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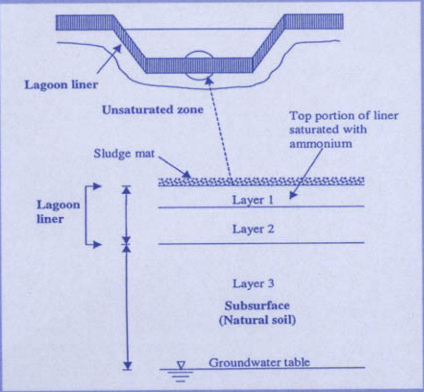
Animal Waste Containment in Lagoons



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Animal Waste Containment in Lagoons

Prepared by
The Animal Waste Containment Committee of
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of the American Society of Civil Engineers

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Chapter 1

INTRODUCTION

Frank Humenik

Historically, animal waste was considered a valuable fertilizer and thus land was applied for sustainable on-farm production of crops and animals. However, with the shift to large production units that relied on grain imports and that had insufficient land for waste application at fertilizer rates, treatment processes that would increase flexibility and reduce amounts of nitrogen and phosphorus became necessary.

1.1 OXIDATION POND

Oxidation ponds, which had been commonly used for municipal wastewater treatment in rural areas, were considered for the treatment of swine waste. Oxidation ponds are shallow basins excavated in the ground for the purpose of storing and treating settled sewage or other wastewater under climatic conditions that favor the growth of algae, namely warmth and sunshine. Bacterial decomposition of the waste produces carbon dioxide, which stimulates growth of algae, which take up ammonia and other plant growth substances. Oxygen is produced during photosynthetic conditions, but during dark periods oxygen is consumed, resulting in a diurnal effect for dissolved oxygen in oxidation ponds. The driving force for this type of treatment is photosynthesis supported by a symbiosis between saprophytic bacteria and algae.

If oxidation ponds are made 3 to 4 ft deep, not overloaded with oxygen-demanding wastes, and hold the sewage for 3 to 4 weeks, a substantial reduction in coliform organisms and biochemical oxygen demand is obtained. The effluent is high in dissolved oxygen, often supersaturated during the daytime. The pond waters are so rich in algal cells that efforts have been made to harvest these cells as a source of protein for animal feed. Five-day biochemical oxygen demand loadings of about 45 lb per acre are generally employed. Oxidation ponds must be cleaned after an interval of several years, and weeds must be kept under control.

McKinney (1970) stated that the use of oxidation ponds for swine waste required far too much land area because of their shallow depth and that, for such large ponds, mixing would be a problem; for example, a 0.40-ha oxidation pond 1.2 m deep would be required for a 200-head hog operation. Most of the early livestock lagoons were expected to function as aerobic lagoons, but usually they became anaerobic because loading was too heavy.

1.2 UNAERATED AEROBIC LAGOONS

Early recommendations given by Ricketts (1960) and Jedele and Hasen (1960) called for about 1.4 m^2 of surface area per hog and a depth of 0.9 to 1.5 m. These lagoons were expected to function as municipal unaerated aerobic lagoons, but because of overloading, they did not.

Clark (1964, 1965) concluded that lagoons for livestock waste must be aerobic if they are to be satisfactory for the producer and the community, recommending approximately 1 ha of lagoon for each 680 hogs at latitude 40 degrees N. However, Muehling (1969) reported that a lagoon would not remain aerobic at this loading rate without some method of restricting solids loading. Dale (1969) recommended the volume of an aerobic lagoon for the midwestern states as 490 ft^3 per 100-lb hog on feed. This was determined using a 5-day biochemical oxygen demand loading of 50 kg/ha of lagoon. Dale also concluded that aerobic lagoons must be cleaned after several years and weeds must be kept under control.

Unfortunately, the use of unaerated aerobic lagoons for animal waste is generally not feasible because of the excessive surface area requirements. Hart and Turner (1965) concluded that unaerated aerobic lagoons cannot be practically used because the high concentration of organics would require excessive amounts of dilution water to develop naturally aerobic conditions.

Humenik and Overcash (1976) reported that design criteria for unaerated aerobic lagoons of 37 to 74 m^3 per 45-kg hog did not result in dissolved oxygen at depths greater than 10 cm below the surface of monitored lagoons.

1.3 ANAEROBIC LAGOONS

Loehr (1968) stated that, in general, the main purpose of anaerobic lagoons is the removal, destruction, and stabilization of organic matter, not water purification. He concluded that anaerobic lagoons offer considerable potential for handling and treating concentrated animal waste, and that anaerobic lagoons are practical only when used prior to further treatment and disposal.

Anaerobic decomposition has become one of the most common treatment alternatives for the swine producer. The anaerobic process can produce undesirable odors, but a major benefit is the flexibility to easily degrade organic material. In the initial stage, the waste materials are biologically converted to simpler organic end-products including aldehydes and alcohol but principally fatty organic acids. During the second stage—methane formation—the organic acids are converted by the methane-forming organisms to gaseous end-products, principally methane and carbon dioxide. Waste stabilization or organic removal is directly proportional to the methane produced.

Anaerobic lagoons began to be used extensively for swine wastewater treatment in the mild climate of the southeastern region of the United States around 1965. Discharge from these lagoons was no longer allowed as a result of federal legislation passed in 1972. Therefore, lagoon systems were modified to meet this no-discharge criteria, which consisted of a lagoon and land irrigation of lagoon liquid at fertilizer rates for nitrogen. These lagoons provided up to 90% removal of nitrogen and storage so that wastewater could be applied conveniently to less acreage than was required for the defecated waste. These systems became very popular in areas where the waste was not utilized for its full fertilizer value by land application because they provided a very labor-free, convenient, and cost-effective manner of managing waste on limited land areas. The benefits of this system are verified by the fact that more than 4,000 swine waste lagoons were in operation in North Carolina in 1999.

Early lagoon loading recommendations for anaerobic lagoons were made on the basis of surface area per hog. When it was realized that these lagoons would not operate as aerobic lagoons, the emphasis shifted from surface area for oxygen transfer to total volume for anaerobic treatment. Anaerobic lagoons were made as deep as possible to achieve this design volume in a manner that minimized surface area for the collection of rainfall, which increases the volume of wastewater to be managed.

Eby (1964) divided the United States into regional areas and based recommendations on municipal lagoons, which varied from 3,750 pigs/ha of lagoon in the northern United States to 15,000 pigs/ha of lagoon in the southern United States. The 1970 interim specifications for animal waste lagoons prepared by the Soil Conservation Service (SCS 1970) were based on work by Eby but incorporated a 25% safety factor. Loading rates are specified as minimum surface area per animal for four different climatic zones within the United States. For the Southeast (Zone B), the loading rate is 1,680 kg of 5-day biological oxygen demand (BOD_5)/surface ha. If the minimum specified depth of 1.8 m is utilized, this is equivalent to 2.2 m^3 of lagoon volume per 45-kg hog.

Loading recommendations for anaerobic lagoons differ widely within the United States, depending on the loading parameter used, geographical conditions, and reference. Recommendations range from about 1 to 24 m^3 of lagoon volume per 45-kg hog (Humenik and Overcash 1976).

Humenik and Overcash (1976) conducted pilot-scale field evaluations of lagoons at different loading rates to determine the relationship between loading rate and treatment performance, including odor comparisons. It was concluded that odor resulted from lagoons loaded according to existing criteria, and thus lagoon loading recommendations would have to be increased. They concluded that, based on periodic field observations and the result of odor panel rankings, there was a discernible odor threshold at approximately 9.2 to 18.4 m^3 per 45-kg. Below this threshold, odor was not manure-like, nor was an odor always detectable. For lagoons with less than 9.2 m^3 per 45-kg hog, odor was not always detectable; but when found, it was characteristic of swine manure and hence deemed offensive. These results led to lagoon loading rates for North Carolina of 1 to 3 ft^3 per 100-lb hog, with 2 ft^3 being the general recommendation and 3 ft^3 per 100-lb hog being recommended where more positive odor control was desired. These rates are still being recommended for North Carolina by Barker (1983) and by Natural Resources Conservation Service Standard 359 for North Carolina (1989).

1.4 LAGOON PROBLEMS

Recently, lagoons have received severe public criticism because of water quality, odor, nuisance, and public health concerns.

1.4.1 Overflow

Lagoons have discharged due to structural failures or overflows, polluting many miles of receiving waters. Increased emphasis is being placed on maintaining lagoon embankment integrity by regular checking of embankments at the bottom of toe slopes. Regulations have been passed to ensure that sufficient freeboard is maintained to contain up to the 24-h, 25-yr storm. In North Carolina, storage is required for two 24-h, 25-yr storms to minimize overflow during heavy rainfall conditions and hurricanes. Regulations also have been passed in North Carolina that require the installation of lagoon liquid level indicators with the provision that, if sufficient storage is not maintained, anyone who observes this condition must report it to the regulatory agency.

The 1972 Clean Water Act established a comprehensive program for protecting and restoring our nation's waters. Among its core provisions, the Clean Water Act prohibits the discharge of pollutants from a point source to waters of the United States except as authorized by a National Pollutant Discharge Elimination System (NPDES) permit. The Clean Water

Act also provides for the development of technology-based and water-quality–based effluent limitations that are implemented through NPDES permits to control discharge of pollutants. Discharges could be allowed under NPDES permits or under upset and bypass conditions. For the feedlot industry, technologies were to be implemented that would prohibit discharge from anything less than a 24-h, 25-yr design storm.

Beginning in about 1998, the National Resource Conservation Service began installing emergency overflows in newly constructed animal waste lagoons in North Carolina. These overflows allow discharge, preventing lagoon rupture when the design storm is exceeded. In 1999, North Carolina enacted procedures for emergency management irrigation of animal waste lagoon liquid on land. This was done because required rainfall storage levels were exceeded as a result of chronic and catastrophic rainfall, including Hurricane Floyd. These provisions have reduced lagoon overflows. In North Carolina, only about four active lagoons and one inactive lagoon overflowed during the flooding that occurred as a result of Hurricane Floyd, which was assessed to represent about a 150-yr storm.

1.4.2 Leakage

Many studies have verified that lagoons leak, especially those constructed before maximum leakage criteria were developed. High levels of nitrate have been recorded around lagoons that are improperly sealed, but it is very difficult to judge the risk-benefits of these increased nitrate levels and to compare them with nitrate levels resulting from crop fertilization. In general, for lagoons that are sealed with clay or material liners and that comply with maximum leakage requirements, groundwater impacts are minimized.

1.4.3 Odor

Lagoon odor is a difficult problem that causes high public sensitivity and emotionalism. There are increasing reports that odor from swine waste lagoons aggravate existing allergies and cause public health problems. Lagoon odor can be reduced by reducing loading rate, by covering lagoons with materials that are becoming cheaper and more durable, and by providing aeration. For odor control, it has been recommended that aeration be provided to satisfy the 5-day biochemical oxygen demand to one-third that value (Muehling 1969) and to about one-fourth of the chemical oxygen demand (COD) (Humenik et al. 1975).

Aeration also can increase the volatilization of ammonia if the nitrogen is not rapidly converted to nitrate. Ammonia volatilization is a major mechanism for reducing nitrogen in both unaerated and some aerated lagoons. Ammonia volatilization has become a recognized issue only recently. It is still relatively unknown how much ammonia volatilizes from aerated and unaerated lagoons because of the difficulty in establishing a standard technique for measuring ammonia volatilization.

Several papers have been published on dinitrogen gas emissions from anaerobic swine waste lagoons (Harper et al. 2000; Harper and Sharpe 1998). Denitrification technologies that convert wastewater nitrogen to harmless dinitrogen gas provide oxidation of nitrogen to nitrite followed by an anaerobic unit with sufficient nutrients for conversion of nitrate to dinitrogen gas. This denitrification process can result in various nitrogen oxides which impact air quality. The presence of dinitrogen gas in anaerobic lagoons requires additional work to determine nitrogen loss mechanisms from aerated and unaerated lagoons and to determine its impact on air quality, environmental quality, and public health.

1.5 LAGOON PHASE-OUT

Lagoon problems with leakage, overflow, offensive odors, ammonia volatilization and potential public health concerns are severely limiting the continued permitting and use of lagoon systems. Some states are considering legislation that would require immediate covering of all unaerated animal waste lagoons. North Carolina is in the process of developing a lagoon phase-out program, which is described in the publication "Framework for the Conversion of Anaerobic Swine Waste Lagoons and Spray Fields" (North Carolina Department of Environment and Natural Resources 2000). The vision set forth in this framework is "to move North Carolina toward a day when swine production produces no ill public health or environmental impacts so that it becomes a sustainable part of North Carolina's Economy." According to the Background section of the publication, "Swine production in North Carolina has mushroomed over the last decade. Despite a decreasing number of swine facilities, the number of hogs has increased threefold to 10 million. The change from small independent farms with small herds to large industrial style operations creates a need for new swine waste management systems. Swine waste in North Carolina is currently managed predominantly through anaerobic lagoons and spray fields. This technology meets current federal and state standards and has been an accepted and encouraged practice." There were about 4,500 active and 1,700 inactive swine waste lagoons in North Carolina in 2002.

