas	951	1,427	15.3	52.8	22.0	239.8	989	1,719
North Carolina	540	810	12.5	53.4	58.0	632.2	611	1,496
Georgia	855	1,282	18.0	55.6	2.0	21.9	875	1,359
Alabama	847	1,270	9.5	34.1	¶	¶	856	1,304
California	231	347	29.0	136.9	32.0	348.8	292	832
Mississippi	413	620	6.1	24.4	¶	¶	419	644
Virginia	297	445	3.4	12.1	17.0	185.3	317	643
Minnesota	41	62	10.2	41.7	46.3	504.7	98	608
Texas	338	507	14.0	50.9	¶	¶	352	558
Maryland	265	398	3.3	8.6	0.1	1.2	269	408
Missouri	88	132	6.6	26.0	18.0	196.2	113	354
Delaware	232	348	0.6	1.5	¶	¶	232	349
Pennsylvania	116	173	18.7	54.3	8.4	91.9	143	320
Oklahoma	142	213	3.7	14.8	¶	¶	146	228
Florida	120	179	11.2	45.1	¶	¶	131	224
South Carolina	84	125	5.7	20.7	5.5	60.0	95	206
Ohio	21	31	17.7	74.1	4.8	51.8	43	157
Tennessee	99	149	1.1	3.6	¶	¶	100	152
Iowa	9	14	8.6	33.3	8.8	95.9	27	143
West Virginia	41	62	0.7	2.0	3.9	42.0	46	105
Oregon	24	36	2.6	11.8	2.3	25.1	29	72
Washington	33	50	5.0	21.5	¶	¶	38	71
Michigan	1	1	5.4	18.5	4.3	46.9	10	67
Nebraska	3	4	5.1	21.8	2.1	22.9	10	49
Wisconsin	14	21	3.4	18.3	¶	¶	17	39
New York	2	4	3.7	12.7	0.5	5.2	7	22
Kentucky	2	2	1.7	6.0	¶	¶	3	8
Hawaii	2	3	0.9	4.9	¶	¶	3	8
Other states	156	233	47.9	182.9	47.1	513.4	251	930
Total	5,966	8,948	272	1,044	283	3,085	6,520	13,078

[*] Includes laying hens and pullets of laying age; pullets of laying age represent 56 percent of the total number produced.

[†] Adapted from U.S. Department of Agriculture (1991).

[‡] Based on 1.5 kg litter $bird^{-1}$ yr^{-1} (Perkins et al. 1964).

[§] Based on 7.00 kg manure $bird^{-1}$ yr^{-1} for laying hens and 1.4 kg manure $bird^{-1}$ yr^{-1} for pullets of laying age (Sims et al. 1989).

[‖] Based on 10.9 kg manure $bird^{-1}$ yr^{-1} (Sims et al. 1989).

[¶] Included in totals for "other states."

Table 14. Chemical properties of broiler litter, chicken manure, and dead-bird compost

	Broiler litter*		Chicken manure*		Dead-bird compost†	
Component	Mean	Range	Mean	Range	Mean	Range
	g kg^{-1} material					
Water	245	20–291	657	369–770	362	217–499
Total C	376	277–414	289	224–328	232	167–270
Total N	41	17–68	46	18–72	18	13–36
NH_4–N	2.6	0.1–20	14	0.2–30	0.5	0.1–1.2
NO_3–N	0.2	0–0.7	0.4	0.03–1.5	0.1	0–0.6
P	14	8–26	21	14–34	12	7–17
K	21	13–46	21	12–32	13	8–20
Cl	12.7	‡	24.5	6–60	‡	‡
Ca	14	0.8–17	39	36–60	20	11–34
Mg	3.1	1.4–4.2	5	1.8–6.6	4	3–7
Na	3.3	0.7–5.3	4.2	2–7.4	‡	‡
	mg kg^{-1} material					
Mn	268	175–321	304	259–378	355	205–600
Fe	842	526–1,000	320	80–560	3,002	807–9,530
Cu	56	25–127	53	38–68	392	48–746
Zn	188	105–272	354	298–388	318	163–539
As	22	11–38	29	‡	‡	‡

* Adapted from Edwards and Daniel (1992).
† Adapted from Cummins et al. (1992).
‡ No data.

pretreating it, and transporting it to the field. The means by which poultry manures are handled are controlled, in large part, by the moisture content of the material.

Solid poultry manure

Most broiler operations result in the production of solid poultry manure, which is referred to as poultry litter or broiler litter. Solid poultry manures contain more than 150 g dry matter kg^{-1}, which makes them amenable to solid waste handling systems (Miner and Hazen 1977). In most states, poultry litter is removed after five or six flocks of broilers, which takes about 1 yr. However, between each flock of broilers, the hard layer of manure that forms at the surface (referred to as "cake") is removed using a "decaker." This implement, which is pulled behind a tractor, lifts the litter off the floor, sifts it through a large mesh screen, and removes large (diameter of greater than 2.5 cm) particles. This material is then applied to land or is used in dead-bird composters as the manure source.

A total cleanout of poultry litter from production houses is typically accomplished with tractor-mounted box scrapers or blades and machinery capable of scooping the material, such as front-end loaders. Upon removal from poultry houses, this material may be directly applied to land or temporarily stored. Manure storage prior to land application, which may occur under roofed structures (dry-stack barns) or well-secured impermeable tarpaulins, allows flexibility in timing of land application (Brodie and Carr 1988). Flexibility in timing of spreading is important for synchronization of plant nutrient needs with nutrient release from poultry manure, which lessens the risk for environmental contamination when these materials are applied to land. Moreover, dry storage reduces the risk of environmental contamination as compared to the risk associated with leaving manure piles exposed.

When solid poultry manures are stored, particularly under roofed structures, they can be subjected to treatments aimed at enhancing their spreading characteristics, maintaining their nutrient composition, or altering their chemical and biological properties via composting. Solid poultry manures that are wetter than normal can be dried via static aeration or by mixing with drier materials, and this drying may be desirable from a weight-reduction or spreading perspective. Drying is particularly desirable if solid poultry manures are to be transported long distances. However, mechanical drying (using fans and/or dryers) of these materials is rarely practiced. During handling of solid poultry manure, considerable N loss from ammonia volatilization can occur. Additions of water-soluble phosphate fertilizers (excluding ammonium phosphates), which react with ammonia in manures to form ammonium phosphates, have been put forward as a means to conserve N (Mitchell et al. 1990). Additions of water-soluble phosphates to solid poultry manures increases the P concentration of the manure, which may be undesirable from an environmental perspective. Additions of aluminum sulfate to litter is probably the best method of avoiding ammonia volatilization (Moore et al. 1995a, 1996). This practice would not only decrease ammonia volatilization, it would decrease P runoff as well.

Runoff of dissolved P from fields receiving poultry litter can occur, even when best management practices (BMP's) are used. The reason for this is that poultry litter contains high concentrations of water-soluble P (often in excess of 2,000 mg P kg^{-1}). This P fraction is readily transported in runoff water during intense rainfall events.

Recent work has shown that the level of water-soluble P in litter can be reduced by several orders of magnitude with the addition of flocculating materials commonly used in wastewater treatment and lake restoration. Moore and Miller (1994) showed that water-soluble P levels decreased from around 2,000 mg P kg^{-1} to less than 1 mg P kg^{-1} litter with the addition of aluminum, calcium, or iron compounds, such as alum, slaked lime, and ferrous chloride. These compounds not only reduce water-soluble P concentrations but also decrease suspended solids, biological oxygen demand, heavy metals, bacterial counts, virus viability, and parasites. Field studies on the effects of chemical amendments to litter have shown that treatment of poultry litter with aluminum sulfate reduces P runoff by as much as 87 percent, compared to normal litter (Shreve et al. 1995). Tall fescue yields were also found to be significantly higher when litter was treated with aluminum sulfate (Shreve et al. 1995).

Composting, which occurs naturally when nonsterile organic substrates are combined with water and oxygen, may be a desirable treatment for poultry manures or carcasses. In the composting process, aerobic microbial decomposition generates sufficient heat energy to raise the temperature of compost mixtures to the thermophylic zone (40 to 75 °C), destroying pathogenic organisms and weed seed as temperatures surpass 60 °C. Composting reduces the volume and weight of original organic substrates, and the end result of successful composting is a material that is biologically stable, odor free, and useful as a potting medium and soil amendment.

Liquid poultry manures

Liquid poultry manures (those containing less than 150 g dry matter kg^{-1}) are generated when manure is scraped or flushed into storage reservoirs, such as tanks, detention basins, aerobic or anaerobic lagoons, and oxidation ditches. Most of the liquid poultry manure is generated in laying-hen operations. Although these materials are generally amenable to hydraulic pumping, those containing between 40 and 150 g dry matter kg^{-1}, referred to as slurries, can present problems to pumping equipment because of their viscosity and potential to plug orifices (Miner and Hazen 1977). Solid-liquid separation via sedimentation or filtration may be necessary when liquid poultry manures with higher amounts of solids are to be pumped.

Although storage in reservoirs often serves to enhance hydraulic properties of liquid poultry manures with regard to ease of pumping, this can result in considerable loss of plant nutrients, particularly N. Ammonia volatilization losses from storage reservoirs range from 25 to 80 percent of original N contained in liquids or slurries (Tisdale et al. 1985). Nitrogen losses are minimized when the liquids or slurries are added to the bottom of storage reservoirs instead of to the surface (Loehr 1984).

Land Application of Manure

Except for small amounts used in animal feed, the major portion (greater than 90 percent) of poultry litter is applied to agricultural land (Carpenter 1992). This application usually occurs no more than a few miles from where the manure was produced. Thus, in states with a large or growing poultry production industry, increasing demands are being imposed on agricultural acreage to efficiently use the nutrients (primarily N and P) contained in manure. In the major poultry producing states, the amounts of nutrients produced in manure exceed crop requirements. Data compiled by National Agricultural Statistics Service (1989), indicate that the amount of P produced annually in poultry manure exceeds that required by the three major crops in several poultry producing states (fig. 12).

Poultry production is often concentrated in regions with small farms, which have very limited acreages for land application. While poultry production provides a fairly good income for these small farmers, problems created by manure use may have major environmental consequences.

Transportation

Generally, transportation of poultry litter is restricted to less than 10 to 20 km. Obviously, being able to transport the manure greater distances from the source of production increases the acreage for application. Assuming poultry litter contains respective N and P contents of 3.4 and 1.7 percent (dry-weight basis), a farmer would have to add 5 times as much poultry litter as 17–17–17 fertilizer to achieve the same N and P application rate.

Transport of solid poultry manure to the field, depending on the distance, is typically done with spreader trucks. Liquid poultry manures (slurries) may be pumped from storage reservoirs into tank-bearing vehicles for transport to the field, which requires agitation (Miner and Hazen 1977). Liquid poultry manures having less than 40 g dry matter kg^{-1} may be handled in the same manner as slurries or may be pumped directly from storage reservoirs though pipeline systems to irrigation equipment at the site of application.

The cost of moving poultry litter is a major obstacle facing the more efficient use of this resource. The recent trend of several neighboring farmers to form cooperatives to compost and compact manure more cost effectively should be encouraged by cost-sharing programs. By composting and compacting, the bulk density of the litter is increased, which reduces the cost of transportation. However, for this to be cost effective, the nutrient content of the litter should be high. Since composting can result in N loss, growers may have to add compounds, such as aluminum sulfate, to the litter to reduce ammonia volatilization during this process.

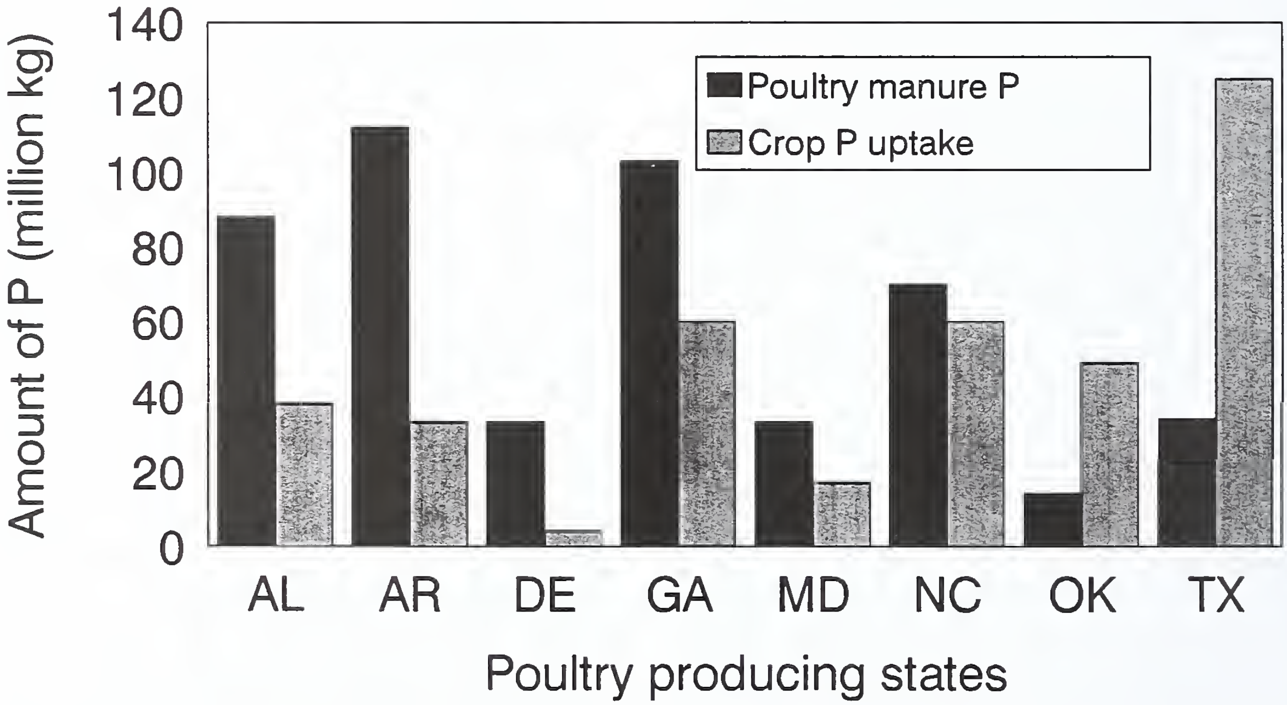

Figure. 12. Amount of P produced in poultry manure and taken up by the three major crops in several poultry producing states in 1988 (adapted from National Agricultural Statistics Service 1989)

Spreading equipment

The type of spreading equipment used depends on the method of storing and handling poultry manure. Traditionally, poultry litter is broadcast directly from the house, using a variety of spreaders. Manure stored in deep pits is removed by scraping and is applied with a spreader. In a few cases, manure stored in shallow pits is removed by flushing and, after large solids have been removed by sedimentation and/or filtration, is applied with an irrigation system. Spreading equipment can vary among contractors. In many locations where the poultry industry has recently expanded, existing farm equipment is used to apply the manure. There has been less progress in improving spreading equipment for solid manure than for liquid manure. Equipment development should involve better control of the application rate and provide even distribution of manure.

Available land base

In states where the poultry industry and/or confined animal operations are concentrated, the land base available for manure application is often limited. This limitation mainly arises from the cost of manure transportation. Consequently, poultry manure is usually applied in the immediate vicinity of the production site, with little regard to the geology, soils, or topography. This inflexibility may result in the application of litter to areas with elevated soil N and P contents from previous applications or with high runoff or leaching potentials. Consequently, in the future, recommended manure application rates should be flexible and account for differing geology, soil, and topography of potential application sites.

Proliferation of the poultry industry has been economically driven. Numerous farmers with limited resources have turned to poultry production as a ready source of income with limited cash outlay. In many areas of the southern United States, intensive poultry production has developed on agricultural land unable to maintain high crop yields due to such factors as erratic weather, sloping topography, or soils that are rocky, shallow, coarse textured, or highly permeable. Local need for N and P in such regions would be lower than in areas of intensive crop production.

The current land base for manure application is dwindling. High transportation costs for manures have led to repeated applications on fields immediately surrounding poultry farms, resulting in a buildup of N and P in soils, particularly P. Manure applications to these soils may be based on soil test P requirements rather than on crop N requirements. Currently, most manure application rates are based primarily on the management of N to minimize nitrate losses by leaching. In most cases this has led to an increase in soil P levels after successive poultry manure applications because most crops require a higher N:P ratio than that supplied in poultry manure. For example, poultry litter has an average N:P ratio of 3 (table 14), while the N:P requirement of major grain and hay crops is 8 (White and Collins 1982). Soils receiving repeated applications of poultry litter for several years accumulate more P then N and have more P than the crop can use (Sharpley et al. 1991a, Sims 1992, Wood 1992).

Basing litter application rates on soil P levels rather than on crop N requirements may mitigate the excessive buildup of soil P and at the same time lower the risk for nitrate leaching to groundwater. However, such a strategy for determining proper litter rate would eliminate much of the land area with a history of continual litter applications, since many years are required to lower soil P levels once they reach excessive levels (Kamprath 1967, Wood 1992). In addition, farmers relying on poultry litter to supply most of their crop N requirements will have to purchase commercial fertilizer N instead of using their own manure N. Although basing rates on soil test P may resolve potential environmental issues, it places unacceptable economic burdens on farmers, that is, the cost associated with transporting the manure and buying additional fertilizer N are too high.

Hydrology of the available land base will also be important in determining whether manure application rates should be based on N or P. If the potential for leaching of soluble chemicals from an application site exists, one could argue that N should be a priority management consideration. Conversely, if runoff and erosion potential far exceed leaching potential, then P would be the main element governing application rates.

As the poultry industry continues to grow in areas where poultry production is already high and where the land base suitable for agronomically and environmentally sustainable manure applications continues to decline, manure will, by necessity, be moved outside of these intense poultry producing areas. Research in Alabama, Arkansas, and Oklahoma is evaluating appropriate application rates and cultural practices for poultry litter as a nutrient source for field crops (corn, cotton, rice, sorghum, and wheat) and bermudagrass (coastal and midland). The major obstacle to using this manure on these crops in non-poultry-producing areas continues to be the cost of transport.

Tillage effects

Application of poultry manure before or during tillage will reduce surface soil accumulation of added N and P and increase distribution of these nutrients in the root zone. If a ground cover can be maintained during times of the year when runoff-producing rainfall is common, environmental risks will be reduced while crop use of N and P will be increased. Preliminary research in Arkansas and Oklahoma using simulated rainfall on soil receiving poultry manure indicated that soil incorporation of manure with tillage reduced N and P loss in surface and subsurface runoff compared to broadcast applications. This effect was attributed to a dilution of manure N and P in the depth of tilled soil.

However, there are two main disadvantages to tilling manure into the soil. First, the time frame for manure application will be restricted to the time frame needed for tillage operations. Second, labor requirements in the short time available for seedbed preparation are increased and can sometimes delay sowing and increase weed problems.

The use of manure on grassland without tillage can be reasonably efficient, especially in areas with a humid climate. This is probably because grass species can use N and P from the manure throughout the whole growing season.

Soil and manure testing

There are many variables associated with poultry management systems that can affect manure quality at the time of application. These include the type and amount of bedding material used, accumulation time, feed, amount and quality of water used to flush the house, location in a storage pit from which the manure is removed, and length of storage before land application. All of these factors can have a big effect on the nutrient composition of the manure applied (Edwards and Daniel 1992). As a result, farm advisors and extension agents in several states are recommending that the N and P composition of both manure and soil be determined by soil test laboratories before manure is applied to land. These tests should be helpful to farmers because there is a tendency to underestimate the nutritive value of manure. Thus, manure analyses are a constructive educational tool for farmers, showing them that manure represents a valuable source of N and P.

In those states where manure analyses are conducted, total N, NH_4–N, and moisture content are generally determined in the analyses. With the use of more sophisticated analytical equipment allowing multi-element analysis in soil test laboratories, total P, K, and other nutrients can also be determined and reported to the farmer upon request. Since most of the N and P in poultry manure is in organic forms (Edwards and Daniel 1992, Wood and Hall 1991), much of the N and P is not immediately available to plants. Thus, for maximum crop production, N and P application rates based on total nutrient content may need to be greater for manure than inorganic sources.

Manure application based on total nutrient content should be adjusted to account for nutrient availability in the soil. Nitrogen availability is related to mineralization of organic N (usually 50 to 60 percent of the organic N fraction) and recovery of added NH_4–N. This availability may be adjusted further to account for the effect of storage time on N mineralization and volatilization and of soil type on NH_4–N fixation. It is generally assumed that 75 to 80 percent of added total P and all of the added K is plant available. A cautionary note to basing application rates on manure analyses must be sounded because of the wide variability in nutrient contents that can be obtained. For example, variabilities associated with sampling the manure alone can be 10 to 15 g N kg^{-1} manure. This could amount to a 25 percent overestimation or underestimation of N content. Thus manure analysis should be used as a guideline only.

Current soil test methods represent, for the most part, plant available inorganic N and P levels in soil. Because of the high organic N and P content of manure, soil test recommendations for manure application rates must account for the mineralization of organic nutrients during the growing season. In addition, poultry manure can provide plant-available N and P for several years after application. Thus, soil tests must also account for the residual effects of poultry manure, possibly resulting in a reduction in application rates in years following initial applications. In many instances it is difficult to account for differences that are due to variable soil, climate, and cropping conditions encountered.

Cost-effective best management practices (BMP's)

Poultry manure is a valuable natural resource if managed properly. In many areas of intensive poultry production, manure applied on hilly land has increased vegetative cover, thereby reducing runoff and erosion potential. These unproductive soils would not normally receive mineral fertilizer; thus, the careful use of poultry manure can reduce environmental degradation.

Before poultry manure is used, the producer should consider which BMP's are needed, based on the crop being grown, timing of application, land base available, and previous applications. Crop type and yield will affect the amount of N and P removed from the production system when the crop is harvested (fig. 13). Obviously, the accumulation of manure N and P within an agricultural system will be reduced if the nutrients are removed from the farm in the harvested crop.

Alternative Uses of Poultry Litter

Poultry litter, when mixed with feed grains, has been used as a successful feed for cattle. Approximately 4.2 percent of the poultry litter produced in the United States is fed to cattle (Carpenter 1992). In some states, high-quality poultry litter (20 percent crude protein and less than 10 percent ash) can be worth as much as \$99 Mg^{-1} as feed whereas the same litter may only be worth \$33 Mg^{-1} as fertilizer (Payne and Donald 1992).

Although disease problems have not been reported from feeding manures to animals under acceptable conditions, copper toxicity has been reported to be a problem in sheep (Fontenot et al. 1971). The poultry litter contained 195 mg Cu kg^{-1} because the chickens had been fed a diet containing high levels of copper sulfate. Currently, most poultry producers feed their broilers an excess of copper sulfate. Although this excess results in faster weight gains, the gains are not due to a change in diet per se, but rather to a change in litter composition (Johnson et al. 1985). There are two possible explanations for this phenomenon: (1) the high copper levels in the litter reduced populations of pathogenic microorganisms or (2) nonbiologically mediated reactions, such as ammonia volatilization, are affected. It should be noted that not all broilers respond positively to this excess of copper in the diet.

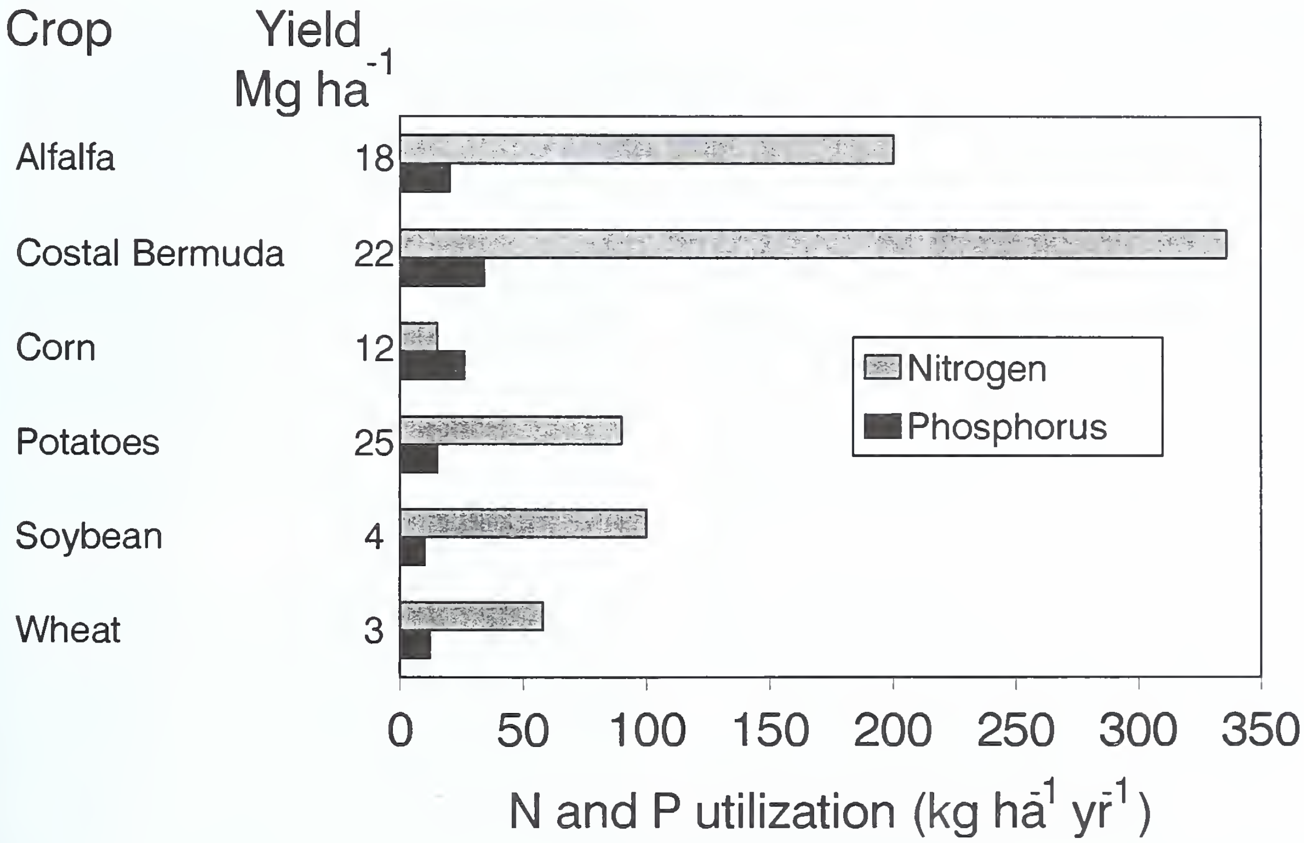

Figure 13. Approximate annual N and P use by several crops (adapted from White and Collins 1982)

Wideman et al. (1995) showed that high copper levels in broiler diets can lead to proventriculitis, a malady characterized by necrosis and enlargement of the proventriculus (glandular stomach). When these birds are processed, the proventriculus ruptures easily during evisceration, contaminating the carcass with stomach contents.