The plan sets forth the following for anaerobic lagoon and spray field conversion:

Given the risk of swine production to ground and surface waters, air, and public health, the [governor] Hunt's administration proposes a widespread conversion of swine waste lagoons and spray fields to new technologies. All swine facilities will be rated to determine their risk to public health and the environment. Facilities that fail to protect public health or the environment will be required to convert to new technologies or close out their lagoon and/or spray field systems. Facilities, which demonstrate through monitoring and other criteria that they are operating in an environmentally sound manner, will have the option of keeping their current waste management systems. Those that choose not to convert must meet stringent new protection standards according to their risk while converted facilities will have lower regulatory and monitoring requirements. This plan is the best way to insure the public and the environment are protected while at the same time enabling swine production to remain a viable part of North Carolina's economy.

The conversion plan divides lagoons and spray fields into three main categories: inactive, active, and future.

Inactive

Inactive High Risk

Lagoons will be required to close out immediately. Landowner will likely be required to empty contents of the lagoon.

Inactive Low Risk

Lagoon will be required to be closed out by specific time period. Landowner may or may not need to empty contents of the lagoon.

Active

Active With Problems

Producers with significant unresolved problems regarding odor, nitrogen emissions, surface water, groundwater, or public health must install new technology.

Active With Risk But No Current Problems

Producers will have the choice to convert to new technologies. Facilities identified with significant risk that retain their current systems will have to demonstrate an excellent compliance record and meet additional regulatory requirements. Depending upon the risks, the requirements would include surface and/or groundwater monitoring systems for pathogens and nitrates, individual permits, best management practices for odor, financial bonding and close out plan requirements, and other strict standards.

Active With No Significant Risk

Producers will not have to install new monitoring systems so long as the annual inspection determines that the facility poses no significant risk to public health or the environment.

Future

New Facilities

Producers will be required to use new technologies, based on principles set forth in the 1997 General Assembly HB1480, to eliminate discharges to surface and groundwater and substantially eliminate odors, atmospheric deposition, and nutrient and heavy metal contamination of soil. Producers that use basins as an auxiliary part of a new technology should be required to use synthetic liners.

This framework also notes that "technologies that rely on the evaporation of nitrogen into the air should no longer be allowed if nutrient sensitive waters are impacted. Requirements for existing and new technology should also apply to swine houses, as they tend to be the predominant sources of odor. Additionally, research on atmospheric deposition, odor, ground water, and other aspects of swine housing will continue."

Criteria that inactive lagoons be emptied and bottom sludge be totally removed for terminal land management in a manner that does not impact soil, air, or water quality are being used nationwide. Procedures are being investigated for natural remediation of animal waste lagoons and phytoremediation technologies for an economically and environmentally responsible technique for decommissioning anaerobic swine waste lagoons.

1.6 CONCLUSIONS

Animal waste treatment systems that result in a liquid stream will require some type of storage basin to hold effluent between terminal land application events. Effluent storage basins for advanced waste treatment systems should not cause the public criticism or emotionalism that has resulted from poorly designed and operated anaerobic lagoon/spray field waste management systems. It is noteworthy that many technical experts state that, if anaerobic lagoons are properly designed, constructed, operated, and maintained, they provide cost-effective treatment of animal waste. Furthermore, if lagoon problems such as overflow, leakage, odor, and ammonia volatilization can be solved, there would be a science-based justification for continued use of lagoons as treatment/storage devices prior to land application at recommended agronomic rates.

Although the current emphasis is on determining the risk-benefits of lagoons and the development of treatment systems that provide alternatives to anaerobic lagoon/spray irrigation systems, the long-term need is the conservation of valuable waste constituents and sustainable animal agriculture. Nitrogen and phosphorus management may become the most important long-term need and opportunity so that these potentially valuable components of animal waste are conserved. The conservation and utilization of valuable animal waste constituents by processing to alternative value-added products must receive increased emphasis. More effective feed-conversion efficiency—through nutritional and other management strategies to reduce waste volume, odor, and constituent amounts—shows great potential for source reduction. The long-term goal must be the conservation and utilization of valuable waste constituents to support sustainable animal agricultural and protect environmental quality.

Chapter 2

FATE OF NITROGEN COMPOUNDS IN ANIMAL WASTE LAGOONS

Alok Bhandari and Kang Xia

Each year, farm animals consume approximately 8 million tons of nitrogen in their feed and excrete about 7 million tons of nitrogen as waste. Most confined animal operations (CAOs) utilize lagoons for the containment and treatment of animal wastes. The occurrence and production of nitrate (NO_3^-) in these lagoons is a major cause of concern. The microbial decomposition of organic wastes in animal waste lagoons results in the production of ammonia (NH_3) or ammonium (NH_4^+), which under aerobic conditions can be converted to NO_3^-. Contamination of potable groundwater with NO_3^- poses a serious risk to human health. Nitrate is not particularly toxic itself, but it can be converted to the highly toxic nitrite (NO_2^-) by human saliva or intestinal bacteria in infants. Nitrite can prevent oxygen from binding to hemoglobin, the oxygen-carrying protein in the bloodstream, resulting in cyanosis or oxygen starvation of tissues. This causes a bluish discoloration in infants, resulting in a condition known as methemoglobinemia, or "blue baby" syndrome, which can be fatal in extreme cases.

High NO_3^- levels in groundwater also have been known to cause adverse health effects in ruminants such as sheep and cattle. Due to health risks to infants, the U.S. Environmental Protection Agency (USEPA) enforces a drinking water maximum contaminant level (MCL) of 10 mg/L for NO_3^--N (or 45 mg/L NO_3^-) established by the Safe Drinking Water Act. In adults, ingestion of 500 to 1,000 mg of NO_3^- can result in mild irritations of the gastrointestinal tract and the urinary bladder. More serious problems, such as nausea, vomiting, and hemorrhagic diarrhea, may occur from uptake of 1 to 2 g of nitrate (Strauch 1987). Some recent estimates of human exposure to nitrates in drinking water are presented in Table 2-1.

An understanding of factors affecting NH_3 and NO_3^- transformations in animal waste lagoons is also important because these chemicals are considered limiting nutrients in surface water bodies. Most rivers and

TABLE 2-1. Estimates of Human Exposure to Nitrates in Drinking Water

Nearly 85 million people in the United States drink water from community water system wells that contain nitrate.

These people are exposed to a median nitrate concentration of approximately 0.63 mg/L.

Nearly 3 million people drink water from wells containing nitrate at a concentration of at least 10 mg/L, the drinking water MCL for nitrate.

Approximately 43,500 of the people exposed to nitrate concentration higher than the MCL are expected to be infants at possible risk of developing methemoglobinemia.

Source: USEPA 1992.

lakes are naturally deficient in nitrogen and phosphorus. This nutrient deficiency prevents the growth of unwanted plants and algae in surface waters that may otherwise result in the production of color and odor in the water. Nitrates entering rivers, lakes, and reservoirs are available for plant growth and are used by algae for protein synthesis. This process of nutrient enrichment of surface water bodies is known as eutrophication. The eventual microbial decay of dead plant material in eutrophic surface waters results in excessive consumption of dissolved oxygen. The resulting low levels of dissolved oxygen in the water may be responsible for fish kills and anaerobic conditions that cause discoloration and odor in water, or dissolution of toxic metals from sediments into the water column.

The ultimate environmental fate of nitrogen in waste lagoons may be affected by a variety of physical, chemical, and biological processes, including ammonification, nitrification, denitrification, adsorption, volatilization, leaching, and fixation. This chapter describes these processes and their effects on nitrogen fate and transport in animal waste lagoons.

2.1 NITROGEN IN ANIMAL WASTE LAGOONS

The rapid growth in animal production facilities has emphasized the need for proper management of animal wastes to avoid potential pollution problems. Nitrogen-containing waste from confined animal operations can enter water bodies through surface runoff or seepage into groundwater. It is important, therefore, that cost-effective methods for nitrogen control be compatible with livestock production operations, convenient to manage, economically feasible, and able to perform satisfactorily under varying environmental conditions.

Most onsite animal waste containment facilities are operated as anaerobic lagoons. These lagoons are maintained with minimal surface reaeration and photosynthesis, resulting in anaerobic conditions. The purpose

TABLE 2-2. TKN and NH_3-N Concentrations in Various Swine, Poultry, and Cattle Waste Lagoons

	Swine1	Swine2	Swine3	Swine4	Swine5	Poultry1	Poultry2	Cattle
TKN	763[a]	827	642	391	450	489	800	274
(mg/L)	(108)[b]	(93)	(205)	(102)	(65)	(94)	(134)	(83)
NH_3-N	701	778	534	336	393	403	737	162
(mg/L)	(107)	(87)	(194)	(99)	(67)	(93)	(136)	(77)

Source: Adapted from Westerman et al. 1990.

Notes:

[a] Mean of monthly samples over a 4-yr period.

[b] Standard deviation.

of containment includes transformation and stabilization of the high-strength organic waste stored in the lagoon. Anaerobic transformations in waste lagoons are mediated by a variety of bacteria categorized mainly into two groups: (1) acidogens, the acid formers, and (2) methanogens, the methane formers. For the anaerobic treatment system to perform well, these two groups of bacteria operate symbiotically to ensure that the organic carbon is converted to methane. A gradual buildup of solids occurs over time, necessitating periodic removal of solids. The rate of solids buildup is usually a function of the amount of waste flowing into the lagoon, the characteristics of the raw waste, and the rate of solids stabilization. The biodegradable portion of the solids undergoes anaerobic digestion with the release of gaseous products.

Nitrogen in wastewaters associated with animal production facilities exists primarily as proteinaceous material and urea. Proteins, which are made up of amino acids, are readily degraded by bacteria, resulting in the production of NH_3 in animal waste lagoons. Some typical values of Total Kjeldahl Nitrogen (TKN), a measure of ammonia and organic nitrogen, and NO_3^- observed in several animal and poultry waste lagoons over a 4-yr period are listed in Table 2-2 (Westerman et al. 1990). Suggested guideline TKN values for some common animal wastes are given in Table 2-3.

TABLE 2-3. Suggested TKN Values for Feedlot Wastes

	Units	Pork	Laying Hens	Feedlot Beef	Feedlot Sheep	Dairy Cattle
Total	% TS	5.6	5.9	7.8	4.0	4.0
Kjeldahl	kg/ton TWW	7.6	15	12	12	3.8
Nitrogen	kg/APU-day	0.039	0.099	0.055	0.043	0.036

Source: Adapted from Taiganides 1987.

TS = Total solids content; TWW = Total wet weight; APU-day = Animal production unit-day.

The predominant fraction of organic nitrogen in a waste lagoon is initially associated with the particulate matter in the untreated waste. Wastes entering a lagoon are exposed to a complex ecosystem, and the physical, chemical, and biochemical activities in this system directly influence the form and ultimate fate of nitrogen. Nitrogen compounds in the waste may undergo biological transformations or oxidation-reduction cycles during the long detention times typical of animal waste lagoons. Incorporation of nitrogen into sludge occurs as a result of direct sedimentation of particulate matter and through nitrogen uptake by plants, animals, and bacteria and their subsequent settling on the lagoon bottom. The bottom sludges usually have very high nitrogen contents.