It is important to remove any foreign materials such as wire, plastic, or glass from the litter before it is used for feed. It is also important to maintain a low ash content. When large quantities of soil are removed with the litter, the ash content increases dramatically. Litter with ash contents exceeding 28 percent should not be fed to cattle.

Poultry litter can also be sold to nurseries and garden stores as an organic soil amendment for homeowners. However, at present the amounts sold in this manner represent much less than 1 percent of the total litter produced. Poultry litter may also be used to produce electricity. A power station using poultry litter became operational in Suffolk, England, in 1992. The power plant cost approximately $35 million and uses 10,000 Mg of litter per year from the area's poultry farms.

Agronomic and Environmental Effects of Poultry Manure Application

Effects on soil properties

In addition to providing nutrients for crop production, poultry litter applications build soil organic reserves. The organic matter benefits crop production via increases in soil water-holding capacity, water infiltration rates, cation exchange capacity, structural stability, and soil tilth. Weil and Kroontje (1979) found that high rates of poultry manure, when incorporated into the soil, resulted in decreases in bulk density and increases in water-holding capacity and water-stable aggregates. Kingery et al. (1993) showed that litter applications resulted in increased organic C and total N to depths of 15 and 30 cm, respectively.

Metals, such as As, Cu, and Zn, are often fed to poultry. This results in average concentrations in the litter of 22, 56, and 188 mg of the three metals kg^{-1}, respectively (table 14). Kingery et al. (1993) found elevated levels of K, Ca, Mg, Cu, and Zn in soils heavily fertilized with poultry litter. Elevated levels of heavy metals in the soil will result in increased uptake by plants, which will be consumed by animals or man. However, normally concentrations do not reach toxic levels. High levels of heavy metals, particularly copper, in the water-soluble fraction of litter can also lead to high concentrations of these metals in runoff water from pastures fertilized with poultry litter. Moore et al. (1995b) found that treating poultry litter with aluminum sulfate significantly decreased heavy metal concentrations in runoff water from tall fescue plots fertilized with poultry litter.

Effects on soil fertility

Poultry litter is generally considered the most valuable of animal manures for use as a fertilizer, due mainly to its low water content. As mentioned earlier, poultry litter contains large amounts of N, P, and K as well as secondary and trace elements. Under certain conditions, however, various salts can build up from excessive poultry litter applications. Soil salinity attributed to poultry litter applications has occasionally been shown to reduce germination and growth of corn (Shortall and Liebhardt 1975, Weil et al. 1979). However, it should be pointed out that poultry litter has long been recognized as an ameliorant to salt-affected soils. Research by Hileman (1973) showed that poultry litter promotes growth on brine-contaminated soils in south Arkansas.

Stephenson et al. (1990) found that the average fertilizer equivalent of poultry litter was 3–3–2 (3 percent N, 3 percent P_2O_5, and 2 percent K_2O) when determined on an "as spread" basis. Poultry litter also contains substantial quantities of B, Ca, Cu, Fe, Mg, Mn, S, and Zn.

Nutrient imbalances in forages due to excessive poultry litter applications have been observed. Grass tetany in ruminants, which is related to the ratio of K to Ca plus Mg in forages, appears to be more likely on soils that receive excessive rates of poultry litter (Wilkinson et al. 1971), possibly due to high K levels in the litter. Therefore, application rates for poultry litter should be limited to 9 Mg ha^{-1} or less for use on tall fescue.

Poultry litter can also be a valuable amendment for rice soils that have been leveled by grading. Miller et al. (1991) showed that rice yields increased as much as 286 percent with poultry litter additions. Although they saw some yield responses when inorganic N, P, K, S, and Zn fertilizers were added at the same rate, these responses did not match those resulting from poultry litter.

Effects on water quality

The customary method of land application for poultry litter is broadcasting without incorporation. However, the same nutrients that make poultry manure a good fertilizer can, under some circumstances, be detrimental to the quality of groundwater and downstream surface water. The potential for water quality degradation from nutrients responsible for eutrophication (N and P), oxygen consumption (organic carbon), and metal toxicities is of particular interest in areas such as northwest Arkansas, where shallow, cherty soils and karstic geology greatly increase the interaction between surface water and groundwater.

One of the primary health concerns with excessive poultry litter applications is nitrate leaching into the groundwater. The U.S. Environmental Protection Agency limits nitrate concentrations in drinking water to 10 mg NO_3–N L^{-1}. Liebhardt et al. (1979) found that excessive applications of litter to corn resulted in nitrate leaching through the profile and elevated nitrate levels in groundwater. Ritter and Chirnside (1982) indicated that 32 percent of the water wells in Sussex County, Delaware, had high nitrate levels (>10 mg N L^{-1}) due to improper poultry litter applications. Kingery et al. (1993) found that long-term applications of poultry litter at relatively high rates resulted in a buildup of nitrate in the soil to a 3-m depth or to bedrock (fig. 14).

From a surface water viewpoint, P is the element of primary concern, since it is generally considered to be the limiting nutrient for eutrophication. Excessive applications of poultry litter to soils result in a buildup of P near the soil surface. Kingery et al. (1993) observed soil test P levels as high as 225 mg P kg^{-1} soil in soils that had long-term applications of poultry litter at relatively high rates (fig. 14).

In a similar study of continual long-term poultry litter application to 12 Oklahoma soils, Sharpley et al. (1993) found that P accumulated in the surface meter of treated soil to a greater extent than N. This reflects the differential mobility, sorption, and plant uptake of N and P in soil.

The movement of soluble and sediment-bound (particulate) P can be predicted using kinetic and enrichment-ratio approaches. Sharpley and Smith (1993) used these approaches to estimate the P concentration

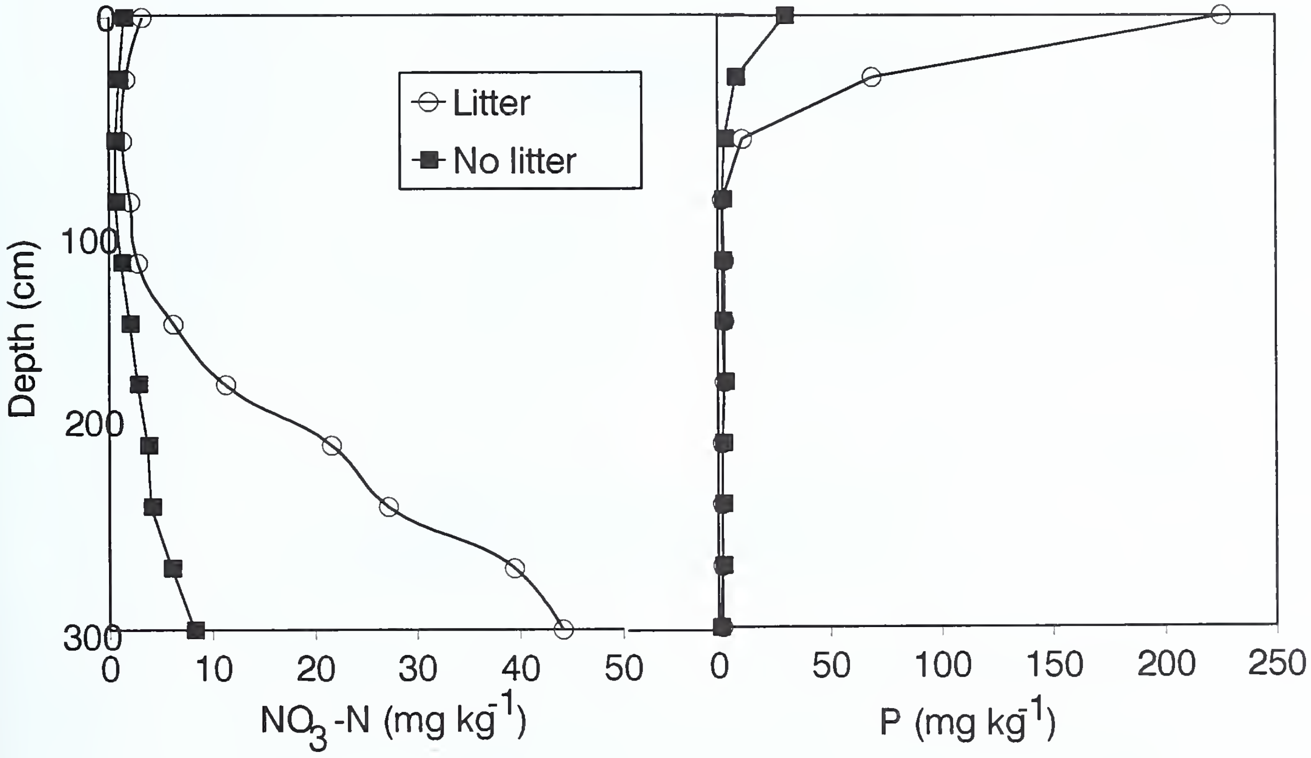

Figure 14. Average soil NO_3–N and extractable P concentrations for 12 pasture pairs in the Sand Mountain Region of Alabama that have received either long-term applications of broiler litter or no litter (Source: Kingery et al. 1993)

in runoff water from a 2.5-cm runoff event (10 kg ha^{-1} yr^{-1} soil loss) for grasslands in Oklahoma. Predicted P concentrations in runoff water from three soils treated with poultry litter were much greater than from untreated soils (fig. 15). On grasslands, erosion is minimal, and thus about 80 percent of the P is transported in a bioavailable form (soluble and NaOH-extractable particulate P available for algal uptake). These concentrations are approximately two orders of magnitude greater than values associated with eutrophication (0.01 and 0.02 mg P L^{-1} soluble and total P, respectively) (Sawyer 1947, Vollenweider and Kerekes 1980). The potential increase in P transport in runoff highlights the need for careful management of surface soil accumulations of P as a result of poultry litter applications on soil susceptible to runoff and erosion.

In Tennessee, Green and Bucham (1992) sampled well water on poultry farms and found that 43 percent of the wells sampled contained fecal coliform bacteria and 8 percent of the wells exceeded 10 mg NO_3–N L^{-1}. They found that well location was an important factor with regard to contamination and indicated that wells should be at least 15.2 m from chicken houses and 30.4 m from stacked broiler litter.

Poultry wastes are known to contain many pathogens, which could potentially contaminate both surface water and groundwater resources. Alexander et al. (1968) tested 44 poultry litter samples for the presence of pathogens. They found 10 different species of *Clostridium*, 3 species of *Salmonella*, 2 species of *Corynebacterium*, 1 species of yeast, and 1 species of *Mycobacterium* (which is occasionally responsible for tuberculosis) in various litter samples. All of the litter samples contained Enterobacteriaceae (other than *Salmonella*), *Bacillus* spp., *Staphylococcus* spp., and *Streptococcus* spp. In Arkansas, the Nation's leading poultry producing state, 90 percent of the surface water bodies (statewide) sampled by the Arkansas Department of Pollution Control and Ecology con-

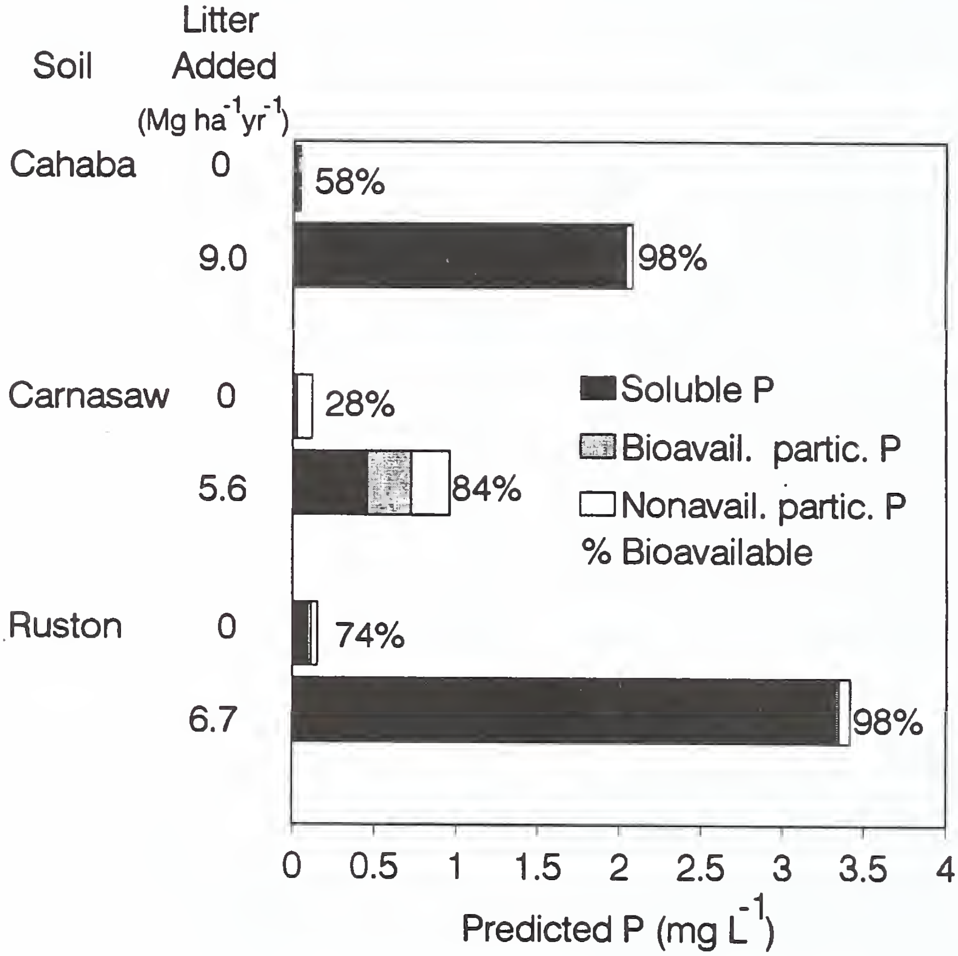

Figure 15. Predicted soluble, bioavailable particulate, and nonavailable particulate P in runoff from grasslands receiving poultry litter. Litter applications were made annually for 12 years to the Cahaba and Ruston soils and for 20 years to the Carnasaw soil.

tained fecal coliform counts in excess of the primary contact standards. However, fecal coliform counts prior to the rise in poultry in this state are not available. Therefore, it is unknown whether these levels are indigenous or, in fact, due to runoff from animal manures.

Viruses have also been reported in poultry litter and may represent a greater threat to water resources than bacteria. These include viruses responsible for newcastle disease and Chlamydia (Biester and Schwarte 1959). At present, very little information on virus runoff from fields receiving poultry litter is available.

Effects on air quality

The number one complaint against animal growers received by state and Federal environmental regulatory agencies involves odor problems (Williams 1992). Much of the odor is due to high levels of ammonia. Volatilization of ammonia results in decreased poultry productivity due to an increase in the incidence of ascites and other respiratory related maladies, such as newcastle disease. Ammonia volatilization also results in tremendous losses of N that could otherwise be used for fertilization of pasture or cropland. Wolf et al. (1988) found that 37 percent of the total N applied on the surface of a pasture was lost via volatilization after only 11 days. With the inclusion of in-house losses, this figure would increase to well over 50 percent of the total N. Another reason ammonia volatilization is detrimental is the negative impact it has on the environment with respect to acid rain (van Breemen et al. 1982, Ap Simon et al. 1987). Another air pollution problem aggravated by ammonia loss from poultry manure is the formation of airborne particles of NH_4NO_3, which contribute to PM_{10}s (particulate matter less than 10 μm in diameter).

The human nose can detect atmospheric ammonia concentrations as low as 5 $\mu L\ L^{-1}$, and some people are susceptible to eye irritation at levels as low as 6 $\mu L\ L^{-1}$. Currently OSHA has not set exposure levels for U.S. poultry workers; however, in Europe the COSHH (Control of Substances Hazardous to Health) has determined exposure limits to humans at 25 $\mu L\ L^{-1}$ for an 8-hr exposure and 35 $\mu L\ L^{-1}$ for a 10-min exposure (Williams 1992). As mentioned earlier, aluminum sulfate has been shown to be extremely effective in reducing ammonia volatilization from poultry litter (Moore et al. 1995a, Moore et al. 1996).

Effects on crop production

Poultry litter and manure have increased yields in many different crops, such as bermudagrass, corn, fescue, orchardgrass, rice, and wheat (Miller et al. 1991, Edwards and Daniel 1992, Wood 1992). Most of the yield increases have been reported to be due to the N content of the litter; however, the response in rice on graded soils that occurs in Arkansas cannot be duplicated with inorganic N, P, K, S, and/or Zn (Miller et al. 1991).

Improving Management of Poultry Manure

Education and technology transfer

Technology transfer in production agriculture has become a fairly familiar process. For example, if a new herbicide is developed, it will undergo field testing by industry and universities, and if proven successful, information on the herbicide will be made available through a variety of mechanisms, including field days, extension brochures, industry field personnel, published journals, and other outlets. A tried-and-proven infrastructure exists for getting the proper information to the potential user in an efficient and timely manner. Equally important, most everyone is aware of the target audience—in this case the growers.

The infrastructure for transfer of technology relating to nonpoint source pollution, especially with regard to poultry waste management, is not as highly developed as that for production agriculture. Researchers in this area can and should become involved in the technology transfer process. For proper planning and conduct of research, the researcher should have input and an ongoing dialogue with every player, including industry personnel, state and Federal agencies, and ultimately the grower. As information is generated, these same players must be informed of developments. The initial target audience for this information is the professionals working in the water quality area, especially those agency professionals deciding which practices will be identified as a BMP. The first step for researchers is to establish scientific credibility of their work by publishing it in journals and presenting it at scientific meetings. Concomitant with the first step, this same information needs to be repackaged and transferred to state and Federal agency personnel working in the water quality area. Information transfer to this group may take several avenues, including workshops, brochures, and seminars. A parallel process needs to occur with representatives of the poultry industry and selected growers. This is a necessary, time-consuming, and dynamic process of identifying a series of BMP's.

Ultimately, the information must be disseminated to the end user or grower. The USDA's NRCS and Cooperative State Research Education and Extension Service provide the critical link between farmers and public agencies. The Extension Service has the primary responsibility of information dissemination to farmers. The USDA–NRCS is the technical arm at the county level that incorporates the BMP's into the farm plan.

Best management practices (BMP's)

The concept of BMP's was introduced in Public Law 92–500, which outlined several rigorous requirements for a practice to qualify as a BMP. The BMP must relate directly to water quality and must be cost effective. This requirement makes it necessary to place a dollar value on water quality. For example, the benefits of a practice that controls animal manure runoff near a trout stream are easier to evaluate than the benefits of implementing the same practice near a less sensitive water resource. Until alternative methods are developed, the process for assessing benefits and cost effectiveness will continue to be decided on a case-by-case basis. Other requirements of BMP's are that they must be acceptable to the grower and must provide economic returns to the grower (otherwise volunteer adoption will be low).

Adverse impacts resulting from land application of poultry manure may be prevented by implementing of effective BMP's. Examples of recommended BMP's include using buffer zones between treated areas and waterways, adding aluminum sulfate to litter between successive flocks of birds to precipitate soluble phosphate, applying litter when there is a low likelihood of rainfall in the near future, and incorporating litter when conditions permit.

Most specialists will agree that implementation of a combination of practices adopted as "best" will, in fact, have a positive effect on quality of runoff from areas treated with poultry litter. However, it is often difficult to determine the effectiveness of individual practices because supporting data can be limited. A lack of data on BMP effectiveness makes it difficult to quantify the water quality effects of BMP implementation and may therefore cast doubt on the appropriateness of policies and the recommendations developed by decision-making organizations.

BMP's are available now that will protect and maintain water quality; others are in the process of being developed and field tested. Some of the recommended practices were initially used for erosion control and have been around for some time, while others are new and were designed specifically for protecting water quality.

These practices focus on controlling the problem at the source, rather than after entry into the aquatic system. Example practices include limiting manure application rates, applying manure only on certain slopes, and applying manure only at a certain time of year. Runoff losses of soluble P are affected by land application of commercial fertilizer and animal manure, and the amount lost in the runoff is directly related to how the materials are managed (Baker and Laflen 1982, Logan 1991). These losses are often linearly related to application rate, with the greatest losses of P occurring when the fertilizer or manure is broadcast and not incorporated (Baker and Laflen 1982, Westerman et al. 1983, Mueller et al. 1984). Rainfall intensity and soil type were also shown to significantly affect the amount of total solids transported. McLeod and Hegg (1984) investigated impacts of different fertilizers (organic and commercial) on runoff quality and reported minimal nutrient losses (less than 4 percent of the total Kjeldahl N and less than 2.5 percent of the total P). The highest nutrient losses occurred on plots treated with commercial ammonium nitrate. Giddens and Barnett (1980) showed that high application rates of poultry litter drastically reduced the volume of runoff water and soil erosion but increased the coliform bacteria in the runoff.

Timing manure applications to coincide with maximum crop uptake and minimum runoff potential will enhance crop use of manure. In Arkansas, computer simulations have shown that windows for optimal timing of application of manure exist (Edwards et al. 1992). However, demands on farmer's daily schedules and use of independent contractors often limit the practicality of precise timing of manure applications. As a result, application timing is possibly the greatest obstacle to better manure management, with many BMP's needing to be done at the busiest times of the year for farmers.

Moving poultry litter to areas where soil N and P levels are low would not only improve crop production but would decrease the likelihood of environmental problems associated with excess litter. In Arkansas, the poultry industry is concentrated in the northwest section of the state in the Ozark Highlands. However, most of the row crop agriculture is located in the eastern portion of the state in the Mississippi Delta. Transporting the litter from the Ozarks to the Delta appears to be one solution to the current problem. However, the cost of transportation is prohibitive unless the government or the industry provides subsidies for such a program.

Various investigators have shown that the level of soil test P also influences the concentration and eventual loss of soluble P in runoff. In fact, a highly significant linear relationship has been demonstrated between the level of soil test P in the surface soil and soluble P concentration of surface runoff (Hanway and Laflen 1974; Romkens and Nelson 1974; Sharpley et al. 1978, 1981; Oloya and Logan 1980). One BMP that several states have implemented is an upper cutoff level for soil test P, above which the grower would not apply P from any sources, including animal manures.

Buffer strips, also referred to as vegetative filter strips and buffer zones, have a proven record of effectiveness in reducing nutrient runoff from fields fertilized with manure. For example, buffer strips installed below cattle feedlots have proven effective in reducing transport of both N and P. Doyle et al. (1977) found that a 4-m fescue buffer zone reduced concentrations of dissolved P by 62 percent and nitrate by 68 percent. Young et al. (1980) observed total N and P reductions of 88 percent and 87 percent, respectively, for a 30-m orchardgrass buffer zone. A sorghum-sudangrass mixture buffer zone performed similarly, with 81 percent and 84 percent reductions in total N and P, respectively. Chaubey et al. (1995) found that 21.4-m vegetative filter strips reduced the mass transport of TKN, ammonium, TP, and PO_4-P by 81, 98, 91, and 90 percent, respectively, from plots fertilized with poultry litter.

Another BMP that has been shown to reduce nutrient runoff from fields fertilized with poultry litter is the addition of aluminum sulfate (alum) to the litter. Alum additions to manure reduce soluble P levels in the litter (Moore and Miller 1994), which results in significantly lower P concentrations in the runoff water (Shreve et al. 1995). Alum also reduces ammonia volatilization from poultry litter, resulting in higher N concentrations in the litter and therefore contributing to significantly higher crop yields (Moore et al. 1995a, 1996; Shreve et al. 1995).

Soluble P in soils that test high in P can also be reduced using chemical amendments. Peters et al. (1995) found that soluble P levels in soils that had received excessive manure applications could be reduced with the addition of alum sludge (a waste product from drinking water treatment plants), bauxite red mud (a waste product from aluminum mining), and cement kiln dust.

Program implementation, agency interactions, costs, and benefits

Ensuring compatibility between poultry manure use and water quality requires a continued and long-term commitment from industry, citizens, and public agencies. To assure a favorable cost-benefit ratio, priority watersheds should be selected to focus sparse implementation funds and expertise. Such watersheds can be selected on a regional, state, or local basis. The criteria for selection should be based on severity of the problem and the benefit to water quality. The complexity of the issue means that management programs will not be easy to establish or maintain. It is also clear that the concept of zero discharge is not workable. In many cases, we may only be able to maintain lakes and streams in their present state and not improve their water quality; we can simply keep them from deteriorating further. The inherent fertility of other aquatic systems may have progressed to such an extent that no improvement is guaranteed regardless of funds expended.

Although BMP's are being developed for dealing with poultry manure, institutional mechanisms for implementing this technology need improvement. In the past, cost-sharing programs generally focused on support of production practices, but recently many programs also support practices that protect water quality. Changing the tax laws is another approach that might accelerate implementation of environmental technology. Voluntary adoption and dissemination of new technologies that protect water quality will only be possible if agricultural producers can be convinced that the adoption of these practices is in their best interest. Dissemination of information on the relative profitability of management options and the importance of agriculture's role in water quality protection will be essential. The successful design of environmentally sound management practices must be coordinated with the institutional mechanism developed to promote adoption. Successful programs will emphasize management, control of the problem at the source by implementation of BMP's, and, perhaps most of all, informal planning sessions between the USDA–NRCS field technician and the grower to produce a field-by-field farm plan that protects water quality.

Sociological benefits

As the human population continues to grow, ever increasing strains are placed on natural resources. Recently, there has been an increased awareness of the pressures being placed on the environment from human activities. Sustainable agriculture appears to be one important means by which we can minimize the impact of food production on the environment. The use of animal manures for fertilization of crops will decrease the amount of inorganic fertilizers needed. This will conserve fossil fuels that are needed to produce these products and should also improve the fertility status of soils by providing a well-balanced fertilizer and by increasing soil organic matter. Also, if more nutrients in manure are recycled through agricultural crops, less of these nutrients will escape to the environment and result in environmental degradation.

Research Needs for Poultry Manure Management

Historically, strategies for land application of animal manures have been based on meeting the N needs of the crop being produced. Although this approach can be justified on the basis of groundwater protection, there is little basis for this approach for surface water protection, since eutrophication of freshwater systems is normally limited by P. Therefore, the question as to whether poultry litter applications should be based on P loading, rather than N loading, has arisen. Research is needed to determine when litter application should be based on N and when it should be based on P.

Soil test P levels clearly influence soluble P concentrations in runoff water from agricultural fields. Thus, fundamental and applied research is needed regarding the critical level above which additional P should only be applied with limitations. Information is needed as to how this critical level will vary with soil types, slope, crops, and management.

Use of critical soil test P levels should be applied at a watershed level rather than at the farm level because P losses are rarely uniformly distributed within a watershed (that is, critical P-contributing areas exist due to land use and natural processes). In addition, the watershed is the logical unit for correlating land use with the impacted water body. To aid in developing a cause-and-effect relationship, runoff models need to be refined to better account for P losses from various land-use scenarios.

The traditional methods of analysis for P in the soil should be reviewed in light of the move to sustainable agriculture and conservation tillage. Under these systems and where land application of manure is practiced, the pool of soil P appears to be changing (Pierzynski et al. 1990, Sharpley et al. 1991a), and this may not be reflected by the traditional soil test. In some cases, soil test results may unnecessarily suggest the addition of P without a possibility of P response due to crop needs being met by mineralization of organic P.

From a water quality standpoint, methods for analyzing runoff are needed that determine the amount of algal-available P in soluble and adsorbed form. Methods such as those outlined by Sharpley et al. (1991b) that identified bioavailable P (BAP) should undergo wider testing by researchers and appropriate agencies. Additionally, some method of relating soil test P to water quality is required. Investigations that examine the relationship between quick tests for soil, labile, and algal-available P should be encouraged (similar to those investigations by Wolf et al. 1985).

Future research should be directed towards understanding the dynamics of different P fractions (soluble, particulate, and especially bioavailable P) transported in runoff and their dynamics in lakes. This research should focus on the mechanism of exchange between sediment and solution P. With the accumulation of fertilizer and residual P at the soil surface, the relative importance of the present partitioning processes may need to be reevaluated. In particular, more accurate simulations of residual soil P release are needed. With the move to low-input agriculture, these improvements will enable evaluation of P transport in runoff from soils with high residual P levels in the absence of additional P inputs.

Although many models are available, it is often difficult to select the most appropriate model to obtain the level of detail of information required. Once the appropriate model is chosen, a major limitation is often the lack of input data to drive the model. This most frequently limits model use, and output will only be as reliable as data input. Because of these limitations, more research should be directed to development of a soil index to identify soil and management practices that may enrich the bioavailable P content of surface waters.

Land management programs that minimize P losses in runoff are needed. While models can provide some direction, the resource manager needs a practical method for handling P such that loss is minimized. Such a program should involve the amount of P in the soil and manure, soil chemical and physical properties, slope, management, time of year, etc. Efforts similar to those by the Phosphorus Index Core Team (PICT), sponsored by USDA–NRCS, should be encouraged.

More applied work is needed on evaluating water quality impacts of existing poultry manure management practices. Additionally, efforts toward developing innovative new practices should be encouraged. For example, Edwards et al. (1992) examined the best time of year to apply broiler litter from a water quality standpoint. Certain times of the year were clearly better than others. Future research for determining the ideal timing of manure applications should balance the timing of nutrient requirements by the crops with the cleanout schedule of the animal rearing facilities.

More research is needed on P precipitation in manure utilizing Al, Ca, and Fe compounds, such as aluminum sulfate. Research is needed to determine which chemical amendment will transform phosphate in poultry manure to an insoluble mineral that is stable for geological time periods. If P runoff can be controlled in this fashion, then application rates of poultry manure could be based upon N loading. The compounds used for P precipitation should also inhibit ammonia volatilization, hence conserving N and decreasing the threat of acid rain.

Runoff studies focusing on the movement of microorganisms from land-applied poultry litter into adjacent water bodies have not been reported in the literature. High counts of indicator organisms, such as those found in the streams and rivers of Arkansas, indicate the possibility of a potential health hazard that so far has received very little attention. Research needs to be conducted on the types and amounts of organisms reaching water bodies from land application of poultry manures and on developing BMP's to deter such movement. The use of filter strips, composting, or chemical litter treatments, such as alum or slaked-lime applications, should help reduce the number of viable organisms entering the aquatic system. More research also needs to be conducted on decreasing ammonia volatilization from poultry litter, both within and outside of chicken houses. Nutrient management studies should also be conducted to determine BMP's that minimize groundwater contamination from nitrate in poultry litter.

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Chapter 4

Swine Manure Management

J.L. Hatfield, M.C. Brumm, and S.W. Melvin

Production of pork is a major agricultural enterprise in the United States, and a majority of the production occurs in the Midwest (Ohio to Nebraska and Minnesota to Missouri) and North Carolina. Seventy-nine percent of the hogs and pigs marketed in 1987 (96.6 million head) were produced in the north-central region of the United States (Bureau of the Census 1989). Iowa has ranked first in hog inventory since 1980, and in 1987 was estimated to have 25.6 percent of the December 1 inventory of 53,795,000 hogs and pigs on farms. Inventory on farms tends to fluctuate between 50 and 70 million swine in a 4- to 7-yr cycle.

In 1980, 21 percent of the growing-finishing pigs (pigs raised from 12 to 100 kg for meat production) and 45 to 50 percent of the nursing and nursery pigs in the United States were housed in confinement facilities (that is, liquid-manure systems; VanArsdall and Nelson 1984). With the large influx of new confinement construction, especially the construction associated with contract production units in North Carolina and Iowa, it is logical to predict a major increase in the percentage of manure captured and stored as a liquid or semiliquid.

The Corn Belt States are expected to remain the primary hog production area, although some shifts within the area will occur. Because of historically lower feed grain prices and lower human population densities, pork production is expected to expand west of the Mississippi River, especially in the western (Kansas, Colorado, and Wyoming) and southwestern fringe (Oklahoma) areas of the Corn Belt (Hurt et al. 1992).

Expansion will be governed in part by individual state laws or constitutional amendments regarding corporate ownership of livestock. These laws can vary tremendously from one state to the next. Currently, Nebraska has an amendment to its constitution restricting nonfarm corporate ownership of livestock destined for slaughter while Wyoming is using municipal bonds as a source of financing to attract corporate production units. Similar differences exist among other states.

One of the primary issues associated with the production of pork and expansion of production is the disposal of the animal manure and the odor associated with animals and manure storage facilities. Traditionally, swine manure has been returned to the land in some manner in the production areas; however, land application has come under attack in rural areas because of the odor problems during application. Estimates are that swine manure production accounts for 12 to 15 percent of the total livestock waste produced annually in the United States (VanDyne and Gilbertson 1978).

Today's swine production systems have become larger, more specialized, and more dependent on purchased feed supplies than in the past. Environmental problems associated with swine production during the 1950's and 1960's were often overlooked. However, swine production was characterized by small, individual systems that relied on recycling of animal manures back to the land as a major nutrient source for the farm. In the last 20 yr, many structural changes have occurred in the industry. These changes have caused concern over the environmental effects of swine manure management. The industry is rapidly consolidating. A recent University of Missouri study (Rhodes 1990) indicates that larger production systems are growing the fastest in terms of percent market share. This study shows that only the farms with annual sales of over 1,000 head are expanding. In 1988 large farms (those with more than 1,000 head) produced over 60 percent of the market hogs.

The environmental effects of swine manure storage systems and application methods are a concern, particularly with respect to surface water and groundwater quality and to air quality as affected by odors and gaseous emissions from large-scale swine production operations.

Manure Production and Composition

Swine manure composition may be estimated from various sources (Midwest Plan Service 1985, American Society of Agricultural Engineers (ASAE) 1990). Some estimates of swine manure's fertilizer components available to the plant are listed in table 15. In ASAE (1990), data are given for estimates of daily manure production for various species and means and standard deviations of physical and chemical characteristics of the manure. Swine are estimated to produce daily raw manure of as much as 8.4 percent of body

Table 15. Mean (in parentheses) and range of values for composition of swine manure from various handling systems

Handling system	Dry matter (percent)	Nutrients available to the plant: Ammonium N	Total N	P	K
For solid manure		---------- g kg^{-1} ----------			
With bedding	15–20 (18)	2.7–4.0 (3.1)	4.0–4.9 (4.5)	1.4–2.6 (1.8)	2.2–3.7 (3.0)
Without bedding	17–20 (18)	2.2–3.6 (2.7)	3.1–4.5 (3.6)	1.0–2.0 (1.4)	2.2–3.3 (2.6)
For liquid manure		---------- g L^{-1} ----------			
Anaerobic storage	2–7 (4)	2.5–3.7 (3.1)	3.4–6.6 (4.3)	0.7–1.6 (1.4)	1.2–3.0 (2.2)
Lagoon	0.3–2.0 (1)	0.2–0.6 (0.5)	0.4–0.7 (0.5)	0.05–0.2(0.1)	0.2–0.6 (0.4)

Source: Adapted from Sutton et al. (1983).

weight (urine and feces). Sweeten (1992) estimated the total production and nutrient content of swine manure in the United States in December 1988 to be as follows:

Number of head	55,299,000
Annual manure production (solids)	14.1 million Mg
Annual N production	0.66 million Mg
Annual P production	0.42 million Mg
Annual K production	0.66 million Mg

More recent calculations indicate that the annual production of P and K should be about 0.23 and 0.37 million Mg, respectively.

Since much of the nation's swine manure can be collected, stored, and spread on the land surface, this manure could be used as a substantial nutrient source for crops. In fact, if all U.S. swine manure was recovered and applied without loss of nutrients, it could supply the nation's corn crop with one-eighth of its N needs and one-fourth of its P and K needs. It is estimated that over 80 percent of the manure is generated in systems where manure could be collected. Manure handling and storage systems may remove a significant amount of N, but P and K are not likely to be significantly affected by treatment.

As of yet, swine production units are not geared toward retaining nutrients in swine manure. One reason for this is that land for manure application is limited. Many units use anaerobic lagoons to digest manure solids and allow the manure to be handled as a liquid. Anaerobic lagoons may volatilize 70 to 90 percent of the N in the manure. Manure N is converted to ammonia in these lagoons and is lost to the atmosphere. By volatilizing this N, anaerobic lagoons allow land requirements to be decreased to 10 percent of the land required for application of slurry manure.

Swine manure tends to be a relatively homogeneous material from production unit to production unit, unlike manure collected from ruminant animals. The swine in the United States are fed diets similar to those fed to poultry. The swine diet is formulated with corn or grain sorghum and soybean meal, and vitamins and minerals are added to prevent deficiency. In addition to Ca and P additions, Zn is added at 50 to 100 ppm, Cu at 5 to 10 mg kg^{-1}, and Se at 0.3 mg kg^{-1} (National Research Council 1988). As a percentage of the total mineral content in the diet, excreted swine manure is estimated to contain 86 percent of the Cu, 100 percent of the Zn, 79 percent of the Mn, 40 percent of the Ca, 74 percent of the Mg, 59 percent of the K, and 66 percent of the Na offered to the pig (Overcash and Humenik 1976). The FDA held hearings on the environmental impact of selenium additions to all animal diets (Muirhead 1992), and there are current regulations on the additions of Se in animal feed.

The major differences in composition of the manure are dependent on the methods of collection, dilution, and storage and are not diet dependent. About 85 percent of the N in a typical corn and soybean diet is digested (McConnell et al. 1972). The majority of the

N excreted from the pig is as uric acid in the urine and organic N forms in the feces. Phosphorus is excreted as phytic acid (an organic matrix derived from the undigested P in cereal grains) and as other complexes that result from growth and digestion processes. Phosphorus is excreted in both the feces and urine. About 40 to 60 percent of the P in a corn and soybean diet is digested (National Research Council 1988).

It is estimated that corn and soybean diets supply sufficient K for swine of all sizes, and therefore supplemental K is not normally recommended (National Research Council 1988). However, K additions may improve swine growth rate (Mabudiuke et al. 1980, Coffey 1987). Little data exist on the digestibility and retention of increased K in the diet. Thus, no predictions can be made as to the impact of supplemental K additions (generally as KCl) on the composition of swine manure.

Swine nutritionists are evaluating the results of using phytase enzymes in diets to enhance the digestion and use of phytase P in cereal grains. In cereal grains the majority of the P is bound as phytate (National Research Council 1988). This form of P is not readily available to nonruminants because they lack phytase, which cleaves the orthophosphate groups from the phytate molecule (Cromwell et al. 1993). Nutritional P needs in pigs are met by adding inorganic P sources to diets because only 15 percent of the P in corn and 25 percent of the P in soybean are available (National Research Council 1988). The addition of the inorganic P sources leads to 65 to 75 percent of the P being excreted in the manure (Lei et al. 1993). Addition of dietary phytase to swine diets will lead to enhanced use of phytate P from the cereal grains and reduce the addition of inorganic P sources. Improved use of phytase P is generally associated with improved protein use, which will reduce the amount of excreted P. In the Netherlands, it is estimated that N and P excretion by pigs can be reduced by 33 percent and 40 percent respectively by the year 2000 through advances in swine nutrition (Jongbloed and Lenis 1992).

Sodium chloride additions to swine diets have decreased over the years, partially in response to concerns about the fate of Na in stored manure. Generally sodium chloride is added to swine diets at the rate of 0.25 to 0.5 percent to prevent Na deficiency symptoms, with 0.25 to 0.3 percent being the most common addition rate. In anaerobic storage pits, Na levels in manure range from 5,000 to 9,000 mg kg^{-1} on a dry matter basis for dietary additions of 0.2 to 0.5 percent (Sutton et al. 1976). On average for all phases of production, it is estimated that 40-kg pigs produce 182 g of volatile solids per day, and the ratio of volatile solids to total solids equals 0.81 (Overcash and Humenik 1976).

The United States industry is improving the overall conversion efficiency of feed to meat of the swine herd. Better use of nutrients leads to a lower rate of converting food to waste. Current estimates of manure production and composition are based on feed-conversion efficiencies of 3.7 to 3.8 kg of feed per kg of meat. However, many producers have made large advances in production efficiency and now report conversions of 3.3 kg of feed per kg of meat or better. Recent advances in reproductive efficiency also have led to less waste generated from sows and boars as a percent of the total waste stream. Thus, previous estimates of waste production and composition may prove to be inaccurate estimators and in many cases will overestimate both the total volume of production and the composition of the waste produced.

Manure Management Systems

A major change in the structure of the pork-producing industry is also impacting the animal waste issue. While total pork production remains relatively constant in the United States (about 92 to 93 million swine were slaughtered in 1992), the number of farms selling hogs or pigs has declined from 1,273,000 in 1959 to under 200,000 in 1990 (Rhodes 1990). By the turn of the century, the number of farms selling pigs is expected to decline to slightly more than 100,000 (Hurt et al. 1992). In 1988 about 69 percent of the commercial hog slaughter in the United States was from 28,700 operations.

While many swine farms have all their production at one site, an increasing number involve two or more sites, either through production contacts or expanded ownership. Thus, the issue of swine manure is becoming an issue of point source production, especially as it relates to livestock ownership and responsibility for the collected material.

Swine manure is handled as a solid, a semisolid slurry, or a liquid, depending on the type of housing and manure handling system used. Each of these systems has some unique features that add complexity to the problems of manure handling and use, and some of these features are discussed in the text that follows.

Systems for handling solid manure

No more than 15 percent of United States swine are raised on farms using systems designed to handle solid manure. These systems are most commonly used in the western Corn Belt. Smaller production systems may make use of extensive housing systems in which small, roofed buildings are used to handle solid manure. Other small production systems may involve the use of pastures or open feedlots for distributing and handling manure.

In pasture production, manure is generally spread "naturally" by the swine as they graze. Rotation grazing will allow manure to be somewhat uniformly distributed in the forage area except for in watering and feeding areas. In this system little manure is collected and spread on other land. Some overloading of manure in specific areas can be expected if feed and watering systems are not moved frequently, since these areas collect a majority of the manure excreted. Pasture production systems are most common in states where smaller swine farms are more common. Certain areas within states, such as Henry County, IL, have been producing swine on pastures for many years. Pasture production is most common in the mid and southern Corn Belt. However, it is estimated that no more than 5 percent of swine are now raised on pasture.

Open feedlot systems are also common with small- to moderate-sized production systems. These systems are not covered by a roof, and the feedlot surfaces commonly have an accumulated manure layer on them. Solid manure is scraped from the feedlot surface periodically. Scraping frequency may vary from once or twice weekly to once monthly. Some manure is lost from the feedlot surface through runoff from rainfall or snowmelt. Unless some runoff containment system is in place, surface water contamination is possible if the runoff from the feedlot can enter a body of water before manure solids are settled or infiltrated into soils during transport in the runoff.

Research has shown that 5 to 20 percent of the manure deposited on an open feedlot can be expected to be transported from the feedlot via water runoff. The fate of manure nutrients is affected by whether or not a solid settling system is built to contain solids. Runoff losses are highest for K and lowest for P, assuming solids are retained in a solids settling system below the feedlot. Solid storage systems are required to store manure between land disposal events. These storage facilities generally consist of an on-grade concrete pad with low walls surrounding the pad to allow manure to be pushed into storage and removed with a blade or a front-end loader. The overall nutrient value of manure from solid systems is quite variable, and N losses during storage of 20 to 40 percent have been reported for these manure systems. Typical concentrations of N on a dry-weight basis for solid manure applied to land range from 0.45 to 0.55 percent for manure containing no bedding and from 0.25 to 0.50 percent for manure containing bedding.

Other solid systems besides open feedlot systems may also use bedding. The most common bedding material is straw, but wood chips or shredded newspaper are sometimes used. These systems may have totally or partially roofed pens in which bedding is added to absorb urine and to provide insulation for the animals inside unheated buildings. Manure and bedding is periodically removed from the pens, and the pens are rebedded to keep animals clean and comfortable. Once removed, the mixture of manure and bedding can be stored on concrete pads with optional low outside walls to help contain the mixture. Stored manure can be stacked in a pile with a front-end loader or stacking elevator.

Solid manure can be applied to the field using regular box spreaders or side-discharge flail-type spreaders. Some box spreaders require an end gate to prevent leakage of the material from the rear of the spreader during transport.

Systems for handling slurries

Most large-scale swine production systems have totally roofed confinement systems. Bedding is purposely not used so that the manure can be handled as a slurry or as a liquid. Manure converted to a slurry is not diluted much, since little water is added in the conversion process. Liquid manure, however, has been diluted quite a bit since significant water is added to assist with manure transport, treatment, and land application. Slurry manure systems are most common in the north-central region, where manure can be recycled back to cropland and where cool temperatures are not as conducive for lagooning swine manure.

Slurry systems commonly use several types of storage structures. The most common system is the below-floor pit covered with a slatted floor. Until recently, a high proportion of all swine confinement systems using slurry manure used a deep-pit storage system. However, in recent years, there has been more concern over air quality problems in buildings resulting from long-term manure storage in the building. Alternatives to an indoor storage system are in-ground storage tanks remote from the building, aboveground storage tanks, and earthen structures. In-ground tanks may be covered or uncovered, but if left uncovered, they must be isolated with a safety fencing to prevent accidents. Uncovered tanks unfortunately can collect significant snow during winter. Round tanks are becoming more popular as remote tanks, since the shape has structural advantages and these tanks are more easily agitated. Aboveground tanks can be constructed from various materials, but concrete and glass-fused steel are the most popular. Earthen structures provide the lowest cost storage system, but adequate soil investigation and construction controls are necessary to minimize groundwater pollution hazards.

Equipment for handling slurry manure is designed for agitation, pumping, transport, and spreading. Vacuum loading tankers are designed for several functions. However, some systems require more agitation and pumping capacity than available with vacuum loading equipment. Agitation equipment usually consists of a propeller or open impeller. Pumps must be able to chop and pass large-diameter solids in the manure. Tank-type manure spreaders can be mounted on trailers or trucks for field distribution of manure. Direct-injection equipment for immediate incorporation of manure is now common. Direct-injection allows immediate covering of the manure to prevent N loss by volatilization, reduces the potential of surface runoff of N and P, and significantly reduces odor potential.

Manure is most often applied to cropland near the swine production unit. A majority of slurry storage systems can store manure for 120 to 180 days, meaning that manure applications to fields are needed two or three times per year. This sometimes leads to problems with having land available for manure application in the middle of the growing season or during winter. It is estimated that 50 to 60 percent of producers use slurry manure handling systems. These units are most common in the Corn Belt. A slurry manure should be stored for at least 180 days before application in the Corn Belt. The longer storage period minimizes manure application problems, and full-year storage is becoming more popular.

Systems for handling liquid manures

Hydraulic flushing systems have been successfully used for 20 yr for quick, efficient removal of manure from swine confinement buildings. Flushing systems require the use of larger manure storage systems, since significant amounts of dilution water are added to the manure during flushing. Anaerobic lagoons are used extensively for storage and treatment and, in many cases, as a recycling system. Recycled treated lagoon water is often used to minimize storage requirements. In areas where lagoon water can be used for irrigation throughout the year and where adequate fresh water is available for flushing, recycling is not generally practiced. Anaerobic lagoons are also popular for swine production systems in areas with a limited land base, since high losses of N can be expected in anaerobic lagooning systems.

Anaerobic lagoons convert manure to a liquid that is low in solids, allowing easier transport and application. Conventional irrigation equipment can be used to apply anaerobic lagoon liquid to land. Even though higher volumes of waste are produced with these systems, the cost and labor requirements for application of liquid manure are lower than for slurries or solids.

Aerated lagoons can also be used as a storage and treatment system for flushing units. Odors are minimized and recycled water is safer in terms of disease prevention, but the cost of mechanically aerating a swine lagoon is relatively high. Capital requirements and energy and maintenance costs have been high enough to prevent this use of aerated lagoons from becoming common on swine farms, even though aerated lagoons are commonly used for this purpose in municipal and industrial sewage treatment systems.

Odors, potential leakage, overflow, and over application of lagoon effluent are the major environmental concerns associated with anaerobic lagoons. Proper design, loading, and management are required to minimize odor problems. Soil investigations and proper construction techniques are required for groundwater protection. Adequate irrigation equipment is needed to dewater lagoons on a regular basis and to distribute the water over farmland. Nutrient management plans should specify loading rates to properly use the manure product.

Anaerobic lagooning systems do lose significant amounts of N to the atmosphere, but P and K are not lost. It is estimated that 80 to 90 percent of the input N is lost to the atmosphere through ammonia volatilization. A high proportion of the P is contained in sludge from the lagoon. Periodic cleanout of this sludge is required for continued efficient operation of the lagoon. These P-rich sludges should be applied to land with caution so that high levels of P buildup will be prevented.