2.2 TRANSFORMATIONS AND REMOVAL

Nitrogen in fresh animal excreta is primarily in the organic form. As shown in Figure 2-1, the ultimate fate of nitrogen in waste lagoons is influenced by a variety of processes: volatilization; uptake by algae and plants; nitrification; denitrification; sludge deposition; adsorption onto

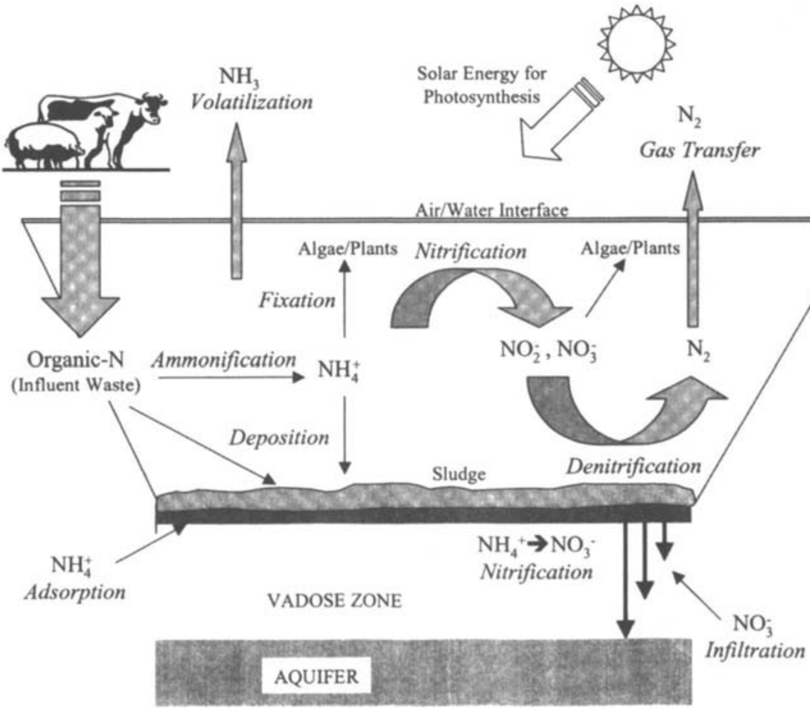


FIGURE 2-1. Fate of Nitrogen in Animal Waste Lagoons.

TABLE 2-4. Selected Characteristics and Nitrogen Removal in the Case of Four Facultative Wastewater Lagoons

Location	Detention Time (days)	Water Temperature (°C)	pH	Total Influent-N (mg/L)	Effluent Nitrite + Nitrate (mg/L)	% N Removal*
Eudora, Kan.	231	14.7	8.4	50.8	0.67	82
Corrine, Utah	42	10	9.4	14.0	0.04	46
Kilmichael, Miss.	214	18.4	8.2	35.9	0.19	80
Peterborough, N.H.	107	11	7.1	17.8	0.19	43

Source: Adapted from Reed 1985.
Note:
*Removal in three cell systems.

sludge solids, bottom soils, and lagoon-liner materials; and subsurface infiltration. The removal of nitrogen from waste lagoons depends on various physicochemical and biological factors, such as pH, temperature, residence time of the waste, mixing of lagoon contents, presence or absence of oxygen, and the microbial ecology of the waste. Table 2-4 lists selected characteristics of four wastewater lagoons across the United States and the extent of nitrogen removal observed in these lagoons.

Biological and chemical processes in the natural environment are capable of transforming nitrogen from one oxidation state to another through electron transfer reactions. The oxidation states of nitrogen in the natural environment range from -3 in the NH_3, NH_4^+, or organic-N forms (most reduced) to $+5$ in the NO_3^- form (most oxidized). Ammonification (or mineralization) converts organic forms of N to NH_3 or NH_4^+, in which the oxidation state of N does not change. The biological oxidation of NH_3, NH_4^+, or organic-N to NO_3^- is known as nitrification. Nitrite (NO_2^-, $+3$ oxidation state) is produced as an intermediate product during the nitrification processes. The biologically mediated reduction of NO_3^- to nitrogen gas (N_2, zero oxidation state) is called *de*nitrification. Nitrite, nitric oxide (NO, $+2$ oxidation state), and nitrous oxide (N_2O, $+1$ oxidation state) are produced as intermediates during the denitrification processes. Fixation involves the assimilation of various inorganic nitrogen species (NH_3, NH_4^+, NO_2^-, and NO_3^-) into organic-N through biological processes. These processes are described in the following sections.

2.2.1 Ammonification

Waste stabilization systems such as animal waste lagoons utilize microbes to produce a sequence of nitrogen transformations, the first step of which is ammonification of the organic nitrogen. This biochemical reaction is the initial step in the mineralization of organic nitrogen. Ammonification occurs during decomposition of animal and plant matter and animal fecal material. The ammonification process results in the release of ammonia as proteins and other nitrogen-containing organic compounds undergo biological conversion and can be expressed as follows:

$$\underset{\text{(proteins, urea)}}{\text{Organic-N}} \quad \xleftrightarrow[\textit{Enzymes}]{} \quad NH_3 + H_2O \longleftrightarrow NH_4^+ + OH^-$$

The enzymes responsible for ammonification are associated with heterotrophic microorganisms with minor participation of fauna.

Rates of ammonification are influenced by factors that include (1) concentration of dissolved oxygen, (2) pH, (3) population of appropriate microorganisms, (4) temperature, and (5) moisture. The rate of ammonification also depends on the biotransformation rate of the nitrogen-containing waste material and the carbon-to-nitrogen ratio of the waste. Ammonification results in an increase in pH, and if the ammonium concentration and pH are sufficiently high, it can lead to volatilization of ammonia into air.

The distribution of NH_3 and NH_4^+ in an aqueous system is largely controlled by the pH of the system. Ammonia is highly soluble in water with an aqueous solubility of approximately 340 g/L at 20°C and 1 atm (Merck Index 1983). Ammonia acts as a weak base in water:

$$NH_3 + H_2O \Leftrightarrow NH_4^+ + OH^- \qquad K_b = 10^{-4.77} \text{ (20°C, 1 atm)}$$

A high solution pH would, therefore, drive the equilibrium toward the presence of ammonia in the neutral NH_3 form (Figure 2-2).

Significant accumulation of NH_3 can occur over long periods of operation in stagnant collection systems such as animal waste lagoons. Presence of nitrogen as ammonia or ammonium ions (NH_4^+) strongly depends on pH and temperature of the waste. At 20°C and a pH 7.0 of most of the NH_3 in water occurs in the NH_4^+ form, while the distribution is about equal between NH_3 and NH_4^+ at a pH of 9.23. At the liquid/water interface of the waste lagoon, ammonia gas may partition into the air, resulting in production of noxious odors. Selected properties and adverse health effects of ammonia gas are listed in Table 2-5.

Electrical conductivity readings have been proposed for quick estimations of ammonia-N and TKN in lagoon supernatants. Using 4-yr averages from six single-cell recycle-lagoons, Westerman et al. (1990) proposed the following linear regression equations relating NH_3-N and TKN to conductivity, C:

$$\text{TKN (mg/L)} = 0.121 \text{ C } (\mu\text{mhos/cm}) - 165 \qquad (2\text{-}1)$$

$$NH_3\text{-N (mg/L)} = 0.132 \text{ C } (\mu\text{mhos/cm}) - 301 \qquad (2\text{-}2)$$

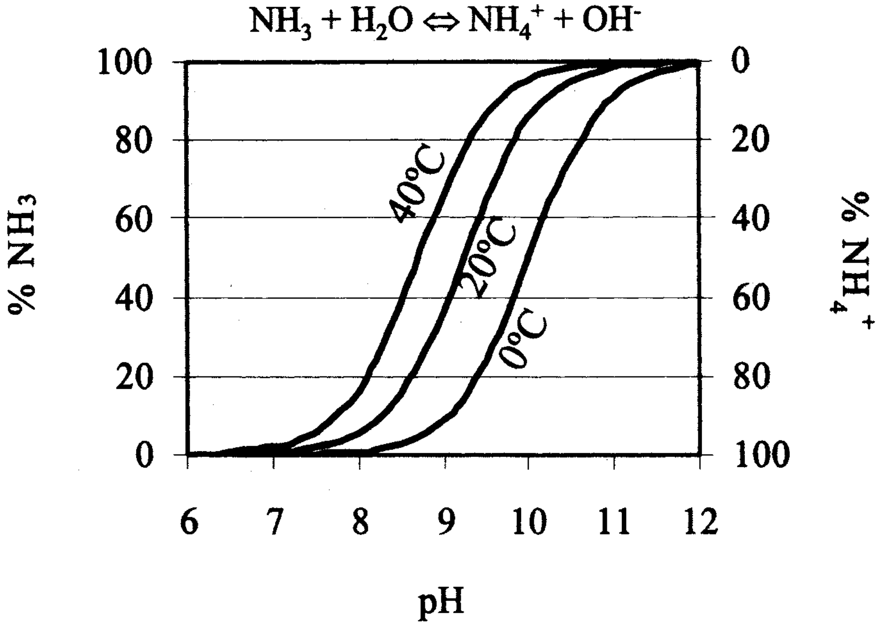


FIGURE 2-2. Effect of pH and Temperature on the Relative Distribution of NH_3 and NH_4^+ in an Aqueous Solution.

TABLE 2-5. Properties and Effects of Ammonia Gas

	Odor	Odor Threshold (ppm)	Max. Allowable Conc.* (ppm)	Concentration Effects Level (ppm)	Exposure (min)	Effect
Ammonia (NH_3)	Sharp, pungent	5	50	400	—	Throat irritant
				700	—	Eye irritant
				1,700	—	Coughing/ frothing
				3,000	30	Asphyxiating
				5,000	40	Fatal

Source: Adapted from Midwest Plan Service 1985.
Note:
*Maximum concentration allowed by health agencies for workers in 8- to 10-h periods.

TKN values were found to be higher in the spring and summer in swine and dairy non-recycle–lagoons (Sutton et al. 1980). Westerman et al. (1990) observed a similar cyclic behavior for NH_3-N in swine and poultry lagoons and attributed it to temperature effects on lagoon biological activity. Boorman et al. (1975) observed a steady increase in TKN in a swine recycle-lagoon over a 6-yr period.

2.2.2 Volatilization

Waste lagoons at CAOs can represent a significant source of NH_3 emissions to the atmosphere in a relatively small geographical area. These emissions have the potential to impact land use choices over several square miles around the lagoons. Harper et al. (2000) measured gaseous emissions from anaerobic swine lagoons to quantify the release of nitrogen as ammonia, nitrous oxide, and dinitrogen gas. Ammonia emissions were found to vary diurnally and seasonally and were correlated with wind speed and water temperature. N_2-N emissions from the lagoons were of the same magnitude as NH_3-N emissions.

Volatilization is an important pathway for permanent nitrogen removal from animal waste lagoons. Volatilization of ammonia takes place when the pH of the system is alkaline and undissociated ammonia is present.

$$NH_4^+ + OH^- \longrightarrow NH_3\uparrow + H_2O$$

Volatilization can be rapid at higher temperatures and depends on the degree of mixing in the lagoon. For example, at a pH of 9.0 and a summertime water temperature of 30°C, approximately 60% of ammonia nitrogen exists in the NH_3 form. Long detention times allow for continued and effective removal of nitrogen by volatilization over the long term. Assuming ammonia volatilization as the primary removal mechanism, nitrogen removal in lagoons can be related to pH, detention time, and temperature according to the following two equations. Eq. (2-3) was developed assuming plug flow conditions in the lagoon (Reed 1985), while Eq. (2-4) assumes a completely mixed reactor configuration (Pano and Middlebrooks 1982):

$$N_e = N_o e^{-k_T\{t + 60.6(pH - 6.6)\}} \tag{2-3}$$

where

N_o = influent total nitrogen (mg/L)
N_e = effluent total nitrogen (mg/L)
k_T = temperature-dependent removal coefficient, $\text{day}^{-1} = k_{20}\theta^{(T-20)}$
k_{20} = removal coefficient at 20°C, $\text{day}^{-1} = 0.0064$
$\theta = 1.039$

T = water temperature (°C)
t = time (days)

$$N_e = N_o\left[\frac{1}{1 + t(0.000576T - 0.00028)e^{\{(1.08-0.042T)(pH-6.6)\}}}\right] \qquad (2\text{-}4)$$

2.2.3 Nitrification

Ammonia resulting from the ammonification process in animal waste lagoons can be, to some extent, converted to nitrate by biological nitrification in the lagoon or in the subsoil. Under aerobic conditions that usually occur near the air/water interface, ammonia-nitrogen can be oxidized to nitrate by microorganisms. Nitrification in aerobic systems is mediated by gram-negative, aerobic, chemoautotrophic, rod-shaped bacteria. Seven genera of autotrophic nitrifiers are known, but bacteria belonging to the genera *Nitrosomonas* and *Nitrobacter* are considered to be the predominant nitrifying organisms in nature. These organisms use ammonia or nitrite as their energy source, inorganic carbon (carbon dioxide or carbonates) as the carbon source for synthesis of cellular biomass, and molecular oxygen as the terminal electron acceptor.