The majority of the systems using anaerobic and aerobic lagoons are used in warm climates. The majority of large operations (1,000 head per year) are using anaerobic lagoon systems to minimize land application areas. These operations are concentrated in the Southeast, the southern Corn Belt, and the southwest Plains. It is estimated that 20 to 30 percent of the manure from swine production is processed in liquid manure systems.

Manure Handling and Disposal

The major concerns associated with managing manure from swine are related to runoff control from open feedlots, storage requirements, and land application of manure collected from confinement facilities. The runoff control issues for handling this type of manure are very similar to those for handling other types of manure.

Storage and land application problems from confined production units occur due to the large volumes of water often associated with the material. Depending on method of collection and storage, the collected material can contain from 90 to 99.9 percent water at the time it goes into storage.

Generally, growing-finishing pigs weighing 21 to 100 kg can be expected to generate 0.39 to 0.45 kg of waste per day on a dry matter basis (Brumm et al. 1980). This manure contains 1.9 percent P, 7.2 percent N, and 3.2 percent K as byproducts of digestion. Depending on the phase of production and the specific production practices of the pork producing unit, the manure may also contain significant amounts of Cu, which is added to the diet as copper sulfate at up to 250 mg Cu kg^{-1} to promote growth. Swine manure may also contain antimicrobial drug residues, which are added to the diet to enhance growth and improve health (Brumm 1978).

Copper levels and drug residues in manure have in some instances limited the uses of the manure. For instance, manure high in Cu is undesirable for anaerobic storage because it reduces biological activity (Brumm 1978), and manure high in antimicrobial drug residues is undesirable for pilot anaerobic digesters designed to generate methane because of the limited biological activity in the methane generator (Fischer et al. 1978).

Refeeding of collected swine manure to swine has been researched and has been tried on several commercial swine units. However, the large volumes of water associated with typical manure collection has meant that dewatering of some type must be used to generate a material that is easily handled. Refeeding swine manure to a different species (generally beef cattle) has been successful on a limited scale. The possibilities of high concentrations of Cu or other potentially toxic elements or drug residues has limited refeeding to beef animals during their growing stage. This limitation has generally minimized the concern of residues entering the human food chain as a result of refeeding. In general, the primary safety concerns associated with using animal manure as animal feed involve potential harmful residues of pesticides, drugs, toxic minerals, and other toxins, and the hazard of disease transmission (American Society of Animal Science 1978).

Some sow herds are fed dewatered swine manure as a means of enhancing colostral immunity for newborn pigs. Refed manure, however, has been recognized as a possible source of internal parasite reinfestation and dysentery spread. Reuse of stored swine manure as either a source of water for flushing or as a nutrient source in the diet has caused concern regarding animal health. Anaerobic storage in either deep pits or lagoons does not affect the survival of roundworm eggs, *Treponema hvodvsenteriae,* and *Salmonella* spp.

Swine manure that contains small or no amounts of antimicrobial drug residues can be used to generate methane. In general, successful methane generation relies on thermophilic bacteria for the conversion of organic wastes to volatile fatty acids and then to methane. With much of the pork production in the United States occurring in the north-central regions, extensive investments in insulated and even heated facilities have been necessary for this bacterial process to be possible during winter weather.

During methane production only 40 to 60 percent of the volatile solids are converted to methane; 1 m^3 of methane is produced per 0.61 kg of volatile solids converted in the digestion process (Sweeten et al. 1981). The conversion of the organic wastes to methane does not decrease the need for disposal of byproducts. Removal of carbon as methane from the waste stream does not decrease the amount of N, P, K, or other significant elements in the digester effluent.

Land Application of Manure

Problems

Decisions on the best ways to apply swine manure to land are complicated by compromises between achieving best soil erosion control and best conservation of nutrients in manure and use of these nutrients by growing crops. For a long time, extension specialists have recommended that manure applied to the soil be incorporated into the soil surface within 24 hr after land application. This practice can significantly reduce odors and can minimize ammonia volatilization so that N in manure is conserved. Fall application of manure to cropland following harvest is often recommended because the risk of damage from soil compaction is minimized. Spring applications are usually accomplished prior to tillage and planting. Frequently in the spring, soils receiving the manure are close to saturation, resulting in significant compaction from spring application. Also, labor availability often favors fall application.

Many producers, as part of their approved conservation plan with the Natural Resource Conservation Service (formerly the Soil Conservation Service), have agreed to significantly restrict or stop any fall tillage practices as a means of maximizing residue cover at the time of spring planting. Thus, land application and tillage in the fall, long an accepted best management practice for manure application, may result in a violation of the NRCS plan and a reduction in government payments under provisions of the 1985 Food Security Act and the 1990 Food Agriculture Conservation and Trade Act. Direct injection by means of tanker wagons equipped with injection devices has become a common technique for land application in much of the Midwest. However, the injection process disturbs the soil as much as major tillage operations.

Limiting fall applications of manure in the future is likely to have an impact on the cost of storing manure. Currently, most regulations require 90 to 180 day storage capacity for collected animal manure. This requirement capacity generally is based on the growing season for the area or state, on the assumption that manure will be applied to land in the spring or fall. If fall application is no longer feasible due to soil conservation concerns or possible runoff concerns, the length of storage may need to be increased to 270 to 360 days. Longer storage not only increases cost but also increases the workload in the spring when many producers are already very busy.

Some farmers may be able to find use for swine manure in the summer. The traditional corn and soybean farmer who also raises pork may need to add some forage crop to provide a land base for summer applications. Forage crops, however, are generally not useful on swine farms, but producers may have to find a use for these crops in the future.

Impact of manure on the soil

Although manure can be used to supply adequate nutrients to grain crops, it is usually difficult to determine the exact nutrient content of the manure. Sutton (1992) stated that the potential fertilizer value of swine manure may range from $2.50 to $3.50 per market hog sold. He outlined some of the potential problems that need to be addressed concerning the use of swine manure as a fertilizer. Currently, there is no rapid, inexpensive method for testing manure before it is applied to land. Without knowledge of the nutrient content prior to application, it is difficult to apply proper rates to meet the soil fertility requirements. Even if proper rates could be determined, application methods can cause poor or inconsistent distribution of manure in a field.

Nitrogen is often thought to be the primary nutrient available in manure. However, there are proportionally larger amounts of P and K available than N because of the losses of N during storage. Little information exists regarding N losses during storage, but studies show that these losses range from 10 to 90 percent. Application of manure in quantities required to meet the N requirements of corn or other grain crops can lead to excessive supplies of both P and K in the soil. The manure value of a fertilizer must be based on the nutrient supply from all elements and therefore must be based on effective testing of the composition of the swine manure. Midwest Plan Service (1985) provided estimates for N losses during storage and land application, and some of these estimates are shown in tables 16 and 17.

Table 16. Nitrogen loss during storage of manure from different manure handling systems

System	Type of manure	Nitrogen lost (percent)
Daily scrape and haul	Solid	15–35
Bedded manure pack	Solid	20–40
Anaerobic pit	Liquid	15–30
Aboveground storage	Liquid	10–30
Earth storage	Liquid	20–40
Lagoon	Liquid	70–80

Source: Midwest Plan Service (1985).

Table 17. Estimates for the amount of nitrogen that is lost within 4 days of applying swine manure

Application method	Type of manure	Nitrogen lost (percent)
Broadcast	Solid Liquid	15–30 10–25
Broadcast/immediate cultivation	Solid Liquid	1–5 1–5
Direct injection	Liquid	0–2
Sprinkler irrigation	Liquid	15–35

Source: Midwest Plan Service (1985).

There are various techniques for estimating the amount of available nutrients from manure applied during the year or during a previous year. This estimate will vary greatly with the type and form of manure applied to land. With solid manure or open-lot manure, little N is found in the ammonia or ammonium form, since much of the N in this form has been volatilized prior to land application. Slurry manures may have at least 50 percent of the total N in ammonium N form, which is readily available following application.

The key to estimating N availability after application rests on the mineralization decay rate expected from the breakdown of organic solids after land application. For swine manure, the estimated first-year contribution from organic N in manure can vary from 25 to 50 percent of the total organic N. This is a function of the breakdown rate of solids in the soil, which is a function of particle size, shape, temperature, moisture, and other environmental factors, including the level of antibiotics in the manure. However, with modern feeding systems, manure particle size is small, and mineralization rates approaching 50 percent of the organic N in the first year may be expected. The second-year rate is usually about half of the first-year rate (or approximately 25 percent), and the third-year rate is about half the second-year rate (12.5 percent). Little additional N contribution is expected from swine manure 3 yr after application.

In addition to providing nutrients for the crop, swine manure and other manures are an energy source for microbial activity that promotes soil structure and aggregation and therefore soil quality; however, the mechanism of the process and the ideal application rates have not been identified. The lower N contents reported in lagoon systems may be adequate for maintaining and promoting microbial action in the soil. Soil aggregation and a stable soil structure would improve the infiltration process, decrease surface runoff, and enhance the effect of other soil conservation practices. Manure application just for the purpose of enhancing the soil, however, may not provide sufficient economic incentive for the use of manure.

Management processes for preventing N loss from manure applied to soil are not well understood. Losses of N (as ammonia, nitrous oxide, or methane from the soil) diminish the value of the manure and also may add to the abundance of greenhouse gases in the lower atmosphere. Preliminary studies indicate that the amount of N lost after application is substantial.

Application rates to the soil are dependent upon the soil, crop, climate, manure composition, and mineralization rate. Proper manure application methods and rates should be incorporated into best management practices designed to manage both crop residue levels and soil quality. The amount of land area needed for the effective use of manure will depend upon the composition of the manure and the treatment of the manure after application. Nitrification inhibitors are sometimes used to arrest the rate of mineralization; however, they have not been fully evaluated under field conditions and the results have been variable among different studies. More research is needed to fully understand the most effective manure application methods and rates for different crops and soil conditions.

Disposal of dead animal carcasses

Along with the usual manure problems associated with pork production, a new concern is the disposal of dead animal carcasses. In many states, the legal requirements for disposal call for incineration, burial, or pickup by a commercial rendering service. However, the number of dead-animal rendering services in the United States has declined significantly (Fats and Proteins Research Foundation 1992).

Because of decreased access to rendering services, increased charges for rendering services, frozen ground in winter months, and high fuel costs associated with incineration, many pork producers are evaluating composting of swine carcasses as a disposal alternative. Research information in support of this practice is limited, however, and it is unclear what the legal aspects of this practice are with regard to current state laws and local health regulations.

Effects of Manure on Environmental Quality

Water quality issues

Environmental water quality problems resulting from swine manure use on land have been related to excess manure generation relative to land available for application and to inadequate manure storage and handling facilities. Excess manure application rates, runoff and leachate from manure application sites, and leakage and overflow from manure storage sites represent major environmental concerns. The problem of a manure surplus on swine production farms is exacerbated by low-cost commercial fertilizers, concentration of large (greater than 10,000 head) production units, reduced availability of labor, narrow profit margins, and higher priced land. Low-cost commercial fertilizers contain the essential nutrients (N, P, K) in a uniform mix and in a form more easily transported and applied to the field. Manure application requires many trips to the same field because of the large volume of water in manure. Concentration of large production units creates a surplus of manure relative to the land area for application without incurring large transportation costs. Often these facilities are located in areas with little land base for application, and other disposal methods must be used. Manure application requires labor, and on-farm labor sources are becoming less available. A smaller labor force coupled with the problem of application during the part of the growing season that is already busy leads to problems of effectively using manure. Crop production is on a narrow profit margin, and grain farmers want to decrease the potential risk by applying a nutrient source that will ensure adequate nutrient supply. The narrow profit margin coupled with the high price of land for either purchase or rent creates a situation in which manure is not an attractive nutrient source.

Population equivalent concepts are sometimes used to evaluate the potential for animal production systems to create water pollution problems. However, it is incorrect to assume that a large amount of manure generated by animals is an indicator of actual water pollution, since manure handling systems should be designed to prevent discharge of manure into water bodies. Manure generation is only an indicator of the total *potential* pollution. Modern manure management systems can and should be designed and operated to meet strict discharge guidelines.

Swine manure has several components that can pollute water. These include oxygen-demanding materials (organic matter), plant nutrients, and infectious agents. Color and odor are potential pollutants of secondary importance. Organic matter serves as an energy source for aerobic bacteria in a receiving stream. Increased bacterial metabolism resulting from a discharge of organic waste into a stream increases the oxygen depletion rate of the stream. If the rate of oxygen depletion exceeds the aeration rate of the stream, oxygen depletion occurs. Decreased or depleted oxygen levels can result in fish kills and anaerobic conditions in the stream or other water body.

Organic matter in wastewater has historically been measured as biochemical oxygen demand (BOD). This is a measure of the amount of oxygen required to metabolize waste during a specified time, usually 5 days. BOD is a measure of the organic “strength” of a manure; strength is measured by the oxygen demand during the 5 days.

Another measure of organic strength of a waste is chemical oxygen demand (COD), which is based on chemical rather than biological oxidation. COD will exceed the BOD value for animal wastes, since animal manure and other waste products contain organic materials resistant to aerobic bacterial degradation. COD/BOD ratios vary from 3.5 to 6.5 depending on species and feed rations.

Reduced organic substances such as ammoniacal N also increase oxygen demand. Relatively high ammonia concentrations are found in liquid manures, anaerobic lagoon effluent, and open feedlot runoff. Estimates of organic strength of different animal waste flows are available in many references (for example, Miner et al. 1966, Mielke and Mazurak 1976, Khaleel et al. 1978, and American Society of Agricultural Engineers 1990).

Figure 16 illustrates the relative strength of various types of waste. Note that raw manures have very high organic strengths compared to other common wastes. However, it should be noted that with the exceptions of accidental discharge or excessive precipitation, little, if any, waste should reach streams or other water bodies from animal production units that are environmentally safe.

Swine manures have high concentrations of plant nutrients. These nutrients are beneficial when properly recycled to land. These same nutrients, however, can pollute water bodies if manure is discharged into the water bodies. Nitrogen and P are the plant nutrients of primary concern. If they enter streams, these nutrients can stimulate the growth of aquatic plants, and these plants may have significant impacts on the acceptable water quality of that stream. In addition, high manure loading rates provide high levels of nitrogen, which can, in turn, increase nitrate concentrations of shallow groundwater.

Another potential water pollution hazard resulting from animal production is disease transmission of water-borne organisms. Several diseases can be transmitted in water from animal to animal and from animal to man (Hensler et al. 1970, Young 1974). Some examples include bacterial infections of *Salmonella, Listeria, Leptospiea, Vibrio, Brucella, Coxiella,* and *Chlamydia*. Other infectious agents such as *Mycoplasma*, fungi, and protozoa (*Cryptosporidium*) can also be transmitted in water. Managers of modern manure management systems must take into account the possibility of disease transmission through the environment and must therefore try to prevent improperly treated manure-laden runoff from reaching water bodies.

If swine manure is not handled and applied properly, it can be a threat to surface water and groundwater quality. Waste loading of swine manure discharges to groundwater or surface water is not well documented.

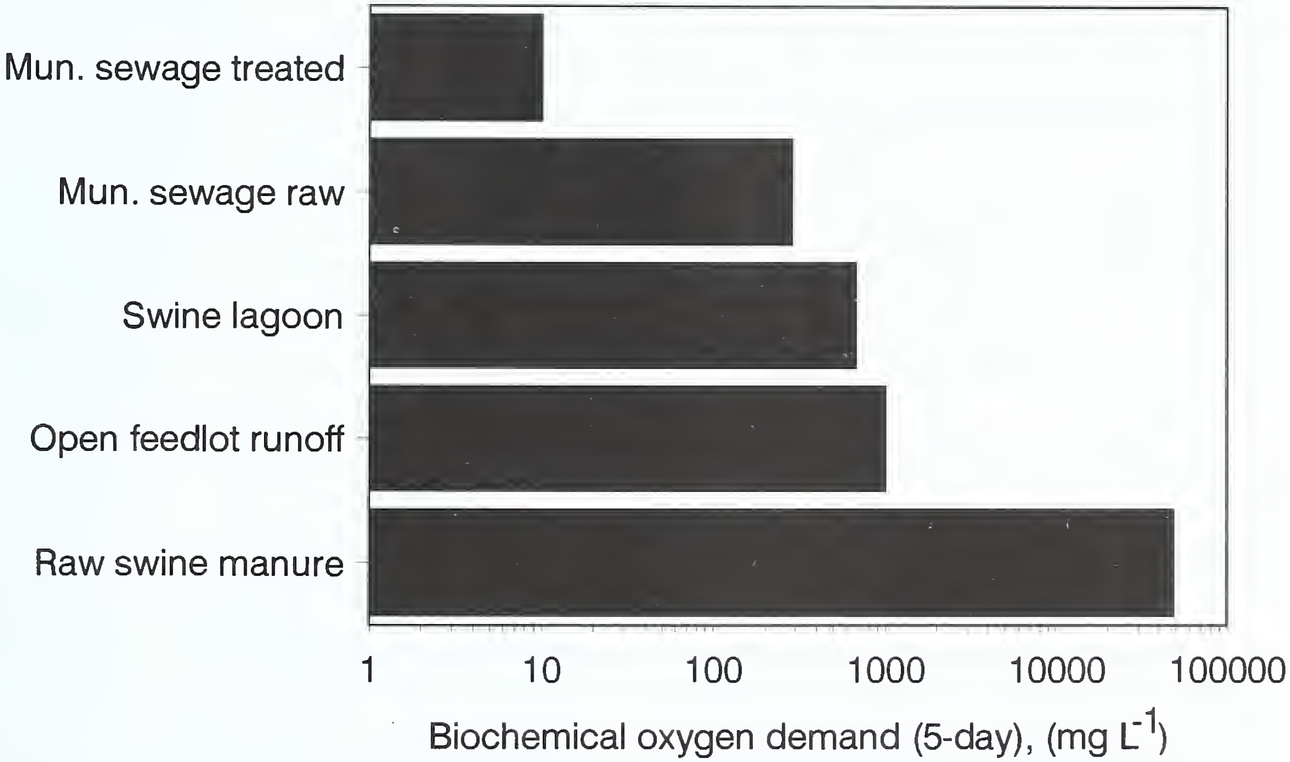

Figure 16. Biochemical oxygen demand of various wastes during a 5-day period

However, research indicates that little manure runs off land when the manure is applied properly. The worst-case scenarios for the incidence of surface runoff would be (1) rainfall occurring after manure is applied to frozen, snow-covered ground or (2) application of liquid manure by irrigation rates that exceed the infiltration rate of soil.

Environmental impacts of manure sites and application problems are just beginning to surface. Excess application rates in fields can lead to increased nitrate concentrations in shallow wells. In many areas, nitrate concentrations in wells often exceed 10 mg L^{-1}. The extent of these problems is not well known, and generally less than 20 percent of rural wells are expected to have nitrate problems; however, the soil above the well and the depth, aquifer material, and position of the well relative to any source will impact the nitrate concentrations.

Another water pollutant commonly associated with outdoor and unconfined animal production is increased sediment in surface water. Animal traffic in pastures, near and along streambanks, and on open feedlots can result in increased erosion in areas with animal production systems. Sediment is normally associated with cropland erosion, but in watersheds with significant permanent surface cover and high water quality, there is a potential impact for sediments from animal production systems to be a problem. Proper design and operation of feedlot runoff control systems and good pasture management can significantly reduce the problem.

Air quality issues

Odor control has become a major environmental concern of the swine industry. Swine producers have identified odor complaints as a major industry environmental issue. Because swine farms are larger and more concentrated, they have a larger potential odor problem. Neighboring residents have apparently become less tolerant of swine odors, since the frequency of lawsuits appears to be increasing. Fewer swine farms are in operation now than in the past, so now neighbors of swine farms are less likely to be associated with the swine industry.

Emission of gaseous wastes from production and manure storage systems has become a major environmental issue in Northern Europe during the past decade. Ammonia discharge from swine production systems is now being regulated in the Netherlands. Ammonia has been associated with acid rain problems in the region. Even though this has not yet been identified as a problem in the United States, there could be some future implications for the U.S. swine industry. Other gases, such as dinitrogen oxide (N_2O), methane (CH_4), and carbon dioxide (CO_2) are all associated with greenhouse effects. Production of these gases is increased through anaerobic treatment of swine manure.

Future Outlook on Swine Production and Research Needs

A study by the University of Missouri (Rhodes 1990) reports that over 50 percent of the nation's hogs and pigs since the mid 1980's have been marketed by farms producing more than 1,000 head per year. Nearly 70 percent of all market hogs in 1988 came from units producing greater than 1,000 head per year. In that year, 1,180 operations producing more than 10,000 pigs per year marketed nearly 19 percent of the nation's commercial slaughter of domestic origin, while a subgroup of larger firms marketing more than 50,000 head produced nearly 6.6 percent of the total. A survey of swine operations in early 1989 found that 30 percent of all operators were planning to expand their farms (Rhodes 1990). Plans to expand were more common among large farms, farms with multiple production units, farms outside the north-central region, and farms having new facilities. Therefore, the structure of the swine production industry continues to change. New, larger farms are expanding outside traditional production regions. Projections are that farms with less than 2,000 head may not be economically viable in the near future (Rhodes 1990).

If the trend of increasing farm size continues, problems with animal rights and animal welfare may arise. In most European countries, an increasing percentage of the breeding herd is being given access to straw bedding during a portion of the gestation and lactation phases of production in response to concerns for the welfare of the swine. If the United States follows the European lead at some future time, either through legislative or consumer pressure, an entirely new set of problems will be created since little information is available regarding the composition, storage, or land application of the resulting high-residue manure material (that is, the manure mixed with straw).

Although another general trend in United States swine production is towards increased confinement production, there is a growing minority of small and not-so-small producers who are intensively producing pigs outdoors. Outdoor pig production allows them to escape some of the high investment costs associated with confinement production units. In addition to the obvious concern regarding surface runoff from these outdoor production farms, there is the issue of nutrient leaching from the intensive production area, especially if stocking rates result in total removal of all vegetation. The United Kingdom now considers some outdoor farms as "nitrate-vulnerable zones" (Worthington and Danks 1992).

Future research on swine manure management must focus on several issues. Manure quality must be enhanced or at least preserved during storage and handling. As discussed previously, quality is affected more by treatment than by diet. Methods that can provide a rapid evaluation of the quality of the manure must be developed. An accurate evaluation of the nutrient content of a manure will be useful in avoiding potential negative environmental impacts to either water or air quality.

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Chapter 5

Management of Dairy Cattle Manure

R.K. Hubbard and R.R. Lowrance

Methods of collection, storage, and use of dairy cattle manure have undergone increased scrutiny during the last 15 to 20 yr. This is in response to local increases in manure quantities (from increases in herd size) and to heightened environmental awareness concerning adverse effects of manure on the quality of surface water and groundwater. Dairy cattle manure contains significant amounts of the primary nutrients (N, P, and K) as well as other essential plant nutrients and hence is an excellent nutrient source for crop growth. However, if excess amounts of manure are applied beyond the use capacity of the crops and soil or if manure is improperly applied, losses by surface runoff and leaching can contribute to eutrophication of surface water bodies or contamination of groundwater.

The primary issue with dairy cattle manure, both now and for the future, is development of management systems that use the resource without adverse environmental impacts. In a number of regions, the amount of dairy cattle manure produced exceeds loading capacity of soils available for manure application. Regulations have been passed in a number of states to protect surface water and groundwater quality from the impact of manure, but similar emphasis has not been placed on cropping systems to make efficient use of the material. More scientific research is needed to gain better information on cropping systems, manure application rates, and fermentation systems for producing methane gas so that manure is used wisely. Additional extension materials and other means are needed to inform cattle producers of best procedures for handling and using manure.

Methods of collection, storage, and disposal of dairy cattle manure have received increased scrutiny during the last two decades (Morgan and Keller 1987). The total number of milk cows and heifers calved in the United States has decreased from 11.2 million in 1975 to 10.2 million in 1991, while average milking-herd size increased from 75.7 to 103.8 cows/herd during these years (U.S. Department of Agriculture 1990, 1993). In 1991, 66 percent of the total cows were in the top 10 dairy states and 50 percent were located in 5 states (Wisconsin, California, New York, Minnesota, and Pennsylvania) (U.S. Department of Agriculture 1993). Over the last 10 yr, the top-10 states have remained relatively constant in ranking, although Michigan, Ohio, and Iowa dropped one place with the move of Texas from ninth to sixth and Missouri was replaced by Washington as the state with the tenth most cows. Large dairies (those having in excess of 1,000 head of cattle) account for a larger percentage of the cows in the South, Southwest, and Far Western states than in the Northeast and Midwest (Newton, personal communication 1995).

As dairy farm size has increased, so has the quantity of dairy cattle manure handled per dairy farm (Morgan and Keller 1987). The increased manure production plus heightened environmental awareness of associated soil and water quality problems has exacerbated the need for management systems that can use the biomass and nutrients in the manure without creating unacceptable air, soil, or water pollution. Problems with excess quantities of manure, however, are not limited to large dairy farms. States such as Wisconsin and Pennsylvania have many smaller dairies, but many of these small dairies produce more manure than they can handle. Regardless of dairy size, when the land use area is insufficient to handle the quantity of manure, problems with manure disposal occur.

Modern dairy management includes a proper balance of feed components so that milk is produced as economically as possible while the health and vitality of the animals are maintained. Nutrients are supplied through feed derived from pasture, hay, silage, and grains. Pasturing is done on either legumes or grasses, with grazing being the oldest and most common method. “Green chopping” or hauling the pasture to the cows is sometimes practiced. Forages used for green chopping may include any crop normally used for pasture or to make hay or silage.

Hay fed to dairy cattle may be made from legumes, grasses, or mixtures of the two. Many dairy farmers consider legume hay to be essential because it provides large amounts of high-quality proteins and calcium along with liberal quantities of vitamins A and D (Coletti 1963). Alfalfa is the most popular legume hay, while red, alsike, and crimson clover are also excellent sources of roughage for dairy cows. Hay made from grasses is generally inferior to legume hay (Coletti 1963).

Silages are used extensively as feed for dairy cattle. They provide succulent feed during the winter months when cows are restricted to dry roughage, make possible the utilization of the entire plant without much loss during bad weather, and can be used as a source of reserve feed during the summer months. The primary silage crop is corn, although acceptable silage can be made with sorghum. Silage also can be made from alfalfa, various clovers, soybeans, pasture mixtures, and oats or other small grains.

Dairy cattle often spend portions of their time in pasture areas, feeding and lounging barns, and milking parlors. From an environmental standpoint, manure dropped in any of these locations may be of concern. However, unless too many cattle are pastured per area of land or unless cattle are allowed free access to streams, lakes, or ponds, manure dropped in pasture areas may be of less environmental concern than that in barns and milking areas. Manure dropped by cattle while in the feeding and lounging barns and milking parlor is in effect a point source of nutrients that must be used. Point sources of pollution include such things as chemical spills, septic tanks, and so forth, and the manure dropped in barns and parlors is a point source in the sense that the land area where it is dropped does not have the capacity to filter the load. Water added from cleaning of tanks or utensils in the milkhouse also contributes to the total amount of manure load.

Dairy cattle manure is a complex material containing feces, urine, bedding, rain or other water, and milkhouse or washing wastes (Midwest Plan Service 1975a). This material contains all of the macronutrients needed for crop growth and has particularly high amounts of nitrogen (N), phosphorus (P), potassium (K), and calcium (Ca). In addition to its nutrient value, application of dairy cattle manure to cropland is known to improve soil organic matter and tilth (Klausner et al. 1974, Christensen et al. 1981).

Surface runoff from dairy feedlots and holding areas has high potential to cause water pollution. Also, mismanagement in the land application of dairy cattle manure has been documented as a cause of water pollution. Both P and N contained in the manure may contribute to eutrophication of surface water bodies. Dairies are often located in regions where land accessibility for manure application is restricted during large portions of the year due to cropping patterns and climate. Therefore, the land application rate of animal manure or liquid from lagoons or holding ponds may often be greater than normally recommended for meeting crop nutrient requirements. Odor from lagoons, holding ponds, or surface application of manure is also an environmental concern.

Proper management of dairy cattle manure involves integrating dairy herd size, available land, topography, climate, soil type, and financial resources to determine the best system. Alternative uses for manure besides land application include composting, refeeding, and production of methane gas via anaerobic fermentation. Overall, dairy operations throughout the United States ideally should use the nutrients and organic matter from the manure to reduce fertilizer and energy costs, while at the same time using treatment systems that do not have negative effects on air quality, surface water bodies, or groundwater quality.

Manure Production and Composition

Because dairy cattle normally spend a large portion of their time in the feeding and lounging barn, milking parlor, and pasture areas, they deposit a large portion of their manure in those areas (Westerman and Overcash 1980). Manure dropped in pasture areas may or may not be of environmental concern, depending on herd size, pasture area and location, and amount of time the animals spend in the area. The major source area for dairy cattle manure, which must be handled, stored, and treated or used, is the building complex containing the feeding barn, lounging barn, and milking parlor.

The daily manure production (feces and urine) per 454 kg of body weight for Holstein cows is approximately 34 kg, of which about 70 percent, or 24 kg, is solids, and 30 percent, or 10 kg, is liquid (North Carolina Agricultural Extension Service 1973). On this basis, the daily manure production of a mature Holstein cow weighing 636 kg is about 48 kg. The properties of dairy cattle manure depend on several factors, including the digestibility and protein and fiber contents of the feed, and the animal's age, environment, and productivity. Table 18 shows estimates of daily manure production and manure properties for a range of animal sizes. Other sources of information on the properties of raw or liquid dairy manure include Information Services, Agriculture Canada (1979), Ghaley et al. (1986), and Van Horn (1990).

When estimates of annual per animal dairy cattle manure production (from Van Dyne and Gilbertson

Table 18. Dairy cattle manure production and characteristics. Values are approximate. The actual characteristics of a manure can easily have values 20 percent or more above or below the table values. The volume of waste that a waste-handling system has to handle can be much larger than the table values due to the addition of water, bedding, and so forth.

Animal size (kg)	Total manure production (kg day^{-1})		Water (%)	Density* (kg m^{-3})	TS† (kg day^{-1})	VS‡ (kg day^{-1})	BOD_5§ (kg day^{-1})	Nutrient content (kg day^{-1})		
	(kg day^{-1})	(L day^{-1})						N	P	K
68	5.9	6.0	87.3	994	0.8	0.7	0.12	0.03	0.006	0.019
114	10.0	9.8	—	—	1.4	1.1	0.20	0.04	0.008	0.034
227	19.5	19.6	—	—	2.7	2.3	0.39	0.10	0.018	0.064
454	39.0	39.3	—	—	5.4	4.5	0.77	0.19	0.034	0.128
636	54.5	54.8	—	—	7.6	6.4	1.08	0.27	0.048	0.181

* Density = best estimate, not ASAE data.
† TS = total solids.
‡ VS = volatile solids.
§ BOD_5 = the oxygen used in the biochemical oxidation of organic matter in 5 days at 68 °F. A standard test to assess wastewater strength.

Source: Midwest Plan Service (1993).

1978) were multiplied by the number of milk cows and heifers that calved in 1991 (10.2 million), the estimated national annual dry weight of manure produced was calculated to be 2.2 x 10^7 Mg. The estimated nutrient content of this material was 7.1 x 10^5 Mg N, 1.2 x 10^5 Mg P, and 5.7 x 10^5 Mg K (Van Dyne and Gilbertson 1978). After losses during storage, transport, and application, the economically recoverable amounts of these nutrients can be estimated as 4.2 x 10^5 Mg N, 1.0 x 10^5 Mg P, and 4.9 x 10^5 Mg K (Van Dyne and Gilbertson 1978).

The actual composition of any particular batch of dairy cattle manure as removed from the milking parlor, feeding, or lounging areas depends on the amount of moisture, the amount of bedding material present, and the rations fed. Bedding incorporated into the manure increases the total solids content, while water added during washing dilutes the material.

Manure Management Systems

Handling of manure

Dairy cattle are housed in buildings and secured using either stanchions or free-stalls. Stanchions limit the forward and backward movement of cattle. Manure from stanchion barns is allowed to collect in gutters where it is manually or mechanically scraped and stacked in storage areas until it can be hauled to fields for spreading and use (Merkel 1981). Farmers with stanchion barns generally use bedding such as sawdust, straw, or wood shavings for the animals. Manual or mechanical scraping of the manure from the rear of the stall or the main alley into a collection gutter is generally done daily.

Characteristics of stacked stanchion barn manure depend upon the length of storage, environmental conditions, and the type and amount of bedding used. Average values for stacked and stored dairy cattle manure are 50 kg day^{-1} $animal^{-1}$ produced with 4,100 to 6,900 mg L^{-1} total N, 700 to 2,500 mg L^{-1} NH_3, and 3,800 to 6,900 mg L^{-1} P (Cramer et al. 1971). Liquid wastes seeping from the stacked manure average 4.5 to 11.0 L day^{-1} cow^{-1}, with 1,200 to 2,900 mg L^{-1} total N, 780 to 2,200 mg L^{-1} NH_3, and 64 to 500 mg L^{-1} P.

Manure produced by dairy cows housed in free-stall barns can be scraped by a front-end loader and stacked in a storage area for later use. In many of the newer setups, the manure is flushed by large volumes of water discharged a few times a day (Merkel 1981). The liquid waste from the flushing aisle is generally discharged to a series of lagoons for treatment. Effluent from the lagoon may be used as the flush water.

Manure collected by either scraping or flushing generally goes to a storage area. In some systems manure is immediately spread on land without storage, but this is not appealing to many dairy farmers primarily because of frequency of disposal. Transport of manure from the storage areas is dependent on the flow characteristics of the material. Dairy cattle manure can be classified as semisolid, semiliquid, or liquid (Sobel 1966). Semisolid manure will not flow with perceptible movement unless given mechanical assistance. Most fresh manure is in this category and, unless flushed, must be manually or mechanically transported. Semiliquid manure is material that has undergone dilution. This type of manure will slowly flow without mechanical assistance and contains between 5 and 15 percent total solids (Merkel 1981). Liquid manure generally contains less than 5 percent total solids (wet basis), flows freely without mechanical assistance, and is associated with feedlot runoff and effluents from milking parlors and treatment systems.

Dairy cattle manure in a solid or semisolid state can be transported mechanically by means of front-end loaders, conveyors, augers, or piston pumps. Hydraulic transport is generally used for handling liquid manure. Considerable information is available on the flow principles involved in hydraulic transport of liquid manure and on designing systems for moving the material via open channels or pipes to an initial storage facility (Merkel 1981, Midwest Plan Service 1985).

Alternative management systems for the manure from the initial storage facility include spreading in solid form, spreading in liquid form, immediate irrigation, and lagooning and irrigation. Storage and spreading in solid form usually involves short-term storage between the time of collection and land spreading. Land spreading in the liquid form has two major disadvantages: (1) cost of the system, and (2) odors associated with agitating and field spreading partially decomposed manure. Systems that use liquid from the initial storage area for irrigation also have the disadvantage of short-term storage availability, and hence wastes must be applied daily by irrigation regardless of weather conditions. Irrigation systems from lagoons allow for long-term storage and treatment of the waste prior to land application.

Disposal systems for liquid manure require separation of the liquid and solids fractions. Separation of settleable or suspended solids from liquid may be accomplished by gravity or employing mechanical devices. Gravitational separation includes sedimentation and flotation using tanks or lagoons; mechanical devices include liquid cyclones and screens. Use of screens is attractive to dairy operators for the following three reasons (Moore et al. 1975): (1) they reduce plugging of liquid handling equipment such as pumps, piping, and sprinkler nozzles, (2) they reduce biological loading on successive treatment components, such as anaerobic and aerobic lagoons, and (3) the solids removed by the screens can be recycled for bedding or feed. Hay, hayledge, silage, or other fibrous material removed from the manure by separator can be used as bedding material (Fairbank et al. 1975). Use of solids for bedding may negatively affect herd health (causing mastitis) and hence has had limited acceptance (Newton, personal communication 1992). The fiber is generally composted to reduce the level of mastitis-causing organisms in the bedding.

After liquids and solids from the manure are separated, the liquid portion is commonly transported to stabilization ponds (lagoons). In these ponds beneficial organisms stabilize the material so that it can be spread on the land or used as flush water for a recycle cleaning system.

Stabilization ponds can be classified according to the mode of degradation: aerobic, facultative, or anaerobic. Aerobic lagoons are aerated so that organic matter is oxidized by bacteria supported by free molecular oxygen. Aeration is most commonly supplied by mechanical aerators that provide sufficient agitation to ensure complete mixing.

Facultative lagoons provide an aquatic environment in which photosynthesis and surface oxygenation supply an aerobic zone in the upper strata. Two other zones exist below the aerobic zone—a facultative zone throughout the central portion and an anaerobic sludge layer at the bottom. The heavier suspended solids (including biologically formed floc) settle on the bottom and undergo anaerobic decomposition. Many lagoons used for treatment of dairy cattle manure were originally classified as aerobic, yet, in fact, they were truly facultative (Merkel 1981).

Anaerobic lagoons are stabilization ponds that can degrade organic matter in the absence of free molecular oxygen. Under anaerobic conditions, the microbial population derives its energy for cell synthesis by reducing oxidized compounds such as NO_3, SO_4, and carbohydrates. Reduction of NO_3 under anaerobic conditions is called denitrification, and considerable N may be lost by this process. For denitrification to occur in anaerobic lagoons the treatment system must have components where NH_4 is oxidized to NO_3 prior to entering the lagoon. Both facultative and anaerobic bacteria are present in anaerobic lagoons. When dairies have two lagoons, the first one generally is anaerobic and also serves as a settling basin, and the second one is facultative or aerobic using a mechanical aerator.

Nutrient losses during storage

Proper management of dairy cattle manure requires conservation of N for later use. Knowing where losses can occur is imperative to conserving N. High levels of NH_3 in freestall dairy barns have been measured, suggesting that manure in such barns might lose substantial quantities of N (Miner et al. 1975). The N is lost through hydrolysis of urea in the urine to NH_3, which is then easily lost by volatilization (Salter and Schollenberger 1939). Work by Muck and Steenhuis (1981) indicates that when barn temperature is greater than 20 °C and barn alleys are scraped only once a day, 80 percent of the urea N (which is approximately 40 percent of the total N in the manure) is lost by volatilization. The greatest N loss probably occurs on the barn floor from the time dairy manure is produced until the time it is spread (Muck and Herndon 1985).

Manure can be stored for months in bottom-loaded storage houses or tanks, and N losses will amount to less than 10 percent (Safley 1980, Muck et al. 1984). Nitrogen losses from anaerobic lagoons and storage have been studied by several investigators (Willrich 1966; Smith et al. 1971; Jones et al. 1973; Koelliker and Miner 1973; Booram et al. 1975; Safley 1980, 1981; Safley and Westerman 1981). However, the wide range of results reported makes it difficult to compare one storage design with another. Bottom-loaded manure storage, because of its crust, is generally believed to conserve N better than top-loaded storage (Muck and Steenhuis 1981).

The Midwest Plan Service (1993) gives estimates of typical N losses between excretion and land application as adjusted for dilution based on the waste handling system. For systems handling solid manure, estimated N losses for daily scrape and haul, manure pack, or open lot are 20 to 35, 20 to 40, and 40 to 55 percent, respectively. Estimates of N losses during land application based on application method are 15 to 30 percent for liquid broadcast, 1 to 5 percent for both solid and liquid broadcast with immediate cultivation, 0 to 2 percent for knifing of liquid, and 15 to 40 percent for sprinkler irrigation of liquid.

Proper management of dairy cattle manure also requires an understanding of where P and K losses occur during handling and storage. Phosphorus and K losses during storage are considered negligible except for those from open lots or lagoons (Midwest Plan Service 1993). In open lots about 20 to 40 percent of the P and 30 to 50 percent of the K can be lost by runoff and leaching. However, much of this P and K can be recovered by runoff control systems such as settling basins and holding ponds. Up to 80 percent of the P in lagoons can accumulate in bottom sludges and hence is lost as a nutrient supply unless the sludge is removed from the lagoons and applied to land.

Land Application of Manure

Land application of animal manure has been practiced for centuries in the temperate zones. The practice developed partly because there was no other place to put the material but also because of the agronomic benefits. Application methods for dairy cattle manure depend on the fluidity of the material. Liquid manure containing less than 5 percent solids can be handled by most irrigation systems (Midwest Plan Service 1975b). This level of solids is typical of that found in feedlot runoff or effluents from a lagoon system or milkhouse. The type of irrigation system selected depends upon topography, soil type, and cropping practice. Disadvantages of irrigation include a high initial investment, high operating costs for pumping, the necessity for good management to avoid runoff or groundwater pollution, high labor demand with low-cost irrigation equipment, odor problems, and NH_3 loss by volatilization.

Liquid manure with 4 percent solids or less can also be applied to land via irrigation known as surface spreading. Material from pipeline systems can be spread by gravity using open ditches, flat irrigation tubing, or gated pipe (pipe with openings at set distances apart). Types of surface irrigation for dairy cattle manure in the surface spreading category include border irrigation, furrow irrigation, corrugations, and wild flooding. In all cases the material should not be applied to a wet area. The system also should be shut off before water reaches the low end of the field to eliminate runoff. Of the four types of land spreading systems for dairy cattle manure, wild flooding has the most uneven water distribution (Midwest Plan Service 1975b).

Semisolid dairy cattle manure, or slurries, have 4 to 15 percent solids and can be applied using manure guns or tank wagons. Large-bore irrigation nozzles can handle heavy slurries (up to 15 percent solids) as well as liquid materials with low solids content. These large sprinklers generally have a capacity of 23 to 91 m^3 hr^{-1} and can cover from 0.2 to 0.8 ha (Midwest Plan Service 1975b). Tank wagons are available for transporting fluid slurries and have capacities ranging from about 1.6 to 11.3 m^3. Slurries must be agitated in the storage tank before they can be satisfactorily pumped into tankers. Tank wagons may either apply manure to the soil surface or inject the manure into the soil with chisel-type injector shanks or moldboard plow attachments. Injection is desirable both for conserving nutrients and to reduce odor problems.

Manure with 20 percent or more solids is generally handled as a solid. Most solid manure spreaders are box type, although open-tank spreaders are available. Ideally, manure should be distributed evenly to the land, but the effectiveness of this distribution depends on the characteristics of the material being spread.

Proper land application of dairy cattle manure should include crediting of the fertilizer value of the material. Manure management system designs are generally based on N excretion loads for a dairy and accepted land application rates for N. Application rates are based on N rather than P for two reasons: (1) the total N content of manure is higher than the total P content and (2) P tends to bind to soil particles (except on very sandy soils) and hence is primarily of environmental concern only if erosion occurs, whereas N is less likely to bind and is therefore more likely to contaminate groundwater.

The most effective method for gauging the nutrient content of a manure is to have samples analyzed by a commercial or university laboratory. Large farm-to-farm variation can occur in nutrient content due to storage, handling, livestock feed, or other farm management differences. Several investigators (Good et al. 1991, Bundy et al. 1992, Wolkowski 1992) have developed methods for calculating the total nutrient contribution of manure, which is derived by multiplying the amount applied by the nutrient content from standard tables. The Midwest Plan Service (1993) gives the approximate fertilizer values for N, P, and K in solid dairy manure as 4.5 g kg^{-1}, 0.7 g kg^{-1}, and 2.5 g kg^{-1}, respectively. For liquid pit manure, the approximate fertilizer values for N, P, and K are 3.7 g L^{-1}, 0.7 g L^{-1}, and 1.9 g L^{-1}, respectively. The approximate values for N, P, and K in lagooned dairy manure are 0.5 g L^{-1}, 0.1 g L^{-1}, and 0.4 g L^{-1}, respectively (Midwest Plan Service 1993).

The nutrients contained in dairy cattle manure (other than those in lagoon effluent) are not immediately available to crops but are released over time. The rate of release depends upon the amount of organic matter applied along with nutrient content, climate, and soil type. Wolkowski (1992) indicates that the N credit increases each successive year of application (up to 3 consecutive years) by approximately 30 percent. The Midwest Plan Service (1993) indicates that organic N released by mineralization during the second, third, and fourth cropping years after initial application is usually about 50 percent, 25 percent, and 12.5 percent, respectively, of that mineralized during the first cropping season. Their worksheet requires calculation of the residual N released by mineralization from previous years as part of the overall N budget. In warm, humid locations with well-aerated, sandy soils, mineralization is rapid and essentially complete in 1 year. However, when manure is applied to grain crops at planting, the availability of N from mineralization does not correspond to plant needs over the season. In contrast to N availability, nearly all of the P and K in manure is available for plant use during the year of application. After a few years of regular waste applications, the amounts of P and K available are about the same as they were after 1 year of application (Midwest Plan Service 1993).

Worksheets provided in Midwest Plan Service (1993) for crediting nutrients in dairy cattle manure provide instructions for calculating the nutrient requirements of the crop and then determining the amount of land necessary to use all of the available waste. Applying enough manure to meet N requirements more than adequately meets crop needs for P and K (Midwest Plan Service 1993). Over time this may cause high accumulation of P, K, and salt in the soil. The economic value of manure fertilizer can be calculated from its available N, P, and K and determining the equivalent commercial fertilizer prices. The equivalent values will change over time as the costs of commercial fertilizer and handling practices change.

One concern with manure applications is soil salinity. Heavy manure applications can increase soil salinity, especially in arid regions where little or no leaching occurs. Salts can inhibit plant growth and depress yields. Sodium and K can alter soil structure and reduce water movement rates. Use of heavy manure wagons can also affect yields by compacting wet soils.

Alternative Uses Of Dairy Manure

In the past few years biogas generation from animal manure has received more attention. Methane production from livestock manure has been shown to be an easily established fermentation process (Stafford et al. 1980, Van Brakel 1980). One-third of the total energy content is released in the form of methane (Sobel and Muck 1983). Hashimoto et al. (1979) and Hill (1982) report that although dairy cattle manure is less readily biodegradable than beef, poultry, or swine manure, the potential for methane production and the benefits of its use on dairy farms are substantial. One problem with using dairy cattle manure for methane production is the large fraction of settleable and floating solids, causing difficulties in pumping the liquified manure as well as accumulation of solids in the base of the reactor vessel (Ecotope Group 1977, Bartlett et al. 1977, 1980; Abeles et al. 1978).

Anaerobic digesters have been successfully used to produce methane in the psychrophilic (below 20 °C) (Lo and Liao 1986), mesophilic (30 to 40 °C) (Lo et al. 1984, 1986; Erdman 1985; Summers et al. 1987), and thermophilic (50 to 60 °C) (Wohlt et al. 1990) temperature ranges. Major concerns about using dairy cattle manure to produce methane include (1) the necessity for and difficulty of mixing, (2) the current lack of process controls for daily operation that are needed to minimize management time and provide the operator with sufficient warning of impending biological upset, (3) the impracticality of long-term methane storage, and (4) the effects of antibiotics in the manure on methane production (Midwest Plan Service 1982). The land area needed for using dairy manure nutrients is not reduced by digester systems because the total amounts of N and P remain in the digester effluent.

Two other alternative uses for dairy cattle manure are composting and refeeding. Composting is a process in which the volatile solids are digested by aerobic microorganisms. Because the process is aerobic, it is relatively free of offensive odors. Dairy cattle manure from stanchion or free-stall barns is considered to be a good material for composting because the addition of the bedding brings the material to a favorable moisture content. Stable compost can be obtained in 19 to 56 days depending on moisture content, air distribution, and temperature (Willson and Hummel 1972). The primary potential benefit of composting is that a value-added product is produced. This product (compost) is useful not just on the farm, but also off the farm, such as in the horticulture industry.

Research on the feeding value of screened manure solids (SMS) obtained from dairy cattle has shown that the SMS are lower in crude protein and higher in lignin and other fiber constituents than the manure prior to screen separation (Johnson et al. 1974a, 1974b). Digestibility and feeding trials have shown that dairy cattle can successfully use this recycled material when it is included as a small percentage of the diet (University of Kentucky 1979). However, the solids are a low-quality feed ingredient, and therefore their use is limited to nonlactating cows or heifers. Hence overall usefulness of the material is limited (Newton, personal communication 1995).

Although methane generation, composting, and refeeding have been shown through research to be successful uses of dairy cattle manure, none of these techniques are currently important on a regional or national scale (Newton, personal communication 1995).

Agronomic and Environmental Effects of Dairy Cattle Manure Application

Application of dairy cattle manure to land affects both the physical and chemical properties of the soil. Manure application, regardless of form, improves tilth, increases water-holding capacity, lessens wind and water erosion, improves aeration, and promotes beneficial organisms (Midwest Plan Service 1985). When manure is applied to the soil surface, it tends to help prevent soil crusting. When injected or mixed with the soil, the manure decomposes more rapidly and the products of decomposition improve soil structure and the general physical condition of the soil (North Carolina Agricultural Extension Service 1973). Application of dairy cattle manure to cropland increases the organic matter content of the soil which in turn improves long-term aggregate stability and decreases bulk density. The result is increased infiltration. Unger and Stewart (1974), Kumar et al. (1985), and Sommerfeldt and Chang (1987) all noted an improvement in soil water retention (in the range of 0 to 15 bar matric potential) for soils receiving manure application.

Dairy cattle manure contains significant amounts of the primary plant nutrients (N, P, and K) as well as other essential plant nutrients, including Ca, S, Mg, and Cl. Considerable research has been done on using dairy cattle manure for crop production (University of Kentucky 1979). Unfortunately, dairy cattle manure has often been applied to land with disposal of the material being the main objective and use of it as a nutrient resource being a secondary concern. The primary objective in using dairy cattle manure should be safe, pollution-free recycling of the manure nutrients. Considerations for proper use of dairy cattle manure should include the texture and fertility level of the soil, the nutrient requirements of the crop to be grown, the nutrient content of the manure, and local climatic factors that will affect the fate of each of the major nutrients. Dairy cattle manure is commonly used for corn production (Safley et al. 1984, Beauchamp 1986) and on grasslands (Hubbard et al. 1987, 1991).