The theoretical amount of oxygen required for complete oxidation of ammonia is 4.57 g/g of ammonium-N (Stensel and Barnard 1992). The USEPA recommends use of the following equation, which incorporates both cellular biomass synthesis (assuming yield coefficients of 0.15 g/g NH_4^+-N oxidized and 0.02 g/g NO_2^--N oxidized) and oxidation considerations and illustrates that the actual oxygen requirement is approximately 4.2 g/g ammonium-N (USEPA 1995):

$$\begin{aligned} &NH_4^+ + 1.83\,O_2 + 1.98\,HCO_3^- \longrightarrow \\ &\qquad \underset{(\textit{biomass})}{0.21\,C_5H_7O_2N} + 0.98\,NO_3^- + 1.041\,H_2O + 1.88\,H_2CO_3 \end{aligned}$$

Nitrification is a two-step biological oxidation of ammonium to nitrate that results in the consumption of alkalinity (reduction of pH).

$$\begin{array}{ccccc} O_2 & & O_2 & & \\ + & & + & & \\ NH_4^+ & \xrightarrow[\textit{bacteria}]{\textit{Nitrosomonas}} & NO_2^- & \xrightarrow[\textit{bacteria}]{\textit{Nitrobacter}} & NO_3^- \end{array}$$

The transformations from ammonium to nitrite and from nitrite to nitrate usually proceed rapidly to the nitrate form. The overall free energy change for conversion of NH_4^+ to NO_3^- is −83.2 kcal/mole. Oxidation of ammonium by *Nitrosomonas* is usually the rate-limiting step resulting in relatively low nitrite levels at any given time.

Key environmental factors that influence the rate of nitrification include (1) temperature, (2) pH, (3) alkalinity, (4) inorganic carbon source, (5) dissolved oxygen concentration, (6) microbial population, and (7) concentrations of ammonia and nitrite. Nitrifiers are very sensitive organisms and susceptible to variations in environmental conditions and the presence of toxic chemicals in waste lagoons. Several compounds inhibit nitrification: Mercaptans, tannins, nitrous acid, and undissociated ammonia are among those that may be encountered in waste lagoons. Nitrifying bacteria are obligate aerobes and dissolved oxygen concentrations above 1 mg/L are essential for biological nitrification. Oxygen limitations in anaerobic animal waste lagoons, however, result in suppression of nitrification and accumulation of ammonia. Low nitrate and nitrite levels observed in unaerated lagoons are indicative of the fact that nitrification does not account for a significant portion of ammonia removal in these systems.

Solution pH and temperature can exert significant influences on biological nitrification. The optimum pH range for nitrifying microorganisms varies between 6.6 and 8.0 (Paul and Clark 1989). At high pH, the transformation of NO_2^- to NO_3^- is inhibited by the presence of NH_4^+. The optimum temperature for biological nitrification ranges between 30°C and 35°C. Nitrification slows down considerably at temperatures lower than 5°C or higher than 40°C. Nitrification rates in temperate climates are slow in summer and winter and are highest in spring and fall (Paul and Clark 1989).

Nitrification kinetics have been modeled using zero-order, first-order, and the Monod equation for limiting substrate. Eq. (2-5) incorporates the effects of dissolved oxygen, temperature, and pH into the Monod expression used to estimate the specific growth rate (μ_N) of nitrifying microorganisms (Crites and Tchobanoglous 1998):

$$\mu_N = \mu_{N\max}\left(\frac{N}{K_N + N}\right)\left(\frac{DO}{K_{O_2} + DO}\right)\left(e^{0.098(T-15)}\right)[1 - 0.0833(7.2 - pH)] \qquad (2\text{-}5)$$

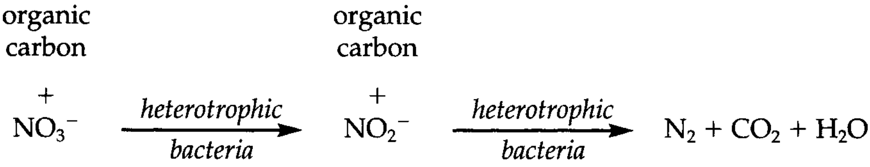
where

$\mu_{N\max}$ = maximum specific growth rate for nitrifiers (day^{-1}), range = 0.4 to 2.0
N = ammonia concentration (mg/L)
K_N = half-saturation constant for ammonia (mg/L), range = 0.2 to 3.0
DO = dissolved oxygen concentration (mg/L)
K_{DO} = half-saturation constant for DO (mg/L)
T = water temperature (°C)

2.2.4 Denitrification

Losses of nitrogen in animal waste lagoons may also occur in the form of nitrogen gas as a result of the ammonification-nitrification-denitrification sequence of reactions. Two major biological pathways are involved in the removal of nitrate from animal waste lagoons: the assimilatory and dissimilatory nitrate reduction pathways. Assimilatory nitrate reduction involves the biological conversion of nitrate to ammonia for subsequent use in the synthesis of cellular biomolecules, such as amino acids, amino sugars, proteins, and nucleic acids. This process occurs at low NH_4^+ concentrations and is not affected by O_2 levels. Due to the high NH_4^+ loading in waste lagoons, assimilatory nitrate reduction does not play a significant part in nitrate removal.

Dissimilatory nitrate reduction or denitrification involves the use of nitrate as the terminal electron acceptor in the absence of molecular oxygen. In the past, this process was referred to as anaerobic denitrification. However, elucidation of the biochemical pathways involved in denitrification has shown that these pathways are modifications of aerobic pathways in which nitrate serves as the electron acceptor (Crites and Tchobanoglous 1998). This has resulted in the use of the term anoxic instead of anaerobic for biological denitrification processes. Under anoxic conditions, nitrate is sequentially reduced to NO_2^-, NO, N_2O, and finally to N_2 gas by facultative denitrifying bacteria. The reductions proceed through several steps.



$$\begin{matrix}\text{organic}\\ \text{carbon}\\ +\\ NO_3^-\end{matrix} \xrightarrow[\textit{bacteria}]{\textit{heterotrophic}} \begin{matrix}\text{organic}\\ \text{carbon}\\ +\\ NO_2^-\end{matrix} \xrightarrow[\textit{bacteria}]{\textit{heterotrophic}} N_2 + CO_2 + H_2O$$

A large variety of heterotrophic bacteria participate in denitrification processes. Denitrifying bacteria derive energy for growth from nitrate reduction and require a source of carbon for cell synthesis. Genera involved in denitrification include *Acinetobacter, Alcaligenes, Bacillus, Glunobacter, Halobacterium, Micrococcus, Moraxella, Paracoccus, Pseudomonas, Rhodopseudomonas, Spirillum, Thiobacillus,* and *Xanthomonas* (Canter 1997).

Zero-order, first-order, and Monod rate expressions have been used to model denitrification reactions. In the case of animal waste lagoons, where the concentration of carbon is not limiting, the denitrification rate is usually zero-order down to very low nitrate concentrations. Other factors affecting biological denitrification include (1) oxygen concentration, (2) nitrate concentration, (3) temperature, (4) pH, and (5) the population of viable denitrifying bacteria. Most denitrifiers grow optimally at or near neutral pH. Denitrification is usually negligible below pH 4. Denitrification can proceed over a wide temperature range from about 5°C to nearly 75°C (Paul and Clark 1989). Denitrification may not be a major nitrogen-transformation pathway in anaerobic animal waste lagoons where the production of nitrate from ammonia is suppressed by low dissolved oxygen concentrations. However, Sund et al. (2001) studied a two-stage (anaerobic and facultative) swine waste lagoon and found that photosynthetic-bacteria–mediated nitrogen transformations were significant in the facultative lagoon. Nitrogen-fixing cyanobacteria were active near the surface while heterotrophic denitrifiers had a significant presence at depths greater than 3.0 m.

2.2.5 Plant and Microbial Uptake

Ammonium is readily incorporated into amino acids, and microorganisms preferentially utilize NH_4^+ rather than NO_3^- for protein synthesis. Nitrate, on the other hand, is the preferred source of nitrogen for plants. NO_3^- is readily taken up by plant roots. Nitrate reduction in plants requires energy, which is supplied from the photosynthetic process. The nitrogen is incorporated into amino acids, proteins, and nucleic acids. Nitrogen uptake may outpace assimilation into biomolecules. In such cases, NO_3^- may accumulate in plant tissues.

2.2.6 Deposition

Nitrogen removal from the bulk liquid in animal waste lagoons also can occur by deposition of nitrogen as lagoon bottoms or sludges. Partially degraded or undegraded waste material containing organic nitrogen may settle under gravity or by adsorption on settling particles. Ammonification processes in sludge bottoms may result in the accumulation of ammonium (NH_4^+). The cationic ammonium thus produced can adsorb or bind strongly to the lagoon bottom or liner materials, which are usually constructed of clay particles that have high cation exchange capacities (CEC).

2.2.7 Adsorption

Nitrogen in the positively charged ammonium form can strongly adsorb to mineral surfaces in soils and lagoon liners. This tendency of NH_4^+ attenuates its offsite migration via surface runoff or leaching into groundwater. NH_4^+ can readily replace other monovalent cations in exchange reactions on mineral surfaces. The fraction of NH_4^+ adsorbed on mineral surfaces that can be exchanged back into solution by other cations is defined as the exchangeable NH_4^+. Conversely, NH_4^+ that becomes fixed within the clay mineral lattice and is not readily exchanged by other cations is defined as the nonexchangeable NH_4^+. Nitrogen fixed in this way is highly stable because of its resistance to biological nitrification and uptake. The dynamic equilibrium for NH_4^+ in soils can be represented as follows (Shen et al. 1997):

$$[\text{Solution NH}_4^+] \underset{k_d}{\overset{k_a}{\Leftrightarrow}} [\text{Exchangeable NH}_4^+] \underset{k_r}{\overset{k_f}{\Leftrightarrow}} [\text{Nonexchangeable NH}_4^+]$$

where

k_a and k_d = adsorption and desorption rate coefficients
k_f and k_r = fixation and release rate coefficients

Adsorption of NH_4^+ on exchangeable sites occurs as a fast reaction (on the order of minutes to hours). The exchangeable NH_4^+ is slowly (on the order of days to months) transformed to nonexchangeable NH_4^+. Adsorption and fixation of NH_4^+ on both exchangeable and nonexchangeable sites increase with solution NH_4^+ concentration. Other major factors that influence binding of NH_4^+ to soils and lagoon liners include mineralogical composition of the soil or liner material, pH, and competing cations.

The type of mineral has by far the greatest influence on sorption of NH_4^+. Negative charges on mineral surfaces resulting from isomorphic substitution and ionization of surface functional groups attract cations like NH_4^+. Surface charge created by isomorphic substitution is independent of soil pH and is called permanent-charge. Permanent-charge is negative in most soils. Ionization of surface functional groups creates pH-dependent charges (Bohn et al. 1985). At low pH, pH-dependent charges tend to be relatively more positive, while at high pH they tend to be relatively more negative (Figure 2-3).

Increase in pH can cause greater NH_4^+ adsorption by aluminosilicate minerals, such as zeolites, that have a mixture of permanent and pH-dependent charges (Kithome et al. 1998). Zeolites exhibit high CEC and high NH_4^+-selective properties. Opposite trends are observed for NH_4^+ adsorption by alumino-silicate minerals such as vermiculites, which have predominantly permanent-charges (Shen et al. 1997). NH_4^+ adsorption on exchangeable sites is greater in acid solutions ($pH < 6.0$) than in alkaline solutions ($pH > 7.5$). NH_4^+ fixation in vermiculite is unaffected by solution pH. CEC values for minerals with pH-dependent charges, such as kaolinite, range between 5 and 20 meq/100 g, whereas for minerals possessing permanent-charges, such as vermiculite, CEC values can range from 100 to 150 meq/100 g (McBride 1994).

Divalent cations such as Ca^{2+} and Mg^{2+} are the major competing cations for NH_4^+ adsorption on mineral surfaces (McBride 1994). The ammonium

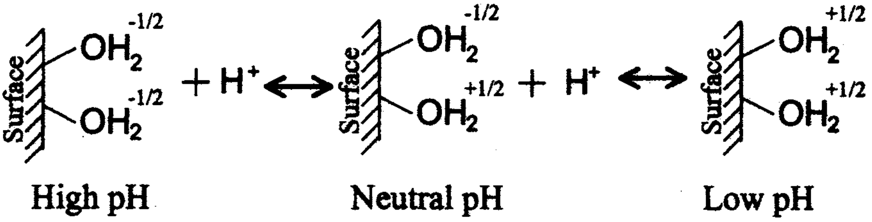


FIGURE 2-3. pH-Dependent Charges on Mineral Surfaces Due to Ionization of Surface Functional Group.

adsorption ratio (AAR) is used to estimate the percentage of the CEC sites that can be occupied by NH_4^+ when the concentrations of NH_4^+ and the principal competing divalent cation in the solution are known (Lance 1972).

$$\mathrm{AAR} = \frac{[\mathrm{NH_4^+}]}{\sqrt{\frac{1}{2}[\mathrm{Ca^{2+}}] + \frac{1}{2}[\mathrm{Mg^{2+}}]}}$$

where concentrations are expressed in milliequivalents per liter.