The major environmental concern with land application of dairy cattle manure is possible contamination of surface waters and groundwaters with excess N and P. Heavy applications of dairy cattle manure have been linked to eutrophication of surface water bodies. Phosphorus is the primary cause of eutrophication, although N may also contribute to this problem. One area of the country where eutrophication has been clearly linked to dairy cattle manure is the area near Lake Okeechobee, FL. Since the early 1970's, dairies north of the lake have been cited as the number one source of P (Sauber 1989). Nitrate leaching is the primary concern for groundwater contamination. Both Hubbard et al. (1987) and Sewell (1975) observed NO_3 leaching to shallow groundwater where excess quantities of dairy cattle manure were applied.

Problems with dairy cattle manure also may occur from surface runoff and leaching in feedlot or land application areas, or by leakage from lagoons. Rainfall-induced surface runoff may carry urine and feces into adjacent streams, rivers, or lakes. Hubbard et al. (1987) showed that as land application rates increased, proportionately more N was lost by surface runoff than by leaching. Dairy cattle manure applied to the soil surface is immediately available for movement by surface runoff, particularly if it has been applied to frozen land. During the spring thaw and snowmelt, nutrients from manure may move freely with runoff.

Water contamination from manure application can occur when application rates are greater than the assimilative capacity of the soil and crops, or when manure is left on the soil surface rather than being incorporated and hence is subject to movement by surface runoff. Application rates may exceed assimilative capacity of the soil when the land area available for manure application is too small relative to the number of cattle or where manure is repeatedly applied to fields closest to the barns or feeding areas. Surface water or groundwater can also be contaminated by farm managers applying commercial fertilizers without accounting for the nutrient value of the applied dairy cattle manure. Unfortunately, some major dairy operations still do not account for nutrients in manure applications when calculating commercial fertilizer application rates. A contributing cause to environmental contamination from dairy cattle manure is the need to get rid of the material on a daily basis. Since milking and feeding areas must be cleaned daily, manure that is generated must go somewhere. Once the holding tank or lagoons are full, the material within them must be applied to land regardless of weather, soil, or crop conditions.

Air quality within or surrounding dairy facilities or where manure is land applied is also a concern. Odors can be a nuisance to producers and can cause complaints and even lawsuits from neighbors. Organic compounds from uncontrolled decomposition of manure include odorous gases such as amines, amides, mercaptans, sulfides, and disulfides (Midwest Plan Service 1985). Noxious gases can irritate both livestock and operators and can be harmful and even lethal. Preventing production and accumulation of gases in the livestock area is accomplished through frequent cleaning of floors, not overfilling storage tanks, not storing manure in facilities for longer than 6 months, and providing adequate ventilation. Immediate plowdown or injection of manure spread on the field will reduce odors.

A relatively new air quality concern is the emission of gases from livestock manure sources. A general warming of the atmosphere due to increases in gases that adsorb radiant energy is called the greenhouse effect, and these gases are known as greenhouse gases. Methane, which is released from decomposing animal manures, is a greenhouse gas. Methane losses from all livestock manure sources account for 37 percent of all greenhouse gas emissions from U.S. agriculture based on carbon dioxide warming equivalents (Center for Rural Affairs 1992). The manure management system is critical in determining the amount of methane emissions. Approximately one-fifth of all methane from U.S. livestock sources is derived from anaerobic lagoons.

Improving Management of Dairy Cattle Manure

Dairy cattle manure must be thought of as a resource and must be managed and used economically without adverse environmental impacts. Unfortunately, many manure managers still think of this material as a waste, that is, something to get rid of, so the material is often disposed of without careful attention to matching crop, soil, and environmental constraints to the manure supply. There is ample evidence, however, that properly managed dairy cattle manure can be used to supply some or all of the nutrients to crops with economic profitability and no environmental harm.

Government regulations have been passed and are enforced in a number of states to protect surface water and groundwater quality from adverse impact by dairy cattle manure. These regulations may specify the size of land-use areas needed in relation to the number of cows and may also require monitoring of wells. For example, the state of Texas requires producers milking herds of more than 250 cows to have a permit stipulating that their dairy produces no nutrient discharge. In south Florida, the State's Department of Environmental Regulation reviews permit applications with the goal of balancing each dairy's nutrient use and discharge. Producers are required to have adequate land disposal resources for manure (Sauber 1989). One weakness of such regulations is that in some states they apply only to new dairies or dairies over a certain size and hence do not protect surface water and groundwater quality from existing or smaller operations.

As discussed in this report, a number of different options exist for using dairy cattle manure without adverse environmental impact. Education and transfer of these technologies to dairy producers is critical so that the manure can be used for supplying nutrients or obtaining energy. Once the material is viewed as a resource rather than a waste and is properly managed, it will be easier to meet government regulatory standards on air and water quality. Also, use of the manure as a resource should lower commercial fertilizer and energy costs and hence result in improved cost/benefit ratios as compared to earlier manure disposal practices.

Research is needed to improve the efficiency and safety of management practices for handling and using dairy cattle manure. Such research should be geared toward developing or improving (1) methods for reducing N losses from manure while in the barn or in storage, (2) methods for using the manure for energy, (3) uniform and efficient application procedures for applying manure to land, (4) cropping systems that efficiently use the manure while providing feed for dairy cattle, and (5) application rate guidelines that result in adequate nutrients for crop growth without adversely affecting the quality of the air, surface water, or groundwater. The most critical information needed at this time pertains to loading rate guidelines. Current research with a triple cropping system (coastal bermudagrass, abruzzi rye, and corn) at Tifton, GA, is being used to determine environmentally safe and economically sustainable liquid dairy manure rates for center pivot application. This information can only be developed by simultaneously determining both crop response and water quality effects under a range of manure application rates (Hubbard et al. 1991, Vellidis et al. 1991, Williams et al. 1991). Similar research is needed for other cropping systems over a range of soil and climatic conditions.

Concerns about environmental impacts of dairy cattle manure have caused changes in laws and management practices in a number of states. In some states new laws now require farmers to use best management practices including monitoring surface water and groundwater quality. These laws have resulted in new dairies purchasing more land on which to use the manure than was previously common practice, and in some states dairies that were unable to meet environmental standards have either moved or gone out of business. Along with the research needs, education-extension packages are needed to aid both existing and new dairies in developing cropping and manure-use systems that meet environmental standards. Extension publications from Wisconsin (Good et al. 1991, Bundy et al. 1992, and Wolkowski 1992) are good examples of information for dairy producers that show how to credit manure applications for nutrient management and protection of water quality. Similar information is needed in all states to help dairy producers use manure as a resource.

Along with research and education-extension packages, economic incentives are needed to accomplish widespread use of dairy cattle manure. A program in Lancaster County, Pennsylvania, currently connects manure producers with interested buyers and could serve as a countrywide model (Anonymous 1992). Farmers are purchasing the manure as a replacement for commercial fertilizer, with some of the manure being transported as far as 500 km from the source. Economically the marketing area is generally limited to about a 150-km radius (Anonymous 1992). Similar programs could work well elsewhere, although some type of subsidy (free material, transportation, or application) may be necessary initially.

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Chapter 6

Agricultural Uses of Coal Combustion Byproducts

R.F. Korcak

Overview of Combustion and Its Byproducts

During combustion, fly ash, bottom ash, and boiler slag are produced. Fly ash produced from the burning of coal has become a generic term for all types of coal combustion byproducts. Specifically, fly ash is that portion of the ash stream composed of particles small enough (0.001 to 0.1 mm) to be carried from the boiler in the flue gas. These particles are either mechanically captured or emitted via the stack. Bottom ash and boiler slag are residues found in the furnace and are common to all types of coal combustion. Both materials generally have a particle size within the range of 0.1 to 10 mm. The amounts of boiler slag produced in the future are projected to decrease due to new boiler technologies. Currently, the ash stream consists of 5 percent boiler slag and about 25 percent bottom ash.

Total ash production varies considerably with the type of coal consumed as well as the source. Anthracite coal produces the highest ash content (about 30 percent), and ash production from bituminous coal can range from 6 to 12 percent ash (U.S. Environmental Protection Agency 1988). Subbituminous and lignite coals produce a range of ash contents from 5 to 19 percent. Currently coal combustion produces, on average, about 10 percent ash (U.S. Environmental Protection Agency 1988).

A number of other byproducts are possible depending upon the type of combustion process and the methodology used to reduce gaseous emissions, such as sulfur oxides. These include flue gas desulfurization (FGD) byproducts (which can be wet or dry), fluidized bed combustion (FBC) byproducts, and coal gasification ash. Coal gasification ash results from the conversion of coal into a synthetic gas or liquid fuels. The ash produced from coal gasification is similar to fly ash, and therefore these two types of ash will not be discussed separately.

FGD byproducts result from postcombustion treatment (scrubbing) of the flue gas with an absorbent [usually lime (calcium oxide), limestone, or dolomite] to reduce S emissions. Such treatment may be performed under dry or wet conditions, and these conditions affect the moisture status of the end product. In the wet method, flue gases pass through a slurry of absorbent in a contact chamber. In the dry method, a fine spray of absorbent is injected into the flue gas stream as it passes through the contact chamber. The water in the fine spray evaporates in the gas stream, leaving a dry powder end product. The wet method tends to be more efficient (about 90 percent) than the dry method (about 70 percent) for removing S from flue gases. Thus, the dry method, or scrubbing, is usually performed when low-S coal is consumed.

The major types of FGD systems currently in use are listed in table 19. These types are classified as recovery or nonrecovery systems based on whether they produce a salable end product such as S, sulfuric acid, or liquid sulfur dioxide. Recovery systems produce a salable end product; nonrecovery systems do not. Because S-based end products from nonrecovery systems have limited industrial uses, alternative uses for these materials need to be developed. Of the FGD systems listed in table 19, direct lime and direct limestone (wet, nonrecovery methods) are the most widely used (U.S. Environmental Protection Agency 1988). The alkaline fly ash scrubber in these two systems is used primarily for S removal from highly alkaline western coals. The dual-alkali process uses a mixture of lime and Na salts for S removal.

Dry scrubbing methods such as spray-drying and dry sorbent injection have been developed by the industry since 1988. A newer FGD system currently under study is the Pircon-Peck process. In this system, Ca phosphate (rock phosphate) is used as an absorbent rather than Ca carbonate (limestone). The byproduct of this process contains both gypsum (Ca sulfate) and acidic P. The initial byproduct is then ammoniated, producing a mixture of gypsum and ammonium phosphate. This mixture provides four of the five nutrients needed in largest quantities by crops. If the Pircon-Peck process is found to be economically feasible, it could make a significant contribution to fertilizer needs.

The simultaneous combustion of coal and an absorbent (usually limestone or dolomite) in FBC results in end products that are very different than those produced from combustion of coal alone. In the furnace Ca absorbs S, thereby reducing flue emissions of S and producing large amounts of a dry byproduct. The

Table 19. Some examples of nonrecovery and recovery types of FGD systems. Recovery systems produce recyclable end products such as elemental sulfur

Nonrecovery systems		Recovery systems	
Wet	Dry	Wet	Dry
Direct lime (CaO)	Spray drying	Wellman-Lord	Alumina/Cu sorbent
Direct limestone	Dry sorbent injection	Magnesium oxide	Activated C sorbent
Alkaline fly ash			
Dual-alkali			

Source: U.S. Environmental Protection Agency (1988).

bottom and fly ashes of FBC systems contain substantially higher concentrations of Ca (mostly as calcium sulfate but with some calcium oxide) than in conventional systems and have an alkaline pH (usually about 12).

A similar new technology for removing S involves the use of a limestone injection multistage burner (LIMB). A Ca-based sorbent is injected into the burner to achieve S removal. Existing burners can be retrofitted to become a LIMB. The dry byproduct obtained from this technology (called the LIMB process) is easier to handle than wastes from a traditional wet scrubber.

Amounts of Byproducts Produced

Large quantities of coal are used annually to produce electricity. For example, coal consumption for this purpose in Georgia alone averages about 24.5 million Mg annually and results in 2.0 million Mg of fly ash and 0.5 million Mg of bottom ash (McIntosh et al. 1992). Therefore, the weight of fly and bottom ashes produced from coal combustion amounts to about 10 percent of the weight of the coal consumed. These figures do not reflect the higher amounts of ash that will result from new technologies such as FGD, a process designed in response to the Clean Air Act. Installation of FGD scrubbers at selected sites in Georgia alone will produce an additional 1 million Mg of ash annually (U.S. Environmental Protection Agency 1988).

Nationally about 62.6 million Mg of fly and bottom ash were produced in 1984 (U.S. Environmental Protection Agency 1988) (table 20). Projected future use of coal will increase this figure to about 109 million Mg annually by the year 2000. These figures do not include the amounts of FGD and FBC material generated.

About 95 percent of current FGD byproduct production is from nonrecovery-type systems (U.S. Environmental Protection Agency 1988). FGD byproduct production in 1985 was about 14.5 million Mg (U.S. Evironmental Protection Agency 1988). This figure includes byproducts from all types of systems—recovery or nonrecovery, and wet or dry. It is estimated that this figure will increase to about 45.4 million Mg annually by the year 2000.

Both FBC and other Ca-based dry byproduct technologies such as the LIMB process are just beginning to be used significantly. Therefore annual production figures for these byproducts from these types of systems are not available. It has been estimated that a 1,000 megawatt FBC plant would generate about 1,800 Mg of dry waste per day or about 0.64 million Mg annually (Ruth 1975). About 110 FBC plants are currently in operation, with an additional 13 plants under construction (J. Tishmack, personal communication). Residue production is over 18.2 million Mg per year. The gypsiferous material resulting from these systems may be among the most suited for agricultural use.

Table 20. Past and projected amounts of byproducts produced by the coal combustion industry

Byproduct type	Byproduct production (million Mg) 1984	1991	Projected for 2000
Ash			
Total*	64	65.2	111
Fly	—	47.3	79
Bottom	—	12.2	27
FGD	15	16	46

* Includes boiler slag.

Source: U.S. Environmental Protection Agency (1988).

Table 21. Concentration ranges for major constituents of ash

Element	Element concentration (g kg^{-1}) In fly ash	In bottom ash
Essential nutrients		
Calcium	5–177	8–51
Iron	8–289	27–203
Magnesium	5–61	4–32
Potassium	2–35	7–16
Silicon	196–271	180–273
Other elements		
Aluminum	11–144	88–135
Sodium	1–20	2–13
Titanium	<1–16	3–7

Source: Utility Solid Waste Activities Group (1982).

Chemical Composition of Ash

Conventional fly ash and bottom ash

The chemical constituents of ash can vary greatly depending upon the coal type, source, and plant operating parameters. Major constituents include Al, Ca, Fe, Mg, K, Si, Na, and Ti. These eight primary constituents make up 95 percent of the ash, and five of the eight are important nutrients in agriculture. The concentrations of these primary constituents are listed in table 21.

Trace element concentrations in ash are also variable and can have a direct impact on the potential agricultural use of the ash. Ranges of trace element concentrations present in a broad spectrum of ash samples are presented in table 22. The values in table 22 are average concentrations, and significant variations in trace element levels occur for various coal sources (eastern, midwestern, and western). Midwestern coal ash is usually highest in Cd, Zn, and Pb; Ba and Sr have the highest concentrations in western coal. Selenium level tends to be greater in eastern and midwestern coals.

Coal cleaning, prior to combustion, can significantly reduce elemental concentrations of S, Se, and other trace elements in ash. The cleaning is performed using physical (usually density differences separating out pyritic S), chemical, or biological precombustion cleaning. The latter two methods are newer and not currently used extensively. Coal cleaning can also have a significant impact on the amount of ash generated. In Virginia, raw coal produced an average ash yield of 9.7 percent (yield based on original weight of coal), whereas cleaned coal produced a yield of 5.7 percent (Randolph et al. 1990). Precombustion cleaning of coal is one of several categories of clean coal technology currently being funded and developed under the Department of Energy (U.S. Department of Energy 1992).

Elemental concentrations of ash also vary with the particular portion of the ash stream sampled. Fly ash contains significantly higher quantities of As, Cu, and Se than bottom ash. Distribution of elements in the ash stream is highly dependent on boiler temperature. Some components of ash such as elemental S and Hg are essentially completely volatilized, thereby reducing their concentrations in bottom ash in conventional coal burning plants.

Currently, fly ash types are classified on the basis of major components. Class C fly ashes contain less than 70 percent but greater than 50 percent of a combination of silica, alumina, and iron oxides and are usually denoted as high-lime, western ashes. If the content of silica, alumina, and iron oxides exceeds 70 percent, ashes are classified as class F; these types are usually generated from eastern coals (Environmental Management Services 1992).

Table 22. Trace element concentration ranges in ash averaged over all ash and coal types

Element	Concentration ($mg\ kg^{-1}$)
Essential nutrients	
Boron	10–1,300
Copper	3.7–349
Manganese	56.7–767
Molybdenum	0.84–100
Zinc	4.0–2,300
Other elements	
Arsenic	0.5–279
Barium	52–5,790
Cadmium	0.1–18
Chromium	3.4–437
Cobalt	4.9–79
Fluorine	0.4–320
Lead	0.4–252
Mercury	0.005–4.2
Nickel	1.8–258
Selenium	0.08–19
Silver	0.04–8
Strontium	30–3,855
Thallium	0.1–42
Vanadium	11.9–570

Source: Tetra Tech, Inc. (1983).

A modification of this fly ash classification system has been proposed. This modification is based on a more detailed chemical composition (Roy et al. 1981). The three basic groupings proposed are silica (Si–Al–Ti oxides), calcic (Ca–Ma–K–Na oxides), and ferric (Fe–Mn–S–P oxides). Such a classification system helps to identify potentially useful byproducts. Expansion of the classification system to include agriculturally related parameters such as plant nutrient availability indices and potential elemental phytotoxicity indices would facilitate communication between different research groups and expedite development of cost-effective and environmentally beneficial uses of these ashes.

Flue gas desulfurization (FGD) byproducts

Regardless of the type of process used to scrub the flue gas, all FGD products include spent reagent in combination with sulfites or sulfates plus unreacted reagent. Additionally, the FGD material may contain water (in wet processes) and coprecipitated fly ash. The quantity of reagent used is usually proportional to the S content of the coal burned but is also a function of the percent SO_x recovery desired and system operating parameters. Generally, wet scrubbers produce material that is slightly smaller in particle size (0.001 to 0.05 mm) than dry scrubbers produce (0.002 to 0.074 mm). Wet scrubber sludge can vary from 16 to 43 percent moisture.

The chemical composition of FGD sludges (table 23) varies depending upon the type of FGD process, type of coal used, quantity of reagent used, amount of fly ash present, and whether or not forced oxidation was used in the treatment process. The degree of forced oxidation used in the process that produces FGD sludges has a significant impact on the potential use of the material in agriculture, since increased oxidation increases the amount of sulfates compared to the amount of sulfites present in the end product.

Dual-alkali and spray drying systems that use a Na absorbent produce FGD sludges containing sodium sulfate (oxidized) or sodium sulfite (reduced). Since Na deteriorates soil structure, these Na-containing FGD sludges will probably not be used for agriculture.

A comparison of the primary chemical components of liquors produced from a direct-lime (calcium oxide) FGD process and a dual-alkali FGD process is shown in table 24. The differences in Ca and Na contents of the byproducts from the two processes are readily apparent and must be taken into consideration when evaluating whether the byproducts are agriculturally useful. The ratio of sulfate to sulfite affects the solubility of the end product. Sulfites are lower in solubility. Installation of an oxidizing step in the FGD process, although an additional expense, aids not only in increasing the solubility of the end product but also increases the potential for agricultural use of the product because it will be higher in gypsum. On the other hand, there is preliminary evidence that sulfite sludges applied to soils several weeks prior to planting are oxidized to sulfates before the crops begin to grow (K.D. Ritchey, personal communication). Consequently, negative effects on plant growth are avoided if the sulfite sludges are applied before rather than during seeding. Investigations on the behavior of sulfite materials in the soil and plant system are being performed at the Agricultural Research Service laboratory in Beckley, WV (R.B. Clark, personal communication).

Table 23. Effect of FGD process and coal source (eastern and western) on the percentage of chemical components and end products (dry weight basis) produced in FGD sludge

FGD process and coal source	Chemical components and end products in FGD sludge (%)			
	Ca-sulfate	$CaSO_3 \frac{1}{2}H_2O$	Ca-sulfite	Fly ash
Direct lime				
Eastern coal	15–19	13–69	1–22	16–60
Western coal	17–95	2–11	0–3	3–59
Direct limestone				
Eastern coal	5–23	17–50	15–74	1–45
Western coal	85	8	6	3
Alkaline fly ash				
Western coal	20	15	—	65

Source: U.S. Environmental Protection Agency (1988).

Trace element concentrations in the solid and liquid component of wet scrubber sludges have been tested, and the results are shown in table 25. Most trace elements remain in the solid FGD material, but a fair amount of B is found in the liquor portion. The percent of soluble B present in the liquor portion may be of significance if liquor is used on the soil and plant system. Plants are sensitive to B concentrations. High available B levels induce plant toxicity. However, where B is deficient, additions of B-bearing sludges may be beneficial. Such potential benefits from careful management are explored later in this section.

Fluidized bed combustion (FBC) byproducts
FBC byproducts also vary in elemental composition. The range of elemental concentrations from a representative FBC plant using eastern coal are presented in table 26. The large amounts of Ca present in the byproduct are primarily in the form of gypsum and unreacted sorbent, calcium oxide. A typical spent bed material has an aqueous pH of about 12 and contains (in percent dry weight) 52 percent calcium sulfate, 33 percent calcium oxide, 0.6 percent calcium sulfite, 0.8 percent magnesium oxide, 0.3 percent sodium chloride, 0.02 percent phosphate, 4.5 percent metal oxides (mainly Fe and Al), and 7 percent silicon dioxide (Korcak 1988). This Ca-rich byproduct should be especially useful because it contains high amounts of acid neutralizers (Ca oxide) and the relatively soluble and mobile calcium sulfate from the gypsum. Generally, trace element concentrations of FBC byproducts are similar to those of other coal combustion byproducts, and the levels of the trace elements will vary depending primarily upon the constituents of the coal and sorbent used during combustion.

Organic Composition and Mineralogy of Coal Combustion Byproducts

There are many incompletely oxidized organic compounds in fly ash. Roy et al. (1981) lists a number of carcinogens and mutagens in ash. It is difficult to track organics in flue gases exiting power plants due to climatic and atmospheric effects on the composition of air entering the burners. Also, changes that occur in the stack prior to fly ash capture may not accurately reflect potential toxicity estimates.

Organics have received little attention in studies on agricultural use of coal combustion byproducts. Research on the transformation and fate of organics in the soil and plant system is difficult. Additionally, stockpiled, weathered ash may present a different organic composition than fresh ash. In assessing the potential dangers of using coal combustion byproducts for agricultural purposes, it is difficult to identify the primary hazards to human health. Based on past studies in related areas, it appears that the primary hazards would be via direct inhalation by operators applying these materials rather than via plant uptake and food consumption. However, the potential hazards for contamination by organics needs to be documented.

Table 24. Chemical composition and pH of liquors emanating from direct-lime and dual-alkali FGD processes (based on burning eastern coal). Concentrations are in mg L^{-1}.

Property or component	Direct-lime	Dual-alkali
pH	8–9.4	12.1
Potassium	11–28	320–380
Sodium	36–137	53,600–55,300
Calcium	660–2,520	7–12
Magnesium	24–420	0.1
Sulfate	800–4,500	80,000–84,000
Sulfite	0.9–2.7	—

Source: U.S. Environmental Protection Agency (1988).

Table 25. Trace element concentration ranges in wet FGD solids and liquors

	Trace element concentration (mg kg^{-1})	
Element	In solids	In liquors
Essential nutrients		
Boron	42–530	2–76
Copper	6–340	<0.01–0.5
Other elements		
Arsenic	0.8–52	<0.01–0.1
Cadmium	0.1–25	<0.01–0.1
Chromium	1.6–180	<0.01–0.3
Fluoride	266–1,017	0.2–63
Mercury	0.01–6	<0.01–0.1
Lead	0.2–290	<0.01–0.5
Selenium	2–60	<0.01–1.9

Source: U.S. Environmental Protection Agency (1988).

In contrast to the lack of information on organics, there is considerable mineralogy data for coal combustion byproducts. Most of this work has been performed on fly ash and has examined particulates. Davidson et al. (1974) found a strong association between fly ash particle size and trace element concentration. Concentrations of Se, Ca, As, Pb, Ni, Cr, and Sn increased with decreasing particle size. Similarly, Phung et al. (1979) found enhanced levels of B, Cr, Mo, Ni, As, and Se in fly ash particle sizes less than 53 µm. Fly ash could become more useful in agriculture if these small fly ash particles could be removed at the power plant site so that the risk of trace element problems are reduced. However, this additional operation may not be economically feasible.

Finer materials should be carefully analyzed and applied to land on a prescription basis according to needs. Bottom ash and FGD materials not mixed with fly ash could probably be used in larger amounts. The same would hold true for bottom ashes from FBC and newer technologies such as the LIMB process. However, these indications, that finer particulates are higher in trace metals, needs further documentation on the full range of byproducts considered for agricultural use.