2.3 Subsurface Transport

The fate of nitrogen in the subsurface depends on the form of nitrogen (organic or inorganic) and the different biochemical and physicochemical processes involved in the transformation of one form of nitrogen to another. Transport processes involved in subsurface migration of nitrogen include diffusion of NH_4^+or NO_3^- and their advective transport in the aqueous phase, as well as adsorption and cation exchange of NH_4^+on soil particles. Biological processes can fix or incorporate nitrogen into microbial or plant biomass.

Ammonium-nitrogen transport from waste lagoons to the subsurface is a diffusion-controlled process. Infiltration of NH_4^+, therefore, depends on its diffusion coefficient and the concentration gradient. Diffusion coefficients for NH_4^+ in Crowley silt loam were found to range from as low as 0.004 cm^2/day with 11% volumetric water content to as high as 0.216 cm^2/day for the flooded soil (Reddy and Patrick 1981). The large concentration gradients at the sludge/water or soil/water interface can create a large driving force for NH_4^+ transport resulting in rapid diffusion of NH_4^+ from the anaerobic sludge bottoms. Ham and DeSutter (1999) calculated ammonium export rates of 2,187 to 2,726 kg/ha^{-1}/yr^{-1} into the subsoil at three swine waste lagoons in Kansas.

Adsorption, cation exchange, and microbial uptake often limit the transport of NH_4^+ in the unsaturated zone below the waste lagoon.

Adsorption to negatively charged soil particles is usually the predominant process controlling the fate of NH_4^+. Infiltration of NH_4^+ can, however, be rapid once the adsorption capacity or the CEC of the soil is exhausted. Under aerobic conditions and in the presence of appropriate microorganisms, NH_4^+ may be oxidized to NO_3^- in the subsurface. Soil properties that control the transport of NH_4^+ include (1) CEC of soil, (2) clay content, (3) soil pH, and (4) the presence of competing cations.

Nitrate-N concentrations as high as 100 mg/L, with frequent detections in the range of 20 to 40 mg/L, have been reported in vadose-zone water below agricultural fields and farming operations that include extensive livestock management (Bouwer 1990). Leaching of nitrates into groundwater is a major cause of concern because of the potential human health impacts from groundwater usage.

Nitrate contamination from animal waste lagoons occurs as a result of the aerobic transformation of organic and inorganic NH_4^+ in waste materials to nitrate via ammonification-nitrification reactions. Nitrate production is much higher in well-mixed or aerated lagoons than in quiescent storage structures. In unmixed lagoons, NO_3^- production occurs primarily near the air/water surface or as a result of mass transfer of oxygen from the atmosphere or its production by algae and plant material.

Nitrate infiltration and migration into the subsurface occurs as a result of diffusive or advective transport. NO_3^- diffuses through soils at much higher rates than NH_4^+. For example, the diffusion coefficient for NO_3^- in the flooded Crowley silt loam was 1.33 cm^2/day as compared to 0.216 cm^2/day for NH_4^+ (Reddy and Patrick 1981). Other soil properties that affect transport of nitrate through the vadose zone may include anion exchange capacity (AEC) of the soil, soil pH, clay and oxide contents, and the presence of other anions, such as SO_4^{2-}. Once nitrate infiltrates into groundwater, it is highly mobile and resistant to degradation.

2.4 CONCLUSIONS

A detailed understanding of the fate and transport of nitrogen compounds in animal waste lagoons is essential because of the potential for surface water and groundwater contamination with nitrate, a compound regulated by the USEPA in drinking water at a concentration of 45 mg/L. Waste lagoons associated with animal operations are commonly designed as containment structures with limited mixing and aeration. Nitrogen input into these lagoons is in the form of ammonium-N or organic-N associated with the animal excreta. Organic-N is converted to NH_4^+ as a result of microbe-mediated ammonification reactions. Under aerobic conditions, the ammonium-N is transformed to NO_3^- by facultative bacteria. The nitrate produced can be reduced to nitrogen gas by denitrifying organisms.

Due to low dissolved oxygen levels in unmixed waste lagoons, nitrification is usually a less significant pathway affecting nitrogen transformations. The predominant pathway influencing the fate of nitrogen compounds is the deposition of organic-N and ammonium-N on lagoon bottoms, resulting in the diffusive transport of ammonium into the subsurface, with the possibility of subsequent oxidation of ammonium to nitrate. Once produced in the subsurface, nitrate, the pollutant of concern, is readily transported over long distances by groundwater.

Chapter 3

SEEPAGE AND TRANSPORT THROUGH ANAEROBIC LAGOON LINERS

Lakshmi N. Reddi, Hugo Davalos, and Mohan V.S. Vonala

Animal waste from concentrated animal operations involving cattle, swine, and poultry is usually stored in earthen-lined anaerobic lagoons or storage basins. The animal waste is often in the form of liquids with suspended solids/organic matter that contains high concentrations of nitrogen. Ninety-five percent of the nitrogen in lagoon waste is in the form of ammonium (NH_4-N) ranging from 130 mg/L to greater than 1,000 mg/L.

Apart from the obvious function of containment, waste lagoons are expected to separate the liquid from the solid phase and to reduce the quantity of solids through a digestion process. Organic matter in the manure is mineralized, leaving a nutrient-rich liquid that typically is used as fertilizer on nearby cropland. Lagoons constructed at sites with coarse-grained soils are often provided with a compacted clay liner to prevent excessive seepage from the lagoon and to limit groundwater contamination below the lagoon. Use of geomembranes (GMs) and geosynthetic clay liners (GCLs) is also increasing in the current lagoon construction practice. At present, many states in the United States are sharply divided on the design standards for waste lagoons. The regulatory standards in general restrict the maximum seepage through the lagoon; however, this seepage rate varies from 0.64 cm/day (0.25 in./day) in the states of Kansas and Nebraska to 0.04 cm/day (1/56 in./day) in the states of Minnesota and Missouri. The inconsistency in the regulations is primarily because some state regulatory agencies accept the notion that, in due course of time, the liner will seal as a result of particulate and biological clogging. Some regulatory agencies support the use of natural/compacted soils with permeability as high as 1×10^{-6} cm/s in anticipation of time-dependent liner sealing.

A common practice of evaluating the suitability of soils as lagoon liner materials is to characterize the soils into four groups (Table 3-1) based on the fines content of the soils and their Atterberg limits (USDA 1997). The Natural Resources Conservation Service (NRCS) suggested this grouping

TABLE 3-1. Grouping of Soil Types According to Their Estimated Permeability*

Group	Description
I	Soils that have less than 20% passing No. 200 Sieve and have a PI less than 5.
II	Soils that have 20% to 100% passing No. 200 Sieve and have a PI less than or equal to 15. Also included in this group are soils with less than 20% passing the No. 200 Sieve with fines having a PI of 5 or greater.
III	Soils that have 20% to 100% passing No. 200 Sieve and have a PI of 16 to 30.
IV	Soils that have 20% to 100% passing No. 200 Sieve and have a PI of more than 30.

Source: USDA 1997.

PI = Plasticity Index.

Note:

*Revised from SNTC Technical Note 716. The grouping was based on permeability tests conducted by NRCS soil mechanics laboratories on about 1,161 test samples.

based on a database of permeability tests that were performed on more than 1,100 compacted-soil samples. The permeability of the soils in each group was believed to vary by a relatively narrow margin. A qualitative understanding of the permeability nature of soils, based on the studies by the NRCS, is shown in Table 3-2. It was indicated that natural soils classified under Groups III or IV usually have permeabilities that result in acceptable seepage losses. According to the NRCS, soils in Group I require a liner to be placed, and "soils in Group II may or may not require a liner. Documentation through laboratory or field permeability testing or by other acceptable alternatives is advised." NRCS cautions that these guidelines are only qualitative and that site-specific hydrogeological conditions may sometimes require liners even for soil in Groups III and IV.

To assess the seepage characteristics of field-scale lagoon liners, the representative soil samples were tested in the laboratory for their coefficient of permeability, k. Although a wide range of permeability tests is available to determine k (Daniel 1994), it has remained common practice to use a compaction permeameter for testing, which involves permeating the sample in the same steel mold in which the soil was compacted using the Standard Proctor method. The coefficient of permeability, k, of the soil samples was obtained using Darcy's law:

$$k = \frac{Q}{iA\Delta t} \tag{3-1}$$

TABLE 3-2. Description of Permeability Nature of the Four Soil Groups

Soil Group	Permeability Nature
I	These soils generally have the highest permeability and, in their natural state, could allow excessive seepage losses. Because the soils have a low clay content, the final permeability value will exceed 10^{-6} to 10^{-7} cm/s.
II	These soils generally are less permeable than the Group I soils but lack sufficient clay to be included in Group III.
III	These soils generally have a very low permeability, good structural features, and only low to moderate shrink–swell behavior.
IV	These soils normally have a very low permeability. However, because of their sometimes blocky and fissured structure, they can often experience high seepage losses through cracks that can develop when the material is allowed to dry. They possess good attenuation properties if the seepage does not move through the cracks.

Source: USDA 1997.

where

Q = total volume of effluent collected
i = pressure gradient applied
A = area of cross section of the soil sample
Δt = time period of effluent collection

To estimate seepage rates in the field, Darcy's law was again employed in a modified form:

$$q = ki \tag{3-2}$$

where

q = seepage rate through the liner per unit area
k = permeability of the liner material, determined using Eq. (3-1)
i = pressure gradient representative of field conditions

Assuming a total head of 20 ft in the lagoon, the gradient can be estimated as

$$i = \frac{(20 + d)}{d} \tag{3-3}$$

where d = thickness of the liner in feet. The seepage rate, q, may then be obtained combining Eqs. (3-2) and (3-3) as

$$q = \frac{k(20 + d)}{d} \tag{3-4}$$

Figure 3-1 shows the schematic of seepage rate calculation. It is q that is regulated by the state agencies. For instance, the Kansas Department of Health and Environment (KDHE) allows a maximum q of 0.25 in./day. Obviously, it is crucial for lagoon owners or operators to be able to demonstrate that the permeability, k, of the liners made of either compacted clays or composite liner systems (with GM/GCL) is such that q is less than 0.25 in./day.

A more rigorous approach to assessing the integrity of a containment system is to assess the quality of the leachate exiting the liner system. This involves evaluation of the mass transport and transfer characteristics with respect to the contaminant of interest (i.e., diffusion, retardation, and decay coefficients). Considering that NH_4-N is the chemical of interest, the laboratory testing would involve permeating the compacted clay samples (representing the field liner) with waste streams containing NH_4-N (preferably those obtained from a real operating lagoon) and monitoring and characterizing the effluent. Although a number of custom-tailored column tests could be conceived for the laboratory evaluation of the mass

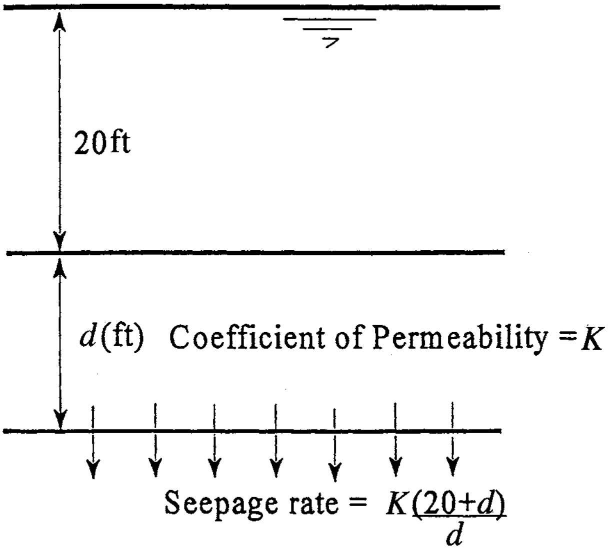


FIGURE 3-1. *Schematic of Seepage Rate Calculation.*

transfer and transport characteristics, the simplest setup would be to use the same compaction permeameters used to assess seepage characteristics, as discussed previously. The diffusion, retardation, and decay coefficients may then be used in an advection-dispersion-reaction (ADR) equation to simulate the transport of NH_4-N through a field-scale liner. The concentrations of NH_4-N exiting the liner may be used as boundary conditions for the soil strata beneath the liner, simulating NH_4-N migration to the water table.