Only recently have studies been initiated on the mineralogy of coal combustion byproducts applied to agricultural soils. The short- and long-term fate of mineral forms in the soil system needs to be examined. Korcak (1988) studied the effects of FBC materials on the soils of an apple orchard. He made an application of 112 Mg ha^{-1} within the rows of an established apple orchard. Since FBC byproduct materials are pozzolanic, that is, they form a hard cementlike surface when reacted with lime, the surface-applied material formed a porous cement that prevented weed growth for up to 4 yrs after application. During the 6 yrs after application, cumulative yields in the orchard increased for three of four cultivar-rootstock combinations. Foliar Mg levels from high-FBC applications increased initially but decreased with time. The decrease over time resulted from the greatly increased Ca status and the decrease in Mg levels in the surface soil horizons caused by leaching.

These apple plots were reexamined 12 yrs after the initial application and 5 yrs after the plots were plowed. X-ray diffraction patterns of remnant cemented pieces of the applied spent bed ash showed that most of the original calcium oxide had converted to calcium carbonate (calcite). Besides calcite, the other dominant mineral present was quartz. Secondary minerals present were gypsum and ettringite. The formation of calcium carbonate with time is expected and creates a soil pH that will not exceed 8.3 (the equilibrium pH for calcium carbonate). Surface pH

Table 26. Major and trace element concentration ranges in fluidized bed combustion (FBC) byproducts compared to ranges normally found in soils

	Element concentration	
Element	In FBC byproducts	In soil
	.... $g\ kg^{-1}$	
Calcium	240–460	7.0–500
Aluminum	4–20	40–300
Sulfur	72–140	0.1–20
Iron	<1–16	7.0–550
Magnesium	5–12	0.6–6
Potassium	<1–8	0.4–30
	.... $mg\ kg^{-1}$	
Phosphorous	380–500	50–2,000
Manganese	210–685	200–3,000
Boron	95–170	2–100
Molybdenum	0.12–0.28	0.2–5
Copper	12–19	2–100
Zinc	29–105	10–300
Nickel	13–29	5–500
Lead	1.5–7.5	2–200
Cadmium	0.5	0.01–0.7
Chromium	9–23	5–1,000
Selenium	0.16–0.58	0.01–2

Sources: Page et al. (1979) and Stout et al. (1988).

values from these plots after 12 years were about 7.6. However, the mineral ettringite is unstable at pH levels less than 10, and this mineral, once solubilized, raises pH even further. Therefore, the presence of even trace amounts of ettringite (which was identified in these soils) indicates the existence of microenvironments with a pH of at least 10 within the soil matrix. The existence of ettringite further indicates that some unreacted calcium oxide was still present, even though the amount was probably small. In conclusion, this study shows that the relatively high application rate of FBC materials (112 Mg ha^{-1}) had a long-lasting effect on the soil environment and soil mineralogy.

Future studies on mineralogical effects of coal combustion byproducts in soil need to focus not just on the effects of the "fresh" byproducts but also on the effects of materials that have been exposed to the soil environment for long periods of time. Such studies will provide information on the eventual fate of trace elements included in these byproducts and on long-term changes in soil chemistry and soil mineralogy.

Nonagricultural Disposal of Coal Combustion Byproducts

Coal combustion byproducts are generally regulated by individual states under solid waste regulations. These regulations vary greatly from state to state, ranging from very stringent to nonrestrictive for on-site disposal. Approximately 80 percent of coal combustion byproducts are treated, stored, or disposed of by means of land management, and the remaining 20 percent are recycled (U.S. Environmental Protection Agency 1988). Land management involves the use of surface impoundments, landfills, mines, and quarries. Impoundments and landfills are the two most widely used; about 77 percent of coal combustion disposal facilities use one or the other.

The overall cost incurred in the management of coal combustion wastes ranged from $2.20 to $34.14 per Mg in 1988 (U.S. Environmental Protection Agency 1988). This cost is generally rising rapidly as are costs for landfill disposal of other wastes. This wide range in cost exists because costs are dependent on the type and size of the waste facility and the characteristics of the waste. Generally, fly ash is more costly to manage than bottom ash or FGD wastes. Since environmentally sound recovery and recycling techniques tend to collect a higher percentage of fly ash, these techniques are likely to significantly increase costs incurred by the industry in dealing with waste streams.

The percentage of end products recovered or recycled after coal combustion varies with the particular end product. Coal ash use increased from 18 percent for the period between 1970 and 1980 to 27 percent in 1985 (U.S. Environmental Protection Agency 1988). However, the current expectation is that this percentage will not soon increase. Less than 1 percent of all FGD products were recovered and used in 1985. The percentage of FGD products recovered should increase as the industry develops more efficient recovery and use processes. A summary of some of the nonagricultural uses of coal combustion byproducts is shown in table 27.

Fly ash and bottom ash exhibit pozzolanic properties whereby the dried material forms a hard cementlike material. Carefully selected ashes are used as poz-

Table 27. Current nonagricultural uses of various coal combustion byproducts

Byproduct	Recovery use	Percent
Bottom ash	Blasting grit, road and construction fill, roofing granules	33
Fly ash	Concrete admixture, cement additives, grouting, road and construction fill, stabilization of hazardous wastes, clay liner additive, magnetite production, asphalt amendment	17
FGD products	Sulfuric acid, sulfur, other sulfur products (currently limited in scope), gypsum	<1
FBC products	Cementation of hazardous wastes, cement additive	?

zolans in the manufacture of cement. However, high concentrations of sulfates or nitrates reduces desirability of these pozzolans for this purpose. Fly ash from FGD systems, however, is unsuitable for many construction-related uses because this ash generally possesses substandard pozzolanic properties.

The byproducts from some FGD processes such as dry scrubbing are used to produce gypsum, which can be used as a replacement for mined gypsum in wallboard production. However, wallboard production can account for only a few percent of the FGD byproducts expected in response to the Clean Air Act, and gypsum produced from FGD byproducts is currently considered to be of lower quality than mined gypsum. FGD byproducts other than gypsum are used to make S products. Newer technologies are under study to increase the production of S-related chemicals from FGD sludges.

Little information exists on the uses of FBC byproducts (table 27), but the number of FBC plants has increased recently. FBC materials may be useful for construction purposes because of their dry nature (which should make them cheaper to haul), and because of their pozzolanic properties. However, the minerals in FBC byproducts absorb large amounts of water (changing volume) and generate heat, making them difficult to use for engineering purposes. The same holds true for similar FGD byproducts from the LIMB process.

Coal combustion byproducts are also used as an amendment for coal refuse piles and nonagricultural lands, especially for reclaiming mined areas. (Jastrow et al. 1981, Fail 1987, Taylor and Schuman 1988, Stehouwer and Sutton 1992). A review on the use of fly ash in mined land reclamation was written by Haering and Daniels (1991). The extremely acidic nature of mined lands, resulting from oxidation of S and sulfites, often requires basic material additions to bring pH into the range where plants can grow and where trace element availability is controlled. Consequently the use of power plant byproducts, which are generally alkaline, can assist in moderating pH to the desired levels to reduce trace element availability. Ongoing projects are evaluating the co-utilization of coal combustion byproducts and organic amendments such as municipal biosolids in disturbed land reclamation. The biosolids provide a N source for plant establishment and growth. As with coal combustion materials, biosolids addition should be accompanied by the maintenance of a suitable pH to keep trace elements in the desired concentration ranges.

The use of fly ash and other coal combustion byproducts with or without the addition of an organic material may allow revegetation without application of a topsoil cap. Addition of a topsoil cap is generally the major expense in reclamation of disturbed lands. Abandoned ash basins have been successfully revegetated with trees without the need for topsoil (Carlson and Adriano 1991).

Agricultural Effects From the Use Of Coal Combustion Byproducts

Overview

To justify its use, any amendment to the soil and plant system must clearly benefit the environment (soil, water, or air) or the crop. These benefits must exceed the costs and hazards, whether one is applying fertilizer, organic mulch, irrigation, or an industrial byproduct. Potential benefits and hazards from the agricultural use of various coal combustion byproducts are noted below, followed by a review of past and ongoing research performed and results obtained on the effects of coal combustion byproducts on soil chemical, physical, and microbiological parameters and on plant growth. This discussion is limited to studies dealing with the use of coal combustion byproducts on agricultural land.

Potential benefits

There are a number of potential benefits of applying coal combustion byproducts to agricultural soils. These benefits can usually be classified as either chemical or physical. Chemical benefits can be derived by supplying essential plant nutrients for crop production (for example, supplying B to a B-deficient soil) or by modifying the balance or availability of chemicals to create a more favorable medium for plant growth (for example, modifying the soil pH and decreasing Al toxicity).

Physical benefits of applying coal combustion byproducts include increased water infiltration and aggregation of the soil, which can be attained through gypsum applications under certain conditions. As noted, a major constituent of most FGD byproducts and residues from FBC and the LIMB process is gypsum. Since gypsum-containing byproducts are the most likely candidates for agricultural use, a brief discussion of the benefits of applied gypsum is included later in this section.

The benefits of applying combustion byproducts cannot always be labeled as purely chemical or physical. As an example, application of high-gypsum FGD material may increase water availability and crop yield by reducing subsoil chemical restrictions on rooting depth but also by increasing water infiltration into the surface soil (a physical benefit).

Potential hazards

The primary potential hazards from agricultural use of coal combustion byproducts are excessive trace element loadings, which increase food chain metals; high soluble salt loadings, which may reduce initial plant growth; high Na loadings, which reduce water infiltration; sulfite damage to crops; and leaching of toxic substances into the groundwater. Although the potential for these hazards exists, all of them can be avoided by judicious application of selected coal combustion byproducts. For instance, careful limitation of the use of fly ashes known to be enriched with trace elements can control the loading of these elements to the soil and keep their concentrations in the beneficial or benign ranges in terms of leaching and plant uptake.

Many coal combustion byproduct materials are highly alkaline and can reduce plant establishment by initially elevating the soluble salt content of soils. One method to alleviate this potential hazard is to apply coal combustion byproducts to the surface and then plow to incorporate the material essentially as a layer below the germinating seeds (R.B. Clark, personal communication). Sulfite byproducts applied at planting have also reduced rates of establishment and crop growth. In some soils, however, oxidation of sulfite to sulfate may be sufficiently rapid that application of sulfite-bearing byproducts a few months before planting will not harm plant growth.

Plant nutrient deficiencies of P and Mg are secondary potential problems that may result from the use of coal combustion byproducts. Application of FGD or FBC byproducts originating from facilities using a Ca-based sorbent can create an imbalance in the soil Ca:Mg ratio. This imbalance may induce a Mg deficiency. Therefore, before an application is made care must be taken to monitor the Ca:Mg ratio of the material applied and in the soil. Fortunately, Mg deficiency is usually easily corrected by a soil application of magnesium sulfate (Epsom salts). The high level of Ca, Fe, and Al in some coal combustion byproducts can combine with P in the soil to form insoluble complexes. These complexes reduce the availability of P to plants, which may result in an induced P deficiency.

However, there may be situations where the formation of insoluble complexes of Ca, Al, or Fe phosphates may be desirable. For example, one of the limitations for using poultry manure on land in intensive poultry producing states is the potential for P pollution of surface and subsurface water supplies. The potential co-use of FBC ash (high in Ca) and poultry manure is currently being examined (R.F. Korcak, unpublished data). Using the high-Ca FBC material with the poultry litter may lead to the formation of insoluble Ca-P complexes that will reduce potential P pollution problems.

Benefits and problems associated with trace elements

Most reports on the use of coal combustion byproducts in agriculture conclude that the most serious potential hazards stem from B, Se, As, and Mo accumulation in soils and plants. However, coal combustion byproducts can act as a supplementary source of Ca, S, B, Mo, Se, and other trace elements when soil contents are deficient for adequate plant growth. Proper rates for applying ash as a nutrient supplement are often site specific, and more studies of rates at different types of sites needs to be done before coal combustion byproducts can be used on a large scale.

Selenium is not an essential element for higher plant growth, although it has been shown to be a required element for some lower plant species. However, Se is an essential element for animal growth. One problem with Se in animal nutrition is that it is needed only in very low concentrations; slightly higher concentrations cause Se toxicity. Recommended food and feed concentrations to provide adequate animal Se range from 0.1 to 1 mg Se kg^{-1} plant material. Food and feed Se concentrations above 5 mg kg^{-1} can cause animal Se toxicity (Mengel and Kirby 1987). It is estimated that one-third of the forage and grain crops in the United States contain below-optimal levels of Se for animal nutrition (Mengel and Kirkby 1987). Welch et al. (1991) provided maps of soil micronutrient availability in soils across the country. These maps show areas where Se, Cu, and Mo are typically high in crops, areas where they are sufficient, and areas where additions of these elements are needed to optimize crop production and animal health.

Coal combustion byproducts can be used in agriculture to supplement crops and soils low in Se, Mo, Cu, Zn, or B (Page et al. 1979, Adriano et al. 1980, Aitken et al. 1984, El-Mogazi et al. 1988, Brieger et al. 1992, Environmental Management Services, 1992). The application of byproducts should be based on crop needs and current soil levels of the particular nutrient. An example showing the amounts of major elements supplied per Mg of bottom ash is shown in table 28. In some ashes B and Se levels are high enough to limit the rate of application of those ashes (Ransome and Dowdy 1987). Additional studies are needed to define mineralogy, solubilities, uptake rates, and plant responses needed to calculate optimum soil application rates for ash.

Soluble salts

The other major concern, besides trace elements, with the agricultural use of coal combustion byproducts is the high soluble salt content of many materials. At high application rates, salt injury can occur to germinating seeds or established plants. The problem of high soluble salts can be alleviated in a number of ways. As noted earlier, surface application of coal combustion byproducts followed by plowing allows seeds to germinate without contacting the high salt zone. A similar technique was used by Jacobs et al. (1991); they banded ash into the soil at a 45° angle to the surface. These two methods isolate the applied material from initial root contact. Most application methods homogenize the applied ash into the surface soil and maximize seed contact. Additionally, the timing of application can have a significant impact on avoiding initial problems related to soluble salts.

Another method to avoid soluble salt problems would be to use weathered or stockpiled material from which a substantial portion of the soluble salts has been removed by percolation and from which some of the oxides and hydroxides have been stabilized by carbonation from air. Weathered versus fresh fly ash was compared in field trials with maize (*Zea mays* L.) (Martens and Beahm 1976). Weathered ash could be used at rates up to 131 Mg ha^{-1}, while salt-related problems occurred at 87.2 Mg ha^{-1} when fresh ash was used. Also of interest was a decrease in the incidence of boron toxicity with weathered ash. As previously noted (table 25), a relatively high percentage of the boron in ash is soluble. Therefore, lower amounts of water-soluble boron will be applied to soils when weathered ash is used. The use of weathered materials also decreases the dust hazard associated with applying fresh dry coal combustion byproducts because bonding and recrystalization during moist weathering reduces the proportion of small particles.

A third method to reduce the potential for soluble salt problems has been the successful use of FBC residues as a soil "cap," wherein a thick (5-cm) layer of FBC residue is applied to the surface and not plowed or

Table 28. Amounts of major elements supplied by a metric ton (Mg) of bottom ash

Compound	Cation	% Oxide by weight	Conversion factor	% Cation	kg element Mg^{-1}
Al_2O_3	Al	13.40	0.53	7.09	70.8
CaO	Ca	6.80	0.71	4.86	48.5
K_2O	K	2.10	0.83	1.74	17.4
MgO	Mg	0.74	0.60	0.45	4.5
SiO_3	Si	60.10	0.47	28.29	280.3
TiO_2	Ti	2.50	0.60	1.50	1.0
Fe_2O_3	Fe	14.40	0.70	10.07	100.5

Source: Bryant and Lacewell (1992).

mixed with the soil (Korcak 1988). Used with horticultural crops, this method provides either sufficient soil mass for the roots to avoid contact with the initial flush of soluble salt or reduces this initial flush to levels that can be tolerated by crops. The cap of coal combustion byproduct remains porous, thus allowing water to infiltrate. An associated benefit of the cap method is that the cap acts as a one-way valve, allowing water to infiltrate but decreasing evaporation from the surface (the large pores let water in, but the cap provides little air exchange).

Effect on soil chemical properties

Because of the alkaline nature of many coal combustion byproducts, a number of studies have examined their effect on modifying soil chemistry, primarily pH. The basic property of coal combustion byproducts measured to quantify the effect on soil pH is the $CaCO_3$ equivalence of the materials. The neutralizing effect of pure $CaCO_3$ is 100 percent and that of coal combustion byproducts usually ranges from 20 to 60 percent. Therefore, if a coal combustion byproduct has a $CaCO_3$ equivalence of 50 percent, twice as much coal combustion byproduct as $CaCO_3$ is needed to neutralize the same amount of soil acidity.

Successful modification of soil pH has been demonstrated with a wide range of coal combustion byproducts. Agricultural applications in most situations will probably be based on soil pH modification.

The FBC residues and oxidized FGD materials also contain significant amounts of gypsum and/or its anhydride. The potential benefits derived from gypsum applications in certain soils make those coal combustion byproduct materials enriched with gypsum strong candidates for agricultural use.

A majority of the S currently being deposited in FGD processes is in the form of Ca-sulfites. Seedlings of some crop species grown in the presence of significant amounts of sulfites are not benefitted as they are with sulfates, and actual growth reductions have been observed (R.B. Clark, personal communication). Increased oxidation in the FGD process can result in the production of sulfates rather than sulfites. However, pilot power plant estimates indicate that this will add about \$6 Mg^{-1} to the cost of the sulfur byproduct.

Another avenue would be to wait until natural processes oxidize the sulfites to sulfates. The somewhat gelatinous nature of the sulfite byproduct hampers drying and invasion of the stored byproduct by the air phase. Consequently the rate of oxidation of sulfites stored in large impoundments is generally extremely slow and often practically negligible. On the other hand, there are indications that the rate of oxidation increases rapidly when the sulfites are applied to soils. Whether this is due to better access to oxygen or the inoculation of the sulfite by oxidizing organisms from the soil is not known, but there are indications that the sulfite can oxidize to sulfate within a few weeks. Timing the soil application to allow oxidation to occur prior to plant growth may facilitate conversion of FGD sulfite-bearing materials to sulfates. Properly managed oxidation in the soil might then change the hundreds of millions of Mg of FGD sulfite-bearing materials that are currently being impounded as a hazardous waste into a sulfate resource with significant value.

Effect on soil physical properties

A number of soil physical and related properties have been positively affected by the use of coal combustion byproducts. Improved soil texture (Chang et al. 1989) with concomitant increase in aeration and reduced bulk density result from application of silt-sized coal combustion byproducts. Although increases in water-holding capacity in some soils have been reported from some ash applications, it is unclear whether this effect translates directly into increased available water for plant growth. The existing literature is not clear on this point. However, an interesting study on water relations and ash application was performed by Jacobs et al. (1991). They banded ash into the soil at a 45° angle to the surface. Corn roots were concentrated at the ash band, which was water saturated after rain occurred. Corn yields increased in the ash-banded plots.

The pozzolanic activity of some coal combustion byproducts can be viewed as either a positive or negative attribute. Ash materials that exhibit pozzolanic activity have been shown to reduce soil hydraulic conductivity as well as root growth. These effects can be lessened by using weathered materials or lower application rates. As noted above, banding ash into the soil can avoid some of these problems. No reports were noted on trenching of coal combustion byproducts in agriculture. Studies are underway to examine trenching (15 cm wide by 120 cm deep) of FBC materials alongside tree rows in establishing apple orchards (R.F. Korcak, unpublished data). The purpose of trenching is (1) to prevent lateral root growth to initiate early fruit bearing and (2) to reduce soil volume exploited by the root systems to facilitate management of tree nutrition by fertigation. Additionally, trenching will allow tree roots the alternative of growing into the fringes of the FBC trench to pick up needed Ca, S, and micronutrients or staying away from the relatively high concentrations of these elements if they are deleterious to root growth.

The soil cap technique also has a positive effect on precipitation use efficiency. A cap of byproduct increases sustained infiltration rates, reduces transpiration by weeds (R.F. Korcak, unpublished data), reduces evaporation losses from the soil surface (K.D. Ritchey, personal communication), increases rooting depth in acid soils (Sumner 1990), and eases plant water use efficiency. The resulting improvement in water use efficiency and consequent reduction in water stress on crops would probably be beneficial in many of the crop-producing areas of the United States.

High-Na ash or FGD byproducts may present a potential sodicity hazard, which is characterized by soil dispersion and a reduction in infiltration rates. Application of byproducts high in Na in dry climates, even if mixed in the soil, could create sodicity as the Na is carried to the surface and deposited. This could also be a potential hazard in humid areas, particularly over longer time periods. Consequently, highly sodic materials should generally be identified, and their application to agricultural soils should be avoided.

Overall, coal combustion byproducts application in agricultural soils should be beneficial to soil physical properties if the type of materials are well characterized before use and if highly sodic materials are avoided. In fact, some of the major advantages of coal combustion byproducts may be in the area of enhanced soil water availability for plant growth. This concept, however, needs additional evaluation.

Effect on soil microbiological properties

The microbiology of the soil/plant system as affected by ash application has received the least emphasis by researchers. Most of the research performed to date has examined either soil microbial activity or soil respiration activity (Cervelli et al. 1987, Pichtel and Hayes 1990). Results of these and other studies are generally inconclusive, although a tendency for reduced soil respiration and microbial number following ash application usually occurs. The exact cause of this response has yet to be firmly elucidated.

Amelioration of reduced soil microbial activity may be made by simultaneous addition of an organic amendment such as municipal biosolids (Pichtel and Hayes 1990). The ratio of organic-C to N in soils has a significant effect on soil microbiology. Little or no N is supplied by ash materials, and the C content varies depending upon the particular ash byproduct. Normally, most C in these materials is inorganic and would have little direct effect on microbial activity in any case. The effect of applied ash on the equilibrium soil C:N ratio requires more research. Additionally, the effect of higher C levels in some coal combustion byproducts as well as the effects of co-utilization with an organic source (for example, municipal biosolids, manures, newspaper, and so forth) should be evaluated.

Use of bottom ash

It is worth singling out conventional power plant bottom ash as a potential soil amendment since this material represents one of the more useful coal combustion byproducts for agriculture. These granular materials are generally applied at rates at or near the lime requirement for particular soils. They improve soil structure and hence water infiltration, and increase soil pH.

A management plan for the agricultural use of FBC bottom ash was recently proposed (Sell et al. 1989). The plan incorporates an economic analysis for the agricultural use of these materials. The plan showed that a 62 percent savings is achieved when land spreading is used instead of the conventional landfill disposal. Additional land management plans need to be developed, perhaps on a state-by-state basis, for the agricultural use of bottom ash. These plans should take into consideration soil type, crops grown, and climatic factors.

Use of FBC and FGD residues

Research has been conducted on the agricultural use of FBC byproducts, but no reviews are available. It is difficult to discern whether FBC materials used in many studies were bottom ash or a combination of bottom ash and captured fly ash. The research has generally involved rates equal to the lime requirement of the soil or multiples thereof. Crops studied include corn and peanuts (Terman 1978), peaches (Korcak et al. 1984, Edwards et al. 1985), forages (Stout et al. 1979), and apples (Korcak 1979, 1980, 1982, 1984, 1985; Wrubel et al. 1982). FBC residue was also used as an amendment for acid mine spoils (Sidle et al. 1979).

Use of high application rates of FBC materials is limited by the high alkalinity produced when the material is mixed with the soil (Terman 1978). Mays et al. (1991) incorporated FBC byproducts at rates of 0, 20, 102, and 508 Mg ha^{-1} for corn, soybeans, tall fescue, and alfalfa. Annual application rates up to 20 Mg ha^{-1} or a single application of 102 Mg ha^{-1} had no adverse effects on the yield of any of the crop plants tested. The highest rate led to crop failure primarily due to high soil pH and very high levels of soil Ca and S. The pozzolanic nature of the byproduct created large chunks of the material in the field.

The database on agricultural use of FGD materials is sparse, particularly for unoxidized materials. Compared to FBC materials, unoxidized scrubber sludges will probably require more careful monitoring and lower application rates. Most scrubber sludges contain some fly ash, and fly ash is often added at the end of the waste stream to aid in stabilization of the slurry (Terman 1978). Scrubber sludges must also be kept from reaching an anaerobic stage due to the potential for generating hydrogen sulfide gas (Raiswell and Bottrell 1991).

The FGD sludges oxidized at the coal combustion plant result in material that is high in $CaSO_4 \cdot 2H_2O$ (gypsum); and if they are not oxidized, $CaSO_3$ (calcium sulfite) predominates (Terman 1978). Calcium sulfate is an agriculturally valuable product and has been widely used to supply calcium to peanuts in a soluble form. It also has potential for decreasing subsurface soil acidity and increasing plant rooting depth and drought tolerance. The dissolution of several gypsum-containing FGD materials was compared to phosphogypsum and mined gypsum (Bolan et al. 1991). The FGD materials were 99 percent pure gypsum while the phosphogypsum was 97.5 percent gypsum compared to 82.5 percent gypsum in the mined material. The overriding difference was the higher (12.4 percent) content of $CaCO_3$ in the mined gypsum. All of the FGD materials and the phosphogypsum had higher dissolution rates than the mined gypsum. Dissolution of all samples was three to eight times faster in the presence of soil than in water.

Gissel-Nielsen and Bertelsen (1988) evaluated a number of FGD products in trials with barley. One of these contained 10 percent SO_3^{2-}, 24 percent SO_4^{2-}, 8 percent fly ash, and 0.5 percent NO_3^-. Although not noted, the high amount of sulfate present apparently indicated some oxidation of the material. They noted that plant Se concentrations were increased from 0.05 mg kg^{-1} in the control to 0.18 mg kg^{-1} at the highest application rate (5 g kg^{-1}) of coal-derived FGD. At these concentrations, Se in plants is considered an adequate source for animal nutrition.

Scrubber sludge containing 4.1 g B kg^{-1} was used as a B source to correct a B deficiency on a loamy sand soil (Ransome and Dowdy 1987). Soybean yields were decreased during the first application year by application of 10, 20, and 40 Mg scrubber sludge ha^{-1} because of elevated salt content. Yields were enhanced by scrubber sludge at all rates by the third year. Adequate soil B for soybean growth was achieved with the 20 Mg ha^{-1} application rate. The type of scrubber sludge, that is, whether it was oxidized or not, was not indicated. The authors also expressed a need to determine the location of B that was apparently leached out of the root zone or otherwise inactivated.

There is a continuing need to examine the potential for the use of FGD byproducts in agriculture. The FGD materials currently being produced, particularly dry, oxidized materials, are among those coal combustion byproducts best suited for agricultural use. Research is underway to examine some of the wet FGD byproducts that are high in calcium sulfite (K.D. Ritchey and R.B. Clark, personal communication). These studies and studies involving new byproducts are needed.

Summary

Total production of coal combustion byproducts will reach nearly 154 million Mg annually in the United States by the year 2000. Besides conventional combustion of coal for electric power, which generates bottom and fly ashes, a number of newer byproducts are generated by this industry. These newer byproducts emanate from the need to reduce sulfur emissions. Typically, the desired desulfurization of the flue gases is accomplished by precipitating the sulfur oxides with calcium in the flues, or in the fire boxes with newer combustion technologies such as FBC systems. The diversity of products is further increased by differences in power plant design, operating parameters, sources and types of coal consumed, and, in the case of FGD, the types of reactive reagents used.

The lack of current use of most of these byproducts, their diversity, and potentials for benefitting agriculture create the need for a database to facilitate agricultural use. The majority of the available database information has been geared toward engineering properties of landfilled ash. An agriculturally oriented database will facilitate the selection of those byproducts exhibiting clearly definable benefits to the soil/plant system and will identify components such as boron, selenium, and heavy metals that should be maintained within certain limits in soils.

Potential agricultural benefits from coal combustion byproducts include alleviating soil trace element deficiencies, modifying soil pH, and increasing levels of needed calcium and sulfur, infiltration rates, depth of rooting, and drought tolerance. FGD products and FBC residues that contain appreciable amounts of gypsum appear to have particularly high potentials for improving water use efficiency, product quality, and productivity of soil-crop systems.

The existing literature on agricultural use of coal combustion byproducts needs to be expanded to include data from long-term exposure of these materials in the soil environment. Potential sites for examination exist. Additionally, cooperative research should be initiated with the Department of Energy and private industry to evaluate the potential agricultural use of byproducts resulting from new Clean Air Act technologies as they are developed. These studies should address not only new byproducts but should also incorporate innovative strategies for application and clear documentation of benefits derived.

Documentation of hazards involved and benefits derived, especially from field studies, will be required to reduce present regulatory barriers to agricultural use of coal combustion byproducts. Current inexpensive on-site disposal costs discourage land application of coal combustion byproducts. However, on-site disposal may result in environmentally hazardous concentrations of certain elements in water supplies and the food chain.

Research Needs

A number of broad research areas need to be approached to discern which materials should be used in agriculture and what data needs to be forthcoming to evaluate these materials. The following ideas should be explored:

1. A coal combustion byproduct database should be developed to incorporate agriculturally important parameters since existing engineering databases are not readily applicable. Such a database would assist in the selection of the most appropriate coal combustion byproducts for agricultural use.

2. Cooperative work should be initiated with appropriate agencies and industry to evaluate new byproducts produced by Clean Air Act technology as these products are being developed. This research needs to be conducted in several climatic zones and with different soil types.

3. Chemical data is needed on the fate of coal combustion byproducts in the soil environment. Studies should be initiated at the laboratory and field level to ascertain the fate of potential contaminants. Old agricultural sites previously treated with coal combustion residues should be identified and evaluated.

4. Assay techniques that can be used to identify potentially hazardous byproducts should be developed. Such assays should be plant oriented, simple to perform, and short term. Parameters to be assayed should include soluble salts, trace element phytotoxicities, and excessive alkalinity.

5. Application methods should be evaluated, including surface incorporation, banding, trenching, and surface capping.

6. Coal combustion byproducts containing significant amounts of gypsum should be examined as potential soil amendments. Research should complement reported and ongoing work on mined gypsum and phosphogypsum.

7. Studies should be initiated to examine the potential benefits of mixtures and composts of coal combustion byproducts and other waste streams. In many cases, it appears that these mixtures would enhance the agronomic value of the byproducts.

8. The chemical behavior of sulfite in the soil environment should be evaluated. A better understanding of the fate of sulfite is needed to manage agricultural use of wet scrubber-type FGD byproducts, which contain significant amounts of sulfite.

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Chapter 7

Agricultural Uses of Phosphogypsum, Gypsum, and Other Industrial Byproducts

R.F. Korcak

This chapter provides information about several industrial byproducts and agricultural uses for these byproducts. The focus is on relatively new industrial byproducts for which significant data have accumulated over the past 10 to 15 yr. Also included are some byproducts that have received little if any attention in the past, for example, urban-oriented residues from concrete manufacturing and fines from the production of rock aggregate. Most of these byproducts may present little if any hazard in agriculture. Parr et al. (1983) and U.S. Department of Agriculture (1978) provide information on industrial byproducts not covered in this publication. These two sources include information on byproducts from the following industries: petroleum, pharmaceutical, pulp and paper, soap and detergent, munitions and explosives, pesticides and organic chemicals, textiles, wood preservatives, milling, meat packing, and canneries.

Phosphogypsum

Phosphogypsum is a byproduct of the phosphate fertilizer industry and emanates from the production of phosphoric acid from rock phosphate. Production of phosphogypsum in Florida is estimated to be 27.2 million Mg annually (Hunter 1989). The composition of phosphogypsum varies depending upon the source of rock phosphate and the process for manufacturing phosphoric acid (Mays and Mortvedt 1986). The approximate composition of phosphogypsum is shown in table 29.

Phosphogypsum material normally has an aqueous pH between 4.5 and 5.0. One problem with using phosphogypsum in agriculture is that it contains radioactive radium and radon. In the late 1980's, the agricultural use of phosphogypsum was suspended by the U.S. Environmental Protection Agency when the agency reduced the level of allowable radioactive radium-226 and associated radon in phosphogypsum by a factor of five. This restriction put some of the phosphogypsum into the nonallowable category and therefore made phosphogypsum illegal for agricultural use (U.S. Gypsum Company 1990). Since then the U.S. Environmental Protection Agency (Federal Register 6/3/92) has permitted the controlled use of phosphogypsum in agriculture if radium-226 levels are <10 pCi g^{-1}. This restriction on the maximum radium radioactivity essentially eliminates the use of southern Florida phosphogypsum because its radium-226 levels are commonly in the range of 15 to 25 pCi g^{-1}. The restriction does not impact phosphogypsum from northern Florida or North Carolina, which generally have lower levels of radium-226 (Richardson, personal communication 1992).

Table 29. Composition of phosphogypsum

Major constituents ($g\ kg^{-1}$)*:	
Ca	200–240
P	1–5
S	150–190
F	5–38
Minor constituents ($mg\ kg^{-1}$)*:	
K	100–800
Mg	8–400
Mo	65
Cd	0.23
Radioactive elements:	
^{226}Ra	10–25 pCi g^{-1}†

* Values are averages of those presented in Mays and Mortvedt (1986), Pavan et al. (1987), Lin et al. (1988), Alva and Sumner (1989), Alva et al. (1990), and Sumner (1990).

† Value is from Mays and Mortvedt (1986) and is for southern Florida phosphogypsum only.

The fate of radium-226 in Florida phosphogypsum was investigated by Mays and Mortvedt (1986). They applied phosphogypsum containing 25 pCi g^{-1} ^{226}Ra at rates up to 112 Mg ha^{-1} to the surface of a silt loam soil and grew successive crops of corn (*Zea mays* L.), wheat (*Triticum aestivum* L.), and soybean (*Glycine max* L.). Application of phosphogypsum even at the 112 Mg ha^{-1} rate had no effect on the radioactivity levels in grain of corn, wheat, or soybeans. The 112 Mg ha^{-1} rate was more than 200 times the normal rate of gypsum used for peanut fertilization. Additionally, they noted no increases in grain Cd levels, but at the highest rate they found that corn growth slowed. They speculated that the slower growth was due to an imbalance of Ca and Mg.

Numerous studies have shown that phosphogypsum can alleviate some detrimental effects of subsoil acidity on plant growth (Alva and Sumner 1989, Alva et al. 1990) when surface applied (Caldwell et al. 1990) or subsoiled (McCray et al. 1991). Sumner (1990) concluded that there was essentially no difference between mined gypsum and phosphogypsum regarding correction of subsoil acidity problems.

Lin et al. (1988) used a mesh bag technique to examine the effect of phosphogypsum versus lime on alleviating poor root growth in a Spodosol B_h horizon. Soil in mesh bags was amended with either lime or phosphogypsum, and the bags were implanted around mature orange trees and sampled for periods up to 139 days. The B_h horizon amended with lime had significantly higher root densities than control soils, but root densities in phosphogypsum-amended soil were not significantly different than those of controls. In this study, phosphogypsum did not decrease exchangeable Al compared to control soils.

Pavan et al. (1987) compared the effect of applications of phosphogypsum, lime, calcium chloride, or magnesite (a magnesium-lime material) on apple trees (*Malus domestica* Borkh.) growing in Brazilian soils. Phosphogypsum and lime significantly increased rooting density in the surface of a high-aluminum soil, but this effect extended to a depth of 60 cm with the phosphogypsum application. Phosphogypsum or lime application significantly increased fruit size and yield compared to other treatments, reflecting the enhanced rooting and increased water supply to the trees.

Sumner (1990) compared the effects of surface application of phosphogypsum vs. mechanical mixing of the soil or mechanically mixing lime into a coarse sandy loam soil with an argillic horizon in the subsoil. Peaches exhibited only a slight response to surface phosphogypsum application but responded significantly to both mechanical treatments. The lack of response to gypsum was credited to the greater sensitivity of peach roots to physical rather than chemical barriers in the subsoil.

Gypsum

Gypsum ($CaSO_4 \cdot 2H_2O$) occurs geologically as an evaporite mineral associated with sedimentary deposits. The most important property of gypsum relating to agricultural applications is its solubility. Although gypsum is only slightly soluble in aqueous solution (solubility of 2.5 g L^{-1} in water), it is more soluble than calcite ($CaCO_3$, solubility of 0.15 mg L^{-1} in water) (Finck 1982). The benefits of gypsum on soil chemical and physical properties are as follows:

Physical benefits	Chemical benefits
Increased infiltration	Increased subsoil Ca
Increased aggregation	Decreased subsoil acidity
Decreased Na adsorption	Reduced exchangeable Al
Reduced root impedance	
Reduced restriction of hardpans	

Reviews on the use of gypsum in agriculture have been published (Oster 1982, Shainberg et al. 1989). However, these reviews mostly discuss the effects on agronomic crops rather than the effects on soil properties. The ameliorative effect of increased surface infiltration from surface-applied gypsum on dispersive and sodic soils is well documented (Kemper and Noonan 1970, Shainberg et al. 1989, Roth and Pavan 1991). Applied gypsum decreases the percentage of Na adsorbed on the soil and increases the free electrolyte concentration; these two effects lead to reduced dispersion and increased flocculation and aggregation of soils (U.S. Department of Agriculture 1954). In high-sodium soils with a pH between 8.5 and 10, applied gypsum raises the soluble Ca concentration to levels greater than that of calcite, thereby precipitating calcite. In turn, pH is reduced to 7.5 to 8.0, and calcite and gypsum coexist. The higher soluble Ca concentrations lead to enhanced flocculation of soil colloids (Lindsay 1979).

The effect of surface-applied gypsum (10^4 kg ha^{-1}) on subsoil mechanical impedance was studied by measuring changes in cone penetrometer index for 2.5 yr after application (Radcliffe et al. 1986). A significant reduction in mechanical impedance and increase in root penetration was noted to a depth of 0.55 m within this relatively short time frame. The marked improvement in root penetration resulting from the gypsum appeared to be more directly related to increased Ca supplied by gypsum, which is known to be essential for rapid meristematic root growth. Greater root growth means that more organic matter is being produced in the soil, and this organic matter aids in aggregation and promotes the invasion of beneficial mesofauna such as earthworms. Earthworm burrows facilitate movements of water, oxygen, and carbon dioxide essential to crop growth.

Gypsum has received considerable attention because of its ability to ameliorate subsoil acidity and therefore improve plant rooting (Sumner and Carter 1988). The primary problem associated with subsoil acidity is the high level of phytotoxicity from soluble Al and, to some extent, from soluble Mn. In some cases these high levels of Al and Mn are related to deficiencies of Ca (McCray and Sumner 1990). Gypsum additions can lead to both negative and positive plant responses (Alva et al. 1990), indicating that the chemistry of gypsum in the soil system is not yet completely understood (McCray and Sumner 1990).

Gypsum provides both Ca and S for crop nutrition and has long been used as a Ca source for peanuts (*Arachis hypogaea* L.). Peanuts have a unique Ca requirement during pod development (Alva et al. 1989), depending on peanut type (Gaines et al. 1991), soil Ca status (Alva et al. 1991), and type and form of applied gypsum (Alva et al. 1989). Repeated annual applications of gypsum to peanuts, however, can cause a P deficiency (Sistani and Morrill 1992), since buildup of excess Ca in soil may cause the P to be “tied up” in the form of calcium phosphate.

Gypsum increased Ca levels in cauliflower (*Brassica oleracea botrytis* L.) but had no effect on reducing tip burn, a physiological disorder commonly associated with Ca deficiency (Rosen et al. 1987). The importance of gypsum being more soluble than calcite was shown by Carter and Cutcliffe (1990) in brussels-sprouts (*Brassica oleracea gemmifera*) grown in a low-calcium soil. Gypsum raised tissue Ca levels and marketable yields significantly during the first growing season after application whereas the effects of calcite took longer.

Gypsum was added to blueberry (*Vaccinium* sp.) to study the effect of adding Ca on upland mineral soils without significantly affecting soil pH (Korcak 1992). Although the blueberry is considered to be acid loving, it showed at least a short-term tolerance for increased soil Ca from gypsum. The practicality of using gypsum to enhance root tolerance to high levels of Al in acid upland soils is under further study.

Continuous applications of gypsum or high rates of surface application can cause problems. One of the problems is excessive Ca buildup, which can induce P deficiency and cause excessive leaching of Mg and K from the surface, particularly in sandy soils. Korcak (1988) applied a high-gypsum byproduct between the rows in an orchard for 6 yr and found that foliar Mg levels were becoming deficient. Deficiencies of P, Mg, or K will cause various plant symptoms, including reduced yield. Another problem with continuous or high application rates is increased soil salt content, which can also damage plants and stunt their growth. Sometimes the so-called disadvantages of a material, however, can be used to the grower’s advantage, depending on the crop and the nutrient level of the soil. For example, Alva and Gascho (1991) added gypsum to peanuts and found that the induced leaching of Mg and K was beneficial to the crop’s growth.

The best way to avoid damaging a crop from gypsum applications is to develop standardized soil analyses that will allow for the determination of safe application rates. Sumner (1990) proposed a soil test that is based on the soil’s ability to absorb salt. The test is based on the fact that soils showing a favorable response to gypsum are the ones capable of absorbing the most salt. The test, however, still needs to be calibrated and standardized for a wide range of soil types.

Leather Manufacturing Byproducts

Leather manufacturing generates about 150,000 Mg of dry sludges annually (U.S. Environmental Protection Agency 1976). Basically, three types of byproducts are produced: solid wastes from splitting and trimming hides; sludges from liming, dehairing, pickling, and chrome tanning; and liquid wastes from each step in the operation (Hughes 1988).

The primary agricultural constraint on the use of byproducts from leather manufacturing has been the Cr content of the sludges. Currently, most processing plants are recycling the Cr in wastes before they leave the plant. This recycling segregates high-chromium waste streams from the other waste streams. Chromium has been of concern due to its potential plant toxicity. Trivalent Cr (CrIII or chromic) is present in the sludges but is not toxic to plants and is immobile in the soil system. However, hexavalent Cr (CrVI or chromate) is phytotoxic and mobile. The possible oxidation of trivalent Cr to the mobile form and the potential for phytotoxicity has been the primary focus of most of the research for agricultural use of leather manufacturing wastes (Chaney 1983).

Another potential problem is the salinity of the wastewater generated. This salinity originates from salt present in the hides prior to tanning. Untreated, this wastewater is not suitable for agricultural use. This problem may be circumvented by composting the wastewater and sludge. Under high rainfall regimes, leaching will occur and salinity of the compost will decrease to acceptable levels.

Dewatered tannery sludge has N contents ranging from 2.5 to 5 percent. When Cr has been adequately excluded from these sludges, the optimum application rates for this sludge should be based on N needs of the crops to be grown. This approach should prevent excessive nitrates from being in soil solution during times when they can be leached into groundwater (Stromberg et al. 1984).

Calcium Silicate Slag

Calcium silicate slag is a byproduct of the production of phosphate from apatite ore in an electric furnace. The slag material has a calcium carbonate equivalent of almost 50 percent and contains trace amounts of unrecovered P along with Ca, Mg, and K plus plant micronutrients. However, the Si content of the material has drawn the most research attention.

Sugarcane and rice are known to produce maximum yields when supplemented with Si (Anderson et al. 1992). Calcium silicate slag, containing about 200 g Si kg^{-1}, has been used successfully as a Si source. In addition to yield increases, Raid et al. (1992) reported enhanced resistance of sugarcane to ringspot (a foliar disease) from the use of calcium silicate slag. They hypothesized that the increased uptake of Si into the leaves helped to create a penetration barrier to certain attacking insects that are disease carriers.

The concentration of unrecovered P in calcium silicate slag is usually less than 10 g kg^{-1}. However, if high amounts of slag are applied to agricultural lands, significant amounts of P will be applied. Much concern has arisen over adverse environmental effects from excessive P fertilization in the Florida Everglades, where sugarcane production is high. The fate of P applied from calcium silicate slag has been studied (Anderson et al. 1992). At application rates of up to 20 Mg ha^{-1}, P applied in slag was found to be biologically inactive and not likely to contaminate drainage waters and waterways.

The concern over adverse environmental effects of increased P in waterways is not limited to the Everglades area. The potential of high-calcium industrial byproducts, such as calcium silicate slag, to reduce solubility of P and keep its concentration in biologically desirable ranges needs to be documented. This type of research is also needed for other high-calcium byproducts such as fluidized bed materials and flue gas desulfurization byproducts from coal-burning power plants.

Anderson (1991) found another potential problem with the agricultural use of calcium silicate slag. In some cases following slag application, Mg deficiency can occur and yields can be subsequently reduced. It is unknown whether this is due to a Si/Mg antagonism, a low soil Mg content, or possibly an imbalance of soil Ca and Mg resulting from the high ratio of Ca to Mg in the applied slag.

Incineration Ash

Incineration of municipal wastes is becoming more widespread. As of 1990 there were 70 municipal refuse incinerators operating in the United States and about 250 facilities in the planning stage (Lisk et al. 1989). Ash materials from incineration are becoming an urban problem. No reports exist on the potential for agricultural use of incinerator ash. Most of this ash will probably be placed in landfills.

The incineration processes used vary from facility to facility, thereby creating ashes with a range of characteristics (for example, ranging from acidic to alkaline). The variability of the end product is one of the primary reasons for not using this ash for agricultural purposes, particularly because of variability in the content of trace elements (Swahney and Frink 1991). Pressures to apply these ashes to land will increase in the future, and therefore baseline data to identify the benefits and hazards of these materials will be needed.

Concrete Manufacturing Residues

Approximately 1.8 Mg of concrete are produced each year per person in the United States, and about 2 to 4 percent of this amount, 36.3 to 72.6 kg per person, is waste. Concrete waste is generally an urban byproduct. For example, in the Washington-Baltimore corridor it is estimated that about 9,000 Mg of concrete waste is produced annually. A portion of concrete waste originates from the solid materials (aggregates used in the concrete) rinsed from delivery trucks. This material is alkaline and high in calcium silicates.

A preliminary investigation of the solid material from truck rinsing is underway (Korcak, unpublished data). Initial results indicate that the material is not phytotoxic even at rates up to 224 Mg ha^{-1} when applied to the surface. As a result of its alkaline and calcareous nature, this material appears to have potential as a liming agent.

Aggregate Industry Fines

The annual production of aggregate materials (sand, gravel, and crushed stone) in the United States is about 1.8 billion Mg (R. Meininger, personal communication 1993), and about 5 to 10 percent of this total is waste fines. The size of these fines ranges from that of fine sand to that of clay, and the fines are collected in settling ponds. Many types of aggregate materials are produced in the United States, partly because of the large number of production plants (Tepordei 1987) and the number of different rock types that are crushed (Tepordei 1992).

A number of attempts have been and are being made to find uses for aggregate materials. Some of the fines are mixed with an organic material (for example, mixed with municipal biosolids or composted with municipal refuse) to create a manufactured top soil.

When rocks fracture, nutrients within the rock material become more available. Enhanced availability is evidenced by the proliferation of root growth in new fractures in rocks. Plant growth and nutrient availability can be increased by incorporating fines from rock crushing operations into soils. Exudates from roots may have a role in mobilizing nutrients from newly fractured mineral surfaces. Information is needed on how much of this mobilization is caused by water and how much is caused by root exudates. This information will assist in understanding and predicting the benefits and possible hazards of agricultural use of freshly crushed fine materials or fines at depths that have not been demineralized by plant growth or acid leaching.

Wood Ash

The combustion of wood waste for producing steam or electricity creates localized sources of ash. An industry rule-of-thumb is that bark from 1 million board feet of logs will provide 1 ton of ash. This equates to more than 908 Mg of ash generated annually in western Montana (Host and Pfenninger 1978). Nationally, 1.4 to 2.7 million Mg of ash are produced yearly from combusting wood wastes from paper mills and saw mills (Campbell 1990).

Ash composition varies with the source of waste wood. Etiegni and Campbell (1991) reported that lodgepole pine sawdust ash had a pH of about 13 and contained 18 to 26 percent Ca, 6 to 9 percent Mg, 0.4 to 11 percent K, and 1.7 to 2.5 percent P. They studied the effect of temperature on ash composition and found that Ca, Mg, and P in ash increased with increasing temperature while K decreased. Schreiner et al. (1938) noted that some unleached hardwood ashes can contain upwards of 6 percent potash, 2 percent phosphoric acid, and 30 percent lime.

Agricultural use of wood waste has increased and will continue to increase due to stricter regulations on open burning or landfilling and due to increased costs of disposing of these materials in landfills. A number of studies performed on the use of wood ash in agricultural situations have been summarized (Campbell 1990). The primary benefits of wood ash include neutralizing soil acidity (lime potential) and providing a source of K plus other macronutrients and micronutrients. Rates for applying ash to land should be based on the soil lime requirement. Applied at the proper rate, ash should present little risk to the environment.

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Appendix

Conversion Factors

The following information is useful in converting metric system units to English system units:

Mass

g = gram = 0.0022 pounds
kg = kilogram = 1,000 grams = 2.205 pounds
Mg = megagram = 1,000 kilograms = 1 metric tonne = 1.10 U.S. tons
mg = milligram = 0.001 grams

Length

m = meter = 1.094 yards
km = kilometer = 1,000 meters = 0.621 miles
cm = centimeter = 0.01 meters = 0.394 inches
mm = millimeter = 0.001 meters
µm = micrometer = 0.000001 meters
nm = nanometer = 0.000000001 meters

Volume

L = liter = 1.057 quarts = 0.265 gallons
kL = kiloliter = 1,000 liters = 265 gallons
µL = microliter = 0.000001 liters
m^3 = cubic meter = 35.3 cubic feet

Area

ha = hectare = 2.47 acres

Application Rate or Crop Yield

kg ha^{-1} = kilograms per hectare = 0.893 pounds per acre
Mg ha^{-1} = megagrams per hectare = 893 pounds per acre = 0.446 U.S. tons per acre
bushels ha^{-1} = bushels per hectare = 0.405 bushels per acre
m^3 ha^{-1} = cubic meters per hectare = 14.3 cubic feet per acre

Concentration

g L^{-1} = grams per liter = parts per thousand
mg L^{-1} = milligrams per liter = parts per million
µL L^{-1} = microliters per liter = parts per million
kg m^{-3} = kilograms per cubic meter = 0.062 pounds per cubic foot
g kg^{-1} = grams per kilogram = percent divided by ten
mg kg^{-1} = milligrams per kilogram = parts per million

Radioactivity

pCi g^{-1} = picocurie per gram = 2.22 radioactive disintegrations per minute per gram of material