3.1 EXPERIMENTAL PROGRAM TO ASSESS SEEPAGE CHARACTERISTICS OF SOUTHWEST KANSAS SOILS

In view of the importance of southwest Kansas for swine industries, soil samples were acquired from three different areas in Stevens County. Native soils are typically used in lagoon liner construction in that region, and these soils are believed to be representative of soils used in existing lagoon liners. Table 3-3 presents the general characteristics of these soils and their U.S. Department of Agriculture (USDA) grouping. Soil No. 1, a coarser material, was acquired with the intent of creating a modified soil by adding bentonite. Soil No. 1B refers to the mixture of 94% Soil No. 1 and 6% commercial bentonite. The fines contents (using hydrometer and sieve analyses) and the Atterberg limits reported in Table 3-3 were determined using ASTM standard tests D-422, D421, and D-4318, respectively. All of these soil types belonged to Group II, hence the emphasis on the laboratory component of this study was to check if the soils meet the seepage standards imposed by the KDHE. According to the USDA soils survey, the three soil types originated from Dalhart-Otero fine sandy loams (Dx), Vona loamy fine sand (Vo), and Vona-Tivoli loamy fine sands (Vx) (Soil No. 1), and from Richfield silt loam (Rm) and Ulysses silt loam (Ua) (Soil Nos. 2 and 3).

Figure 3-2 shows the compaction characteristics of the three soil types. The optimum molding water content varied from 10% to about 23%, and

TABLE 3-3. Physical Characteristics and USDA Grouping of the Three Soil Types from Southwest Kansas

Soil No.	Liquid Limit (%)	Plastic Limit (%)	Plasticity Index (PI)	Percent Fines (−200)	USDA Group
1B	25	2	23	8	II
2	39	15	24	15	II
3	45	23	22	17	II

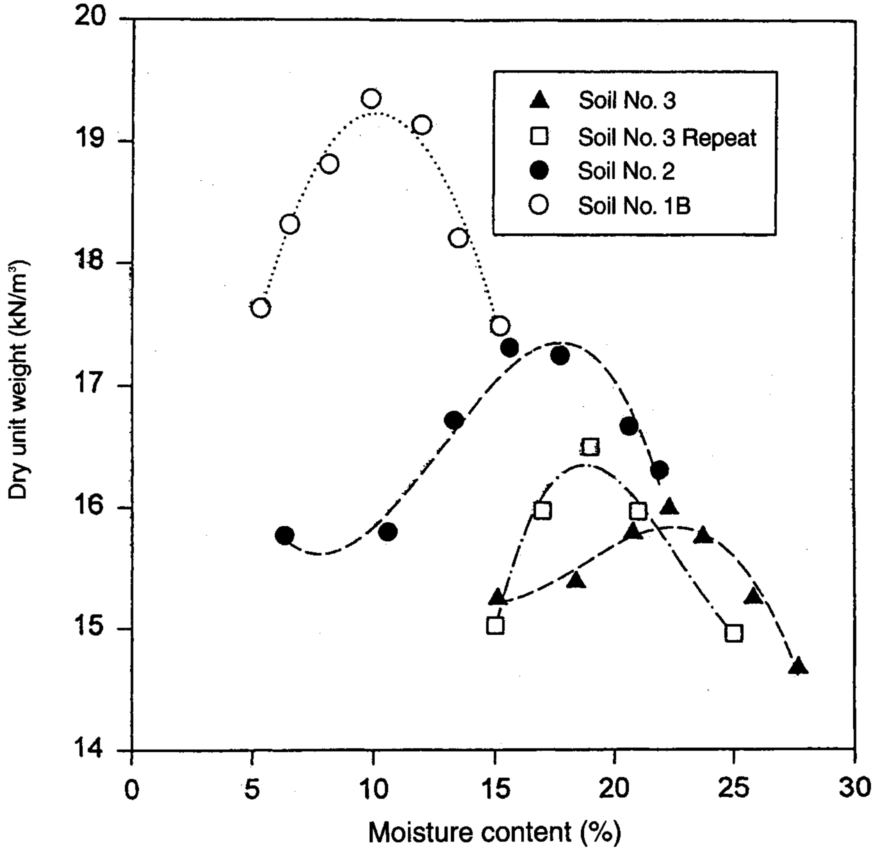


FIGURE 3-2. Compaction Characteristics of Selected Soils from Southwest Kansas.

the peak dry densities exhibited a wide range, from 15.5 to 19.3 kN/m^3. Of importance in these results is the behavior of the bentonite-mixed soil sample, which had a higher maximum dry density than the other two mixtures. As will be evident from the seepage results described in this chapter, modifying coarse soils with nominal percentages of bentonite is a viable alternative to natural borrow materials. Soil No. 3 was obtained in two different batches from the field site. This allowed an examination of the natural variability of the material. As seen in Figure 3-2, the compaction curves for the two soils differed to some extent. However, this difference is not significant enough to cause major differences in hydraulic conductivity.

After the compaction characteristics of the soils were identified, soil samples were prepared in the same molds with selected molding water contents for determining the seepage behavior of soils. There were two main reasons for selecting this method to determine the permeability as opposed to the several existing methods, including flexible wall permeameter. First, compaction permeameters enable a simple and quick determination of leachate quantity and quality as opposed to flexible wall permeameters (Shackelford and Daniel 1991a, 1991b; Daniel 1994). Second, although they are known to suffer from sidewall leakage and overestimation of the permeability, the seepage estimates will be upper bound and will be on the conservative side from the viewpoint of meeting regulatory standards.

A total of 22 soil samples was tested: three for Soil No. 1B, seven for Soil No. 2, and twelve for Soil No. 3. Emphasis was placed primarily on Soil No. 3 because of the importance of its place of origin to regulatory agencies. Samples belonging to the three soil types were first permeated with water (for permeability determination) under gradients provided by air pressure. A subset of the samples was subsequently permeated with the waste liquids for leachate quality determination. The effluent from the permeameters was closely monitored at regular time intervals. Prior to application of gradients, the soil samples in the permeameters (each 5 cm thick) were allowed to saturate under standing liquids. In all cases, permeabilities were determined at the gradients corresponding to the maximum operating head of lagoons [i.e., 6.1 m (20 ft)].

Because the seepage characteristics of compacted clays are known to exhibit variability, reproducibility experiments were conducted on duplicate soil samples. Also, a few experiments were duplicated by different personnel to account for possible human errors, as per the suggestions of the regulatory agency. Permeability experiments also were conducted at several points on the compaction curve, including optimum water content, wet of optimum, and dry of optimum.

After a steady flow rate of effluent was observed in each of the experiments, the permeability experiments were terminated, and water on the top of soil samples was replaced by samples of swine or livestock waste acquired from various facilities in both Kansas and Iowa. About 10 gal. of waste suspension were transported to the laboratory, and subsamples from the suspension were pipetted into the permeameters. Although the waste suspension was thoroughly agitated prior to pipetting, it was thought that the suspended solid quantities in the subsamples were less than those of the main suspension. This was not viewed as a limitation of the study since under-representation of particles in the suspension can only give an upper bound value of leachate quantities. The leachate quantities were collected in 5-ml sample bottles and were periodically sent for chemical and biological analyses (primarily Cl, NH_4-N, NO_3-N, and microbial counts). The influent was replenished at approximately 2-week intervals. At the time of each replenishment, control samples of the waste influent were sent for analytical testing.

The permeabilities obtained for the three soil types are shown in Table 3-4. The entries in bold type correspond to the long-term experiments where waste liquid was used as the influent after permeability determination with water. These experiments ran up to 4 months, and the permeabilities reported in Table 3-4 were the averages of values measured at 1-day intervals. As shown in Table 3-4, the permeabilities varied in a range of two orders of magnitude, with the maximum and minimum

TABLE 3-4. Results from Permeability Tests and Seepage Estimates for a Static Head of 6.1 m (20 ft) and a Liner Thickness of 0.92 m (3 ft)

Sample No.*	Permeability (cm/s)	Seepage Rate Under 6.1-m Head (cm/day)
1B.10	7.42E-09	4.91E-03
1B.14	**142E-07**	**9.41E-02**
1B.7	1.70E-08	1.12E-02
2.12.1	9.05E-08	6.00E-02
2.12.2	1.33E-07	8.80E-02
2.17.1	1.78E-08	1.18E-02
2.17.2	2.11E-08	1.39E-02
2.22.1	4.95E-07	3.28E-01
2.22.2	**2.33E-07**	**1.54E-01**
2.22.3	9.20E-08	6.09E-02
3.18.1	1.09E-08	7.22E-03
3.18.2	**4.75E-09**	**3.15E-03**
3.18.3	**2.33E-08**	**1.54E-02**
3.20.1	**1.30E-08**	**8.60E-03**
3.20.2	**2.94E-08**	**1.95E-02**
3.22.1	2.01E-08	1.33E-02
3.22.2	**2.00E-08**	**1.32E-02**
3.22.3	**2.33E-08**	**1.54E-02**
3.24.1	2.88E-07	1.91E-01
3.24.2	**4.27E-07**	**2.83E-01**
3.24.3	6.18E-08	4.09E-02
3.26	2.41E-08	1.59E-02

Note:
*Sample no. notation = x.y.z, where x = soil type, y = molding water content, and z = sample number.

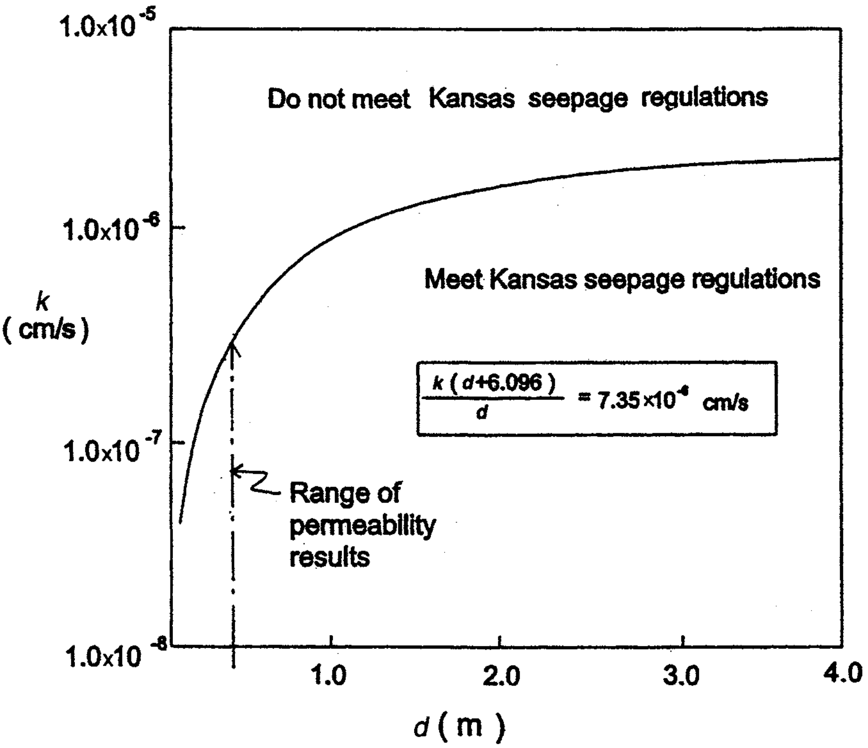


FIGURE 3-3. Relationship between Permeability and Seepage Rate.

coefficients of permeability being 4.95×10^{-7} cm/s and 4.75×10^{-9} cm/s, respectively. The corresponding seepage rates in the field for a liner of thickness $d = 0.9$ m (3 ft), and a static head of 6.1 m (20 ft), varied between 0.33 and 0.0032 cm/day. For the range of k observed in the present laboratory investigation, the field lagoon liners will meet the KDHE standards (i.e., seepage should be less than 0.64 cm/day) even if the liner thickness is as low as 0.5 m (1.6 ft). This is shown schematically in Figure 3-3.

3.2 EXPERIMENTAL PROGRAM TO ASSESS TRANSPORT CHARACTERISTICS OF SOUTHWEST KANSAS SOILS SUBJECTED TO LAGOON WASTE

For the three samples that were tested with swine manure (1B.14, 2.22.2, and 3.24.2), the permeabilities declined somewhat immediately after the introduction of the waste liquid and then leveled off, as shown in Figure 3-4. In all of the cases, the reduction occurred in the initial time periods, hence it was attributed to the physical clogging of samples

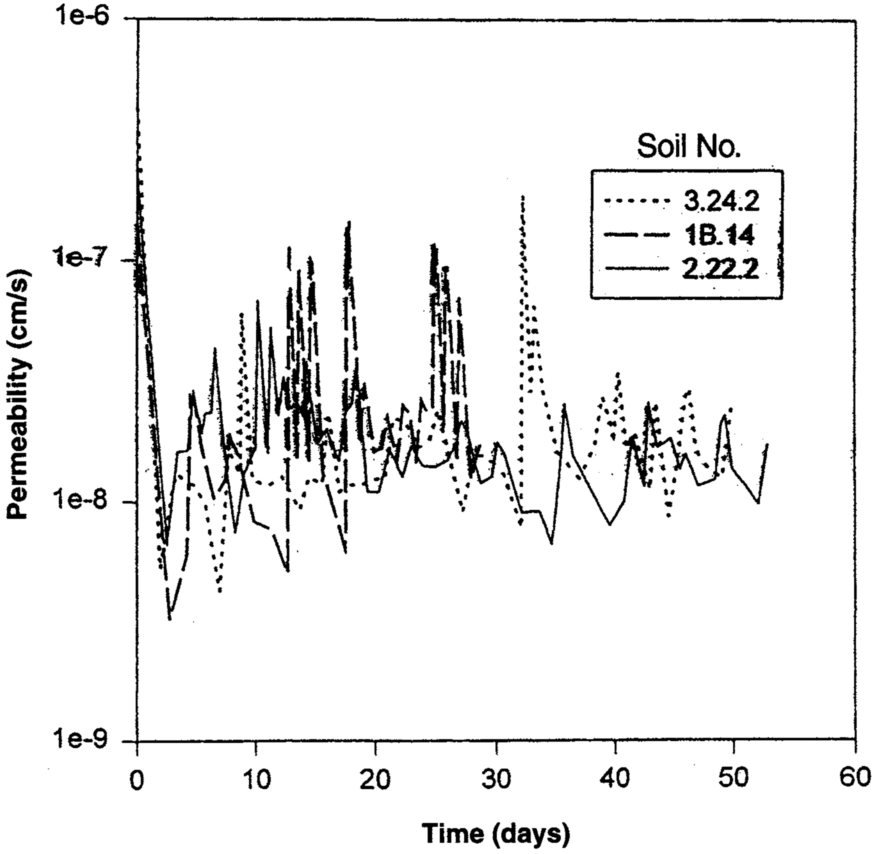


FIGURE 3-4. Permeability versus Time for the Three Soil Types Subjected to Swine Waste from Iowa Farms.

caused by fine particles in the influent suspension. The local variations in the permeabilities seen in Figure 3-4 are attributed to the sample collection scheme, which involved minor variations in emptying the small polyethylene tube connecting the vial to the effluent port of the permeameters. Samples of Soil Nos. 1B and 2 had to be disassembled due to lack of cooperation from the farms that supplied the soils and waste liquids, and because of some unforeseen legal implications. Throughout the remainder of this chapter, discussion is restricted to samples belonging to Soil No. 3.

Permeability variations for samples subjected to livestock waste and swine waste from southwest Kansas are shown in Figures 3-5 and 3-6, respectively. The permeabilities started declining after being steady during the first 2 months. The exception is Sample 3.18.2, which maintained low permeabilities throughout. Microbial counts for the effluent samples during the testing period are shown in Figure 3-7. In spite of the sharp

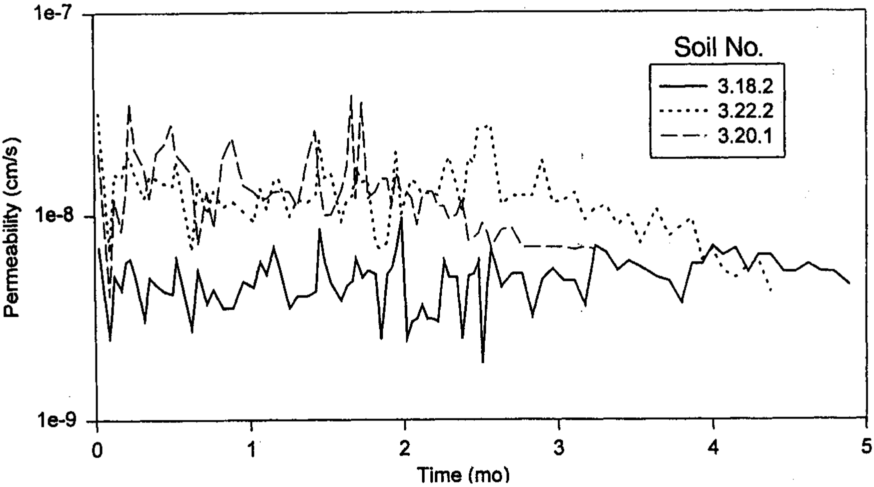


FIGURE 3-5. Permeability versus Time for Soil Type 3 Subjected to Cattle Waste from Southwest Kansas.

increases in microbial counts observed in all three cases during the first 2 months, the biomass growth inside the sample did not seem to be sufficient to bring down the permeability of the samples. The fastest breakthrough of NH_4-N occurred right around 2 months for the laboratory sample. This indicates that it may not be a conservative design practice to

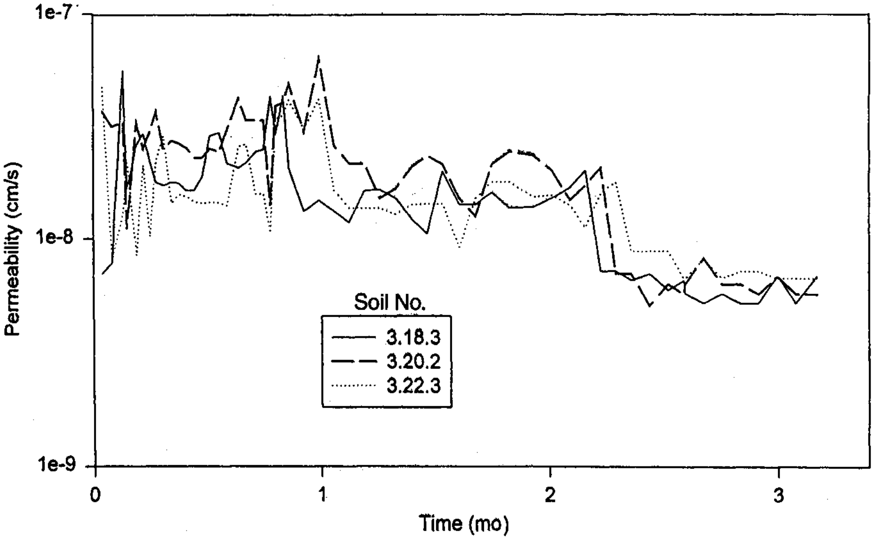


FIGURE 3-6. Permeability versus Time for Soil Type 3 Subjected to Swine Waste from Southwest Kansas.

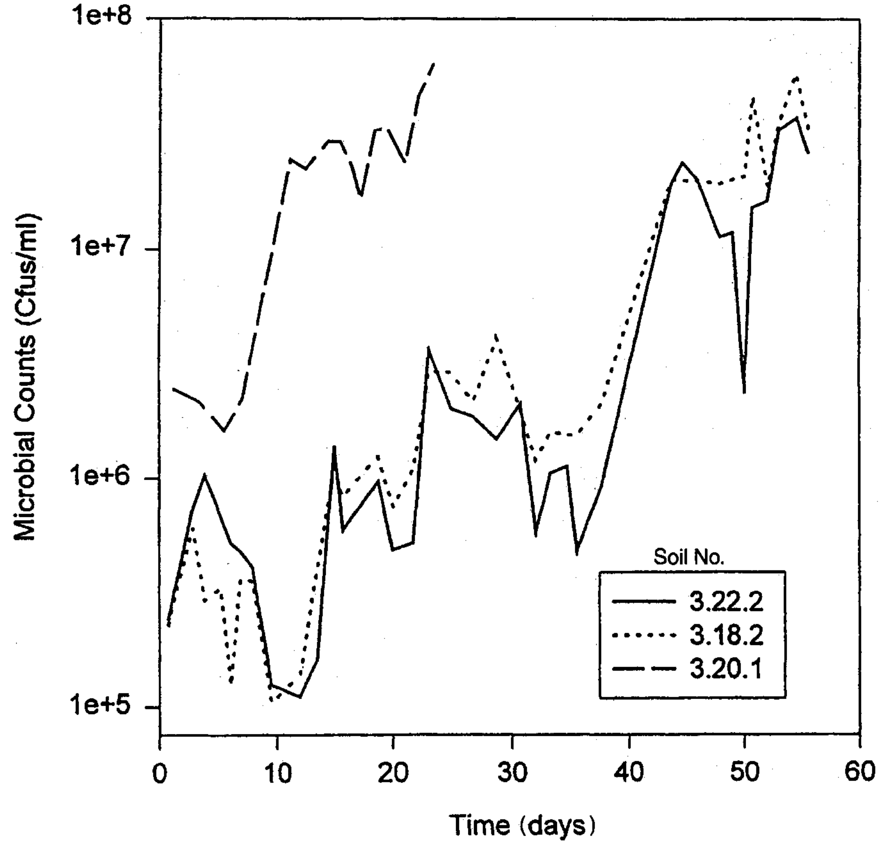


FIGURE 3-7. Microbial Counts in the Effluents from Samples Subjected to Cattle Waste.

account for time-dependent reductions in permeability of clay liners due to biological clogging. Literature is sharply divided on the issue of whether biological clogging is prominent in lagoon liners. Although a 3-fold reduction in permeability in the case of clays and a 25-fold reduction in the case of sands was reported in the literature (Barrington et al. 1987a, 1987b; Barrington and Broughton 1988; Barrington and Madramootoo 1989; Rowsell et al. 1985), much of the reduction was attributed to the mat of organic sludge and sediment settled on the top of the liner material. Physical clogging involving sealing of the pores by particulates was in general considered the primary sealing mechanism. Considering that the sides of the lagoon may not allow for the formation of a sealing layer, it appears prudent not to consider time-dependent reductions in permeability of the liner in the design process.

Chloride and NH_4-N concentrations in the effluent are shown in Figures 3-8 and 3-9 for the three samples tested with livestock waste from

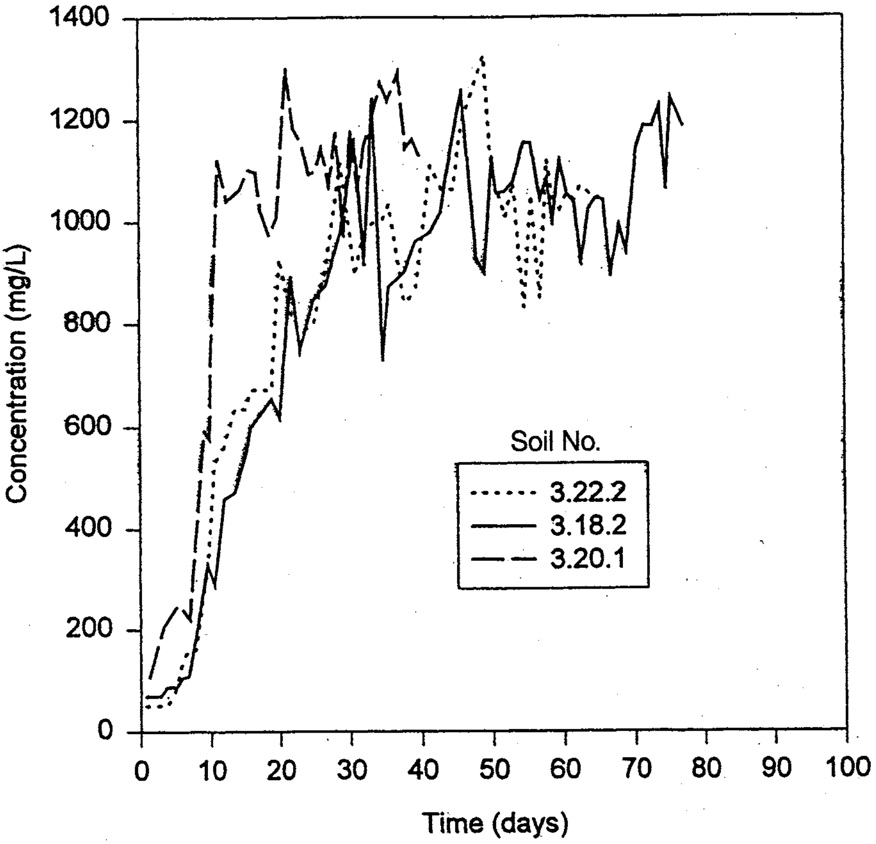


FIGURE 3-8. Chloride Concentrations in the Effluents from Samples Subjected to Cattle Waste.

Kansas farms. In general, high concentrations of ammonium and chloride were observed in the effluents of all three samples. Considering the average concentrations in the influent of ammonia (235 mg/L) and chloride (1,020 mg/L), it is clearly seen that breakthrough was achieved relatively faster in the case of chloride, whereas it was significantly retarded and decayed in the case of NH_4-N. The decay of NH_4-N is attributed to its consumption by microorganisms. Nitrogen is available as a source to the organisms in two forms: NH_4^+ and NO_3^-. The effluent concentrations of NO_3-N, not shown here, are all steady at about 0.5 mg/L. When nitrogen is present in both forms, organisms prefer the ionic source of N in NH_4^+ for protein synthesis. Preferential uptake of NH_4^+ over NO_3^- has been demonstrated in several studies (Paul and Clark 1996).

The retardation of ammonium is expected because of the cation adsorptive capacity of negatively charged clay particles. Significant retardations observed in Figure 3-9 can be understood in terms of the potential NH_4

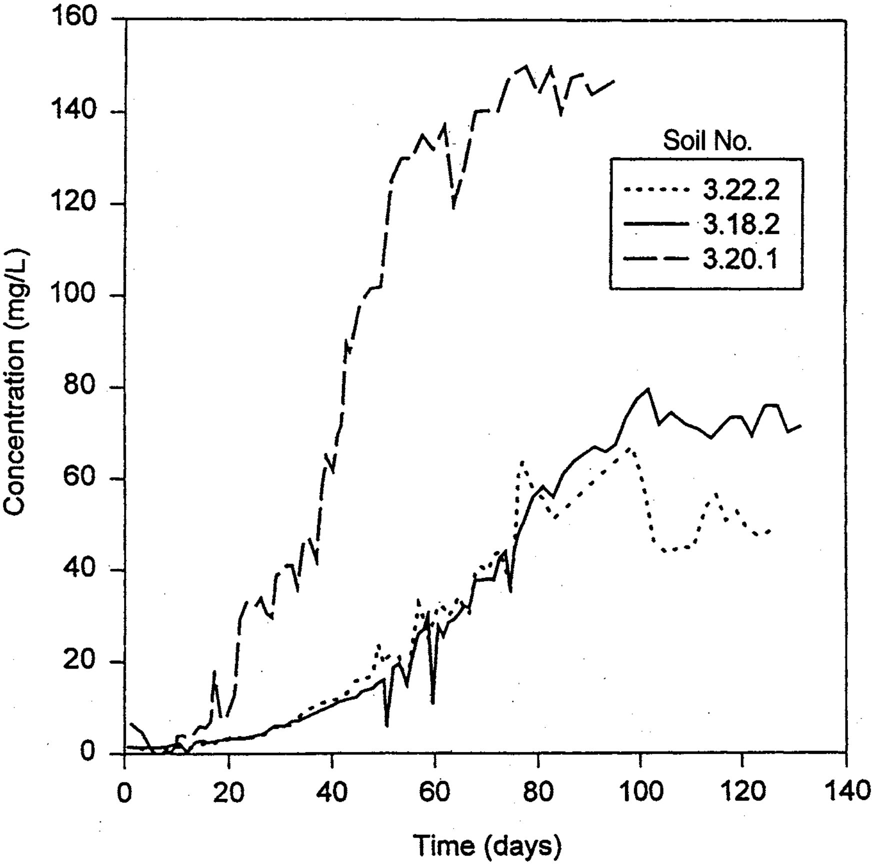


FIGURE 3-9. NH_4-N Concentrations in the Effluents from Samples Subjected to Cattle Waste.

saturation rates of soils. The adsorption of ammonium is primarily dependent on the cation exchange capacity (CEC) of the soil and the concentration of the other competing cations in the waste (e.g., calcium and magnesium). Using representative concentrations of the various chemical constituents from waste lagoons in Kansas, and using the lower bound CEC of soils at the site from where the soils of this study were taken (10 cmol/kg), Ham et al. (1998) showed that annual ammonium losses into the bottom soils of lagoons could be adsorbed in an equivalent soil depth of only 7.5 in. Such low cation saturation rates also were documented by Barrington et al. (1987b), who reported that NH_4 in natural dairy and hog slurries could be adsorbed at the rate of 0.01 to 0.04 m/yr in clays with a CEC of 25 cmol/kg. Thus, it is possible that the clay liners may be used as reservoirs to conserve nitrogen in the form of NH_4 for possible use as fertilizer when the lagoon is emptied and cleaned.

The mass transport parameters of chloride and ammonium were obtained, fitting the solution of an ADR equation to the experimental data. The governing differential equation for the solute transport in the compacted-soil column can be written as follows (van Genuchten 1981; Shackelford 1993):

$$\frac{\partial c}{\partial t} = \frac{D_h \partial^2 c}{R_d \partial x^2} - \frac{v_s \partial c}{R_d \partial x} - \lambda c \qquad (3\text{-}5)$$

where

c = concentration of the solute
D_h = dispersion coefficient
R_d = retardation coefficient
λ = decay coefficient
v_s = seepage velocity in the soil column
x and t = space and time coordinates, respectively

The analytical solution for the above differential equation and for the initial condition, $c = 0$ $(x \geq 0)$, and boundary conditions, $c = c_0$ $(x = 0)$ and $dc/dx = 0$ $(x = \infty)$ is given by Ogata and Banks (1961):

$$\frac{c(x,t)}{c_0} = \frac{1}{2}\exp\left[\frac{(v_s - U)x}{2D_h}\right]\operatorname{erfc}\left[\frac{R_d x - Ut}{2\sqrt{D_h R_d t}}\right] + \frac{1}{2}\exp\left[\frac{(v_s + U)x}{2D_h}\right]\operatorname{erfc}\left[\frac{R_d x + Ut}{2\sqrt{D_h R_d t}}\right] \qquad (3\text{-}6)$$

where $U = (v_s^2 + 4D_h R_d \lambda)^{1/2}$.

A simple program was written in Matlab to compute the squared differences between experimental observations and the analytical solution (Eq. (3-6)) for chosen values of R_d, D_h, and λ. In the case of chloride, $R_d = 1$ and $\lambda = 0$ because it is a conservative solute. The values of R_d, D_h, and λ ultimately selected (shown in Table 3-5) were those corresponding to the minimum squared differences. A representative fitting curve used to obtain these values is shown in Figure 3-10. It is important to note the significant differences in the decay coefficient among the three samples of the same soil type. It is apparent that the microorganism uptake of NH_4-N may depend on the pore structure, which is influenced by the molding water content. The decay coefficient plays an important role in NH_4-N transport. Pore scale examination of microbial colony formation may lead

TABLE 3-5. Best-Fit Estimates of Fate and Transport Characteristics of NH_4-N

Sample No.	Longitudinal Dispersivity, D_h (cm^2/yr)	Retardation Factor, R_d	Decay Coefficient (yr^{-1})
3.18.2	69	4.0	1.0
3.20.1	116	5.6	0.2
3.22.2	42	4.2	0.6

to useful insight on how molding water contents influence microbial activity; however, that is beyond the scope of the present work.

In a preliminary study, samples of GMs and GCLs were subjected to the lagoon waste in plexiglass columns to evaluate their transport characteristics. This study, in progress at the time of this writing, appears to indicate that GCLs could act as reservoirs for NH_4-N and may significantly delay migration of NH_4-N (Figure 3-11). Geomembranes, on the other hand, do not tend to diffuse noticeable quantities of NH_4-N. These materials also appear to retain their integrity (fabric and structure) even after prolonged contact with NH_4-N.

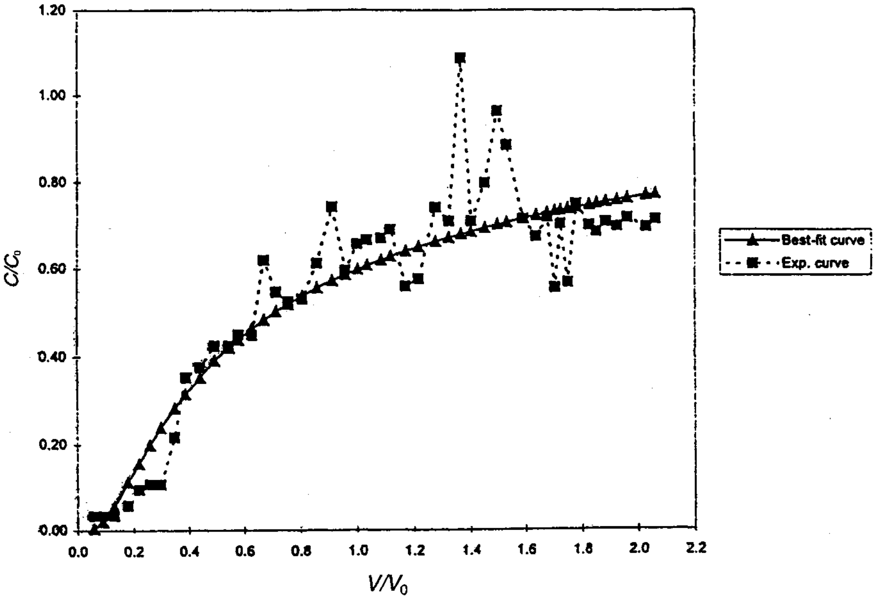


FIGURE 3-10. A Representative Best-Fit Curve versus Experimental Data (Data Shown Xre for Chloride in Soil Sample 3.22).

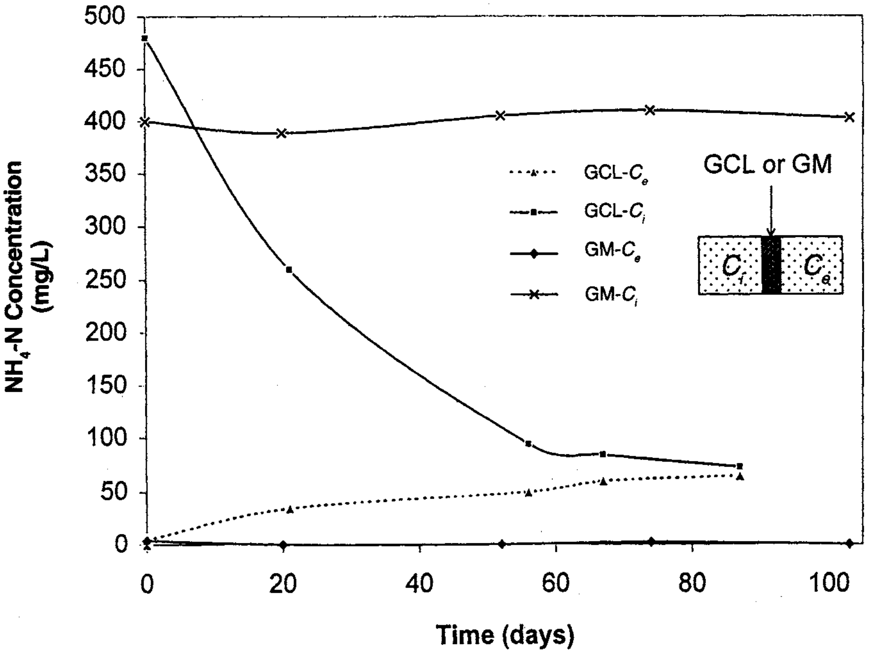


FIGURE 3-11. Diffusion of NH_4-N through Geomembrane and Geosynthetic Clay Liner Samples.

3.3 TRANSPORT THROUGH FIELD-SCALE LINERS

The best-fit estimates shown in Table 3-5 were used in the analytical solution (Eq. (3-6)) to obtain possible breakthrough curves for field-scale liners. The results for liners of three different thicknesses made of Soil No. 3.22 are shown in Figure 3-12. It is clear that maximum NH_4-N loading into the natural soils could be decreased by 90% of the lagoon source, and the time for its breakthrough could be greater than 10 yr for a liner thickness of 0.9 m (3 ft). Removal of a few top centimeters of the NH_4-N–saturated layer prior to this period (during cleanup of the lagoon bottom) and its replacement with a clean soil could significantly improve these numbers. As an example, consider the case where the top 10 cm of the liner is scraped and replaced with a new layer of the same material. The relative concentrations at the bottom of the liner could be drastically reduced if this was done at 5-yr intervals. As shown in Figure 3-13, the C/C_0 after 25 yr at the bottom of a liner that had undergone scrape-and-replace three times at 5-yr intervals was reduced to 0.085. However, practical issues surrounding the feasibility of such a replacement without causing damage to the liner must be examined prior to implementing such a recommendation.

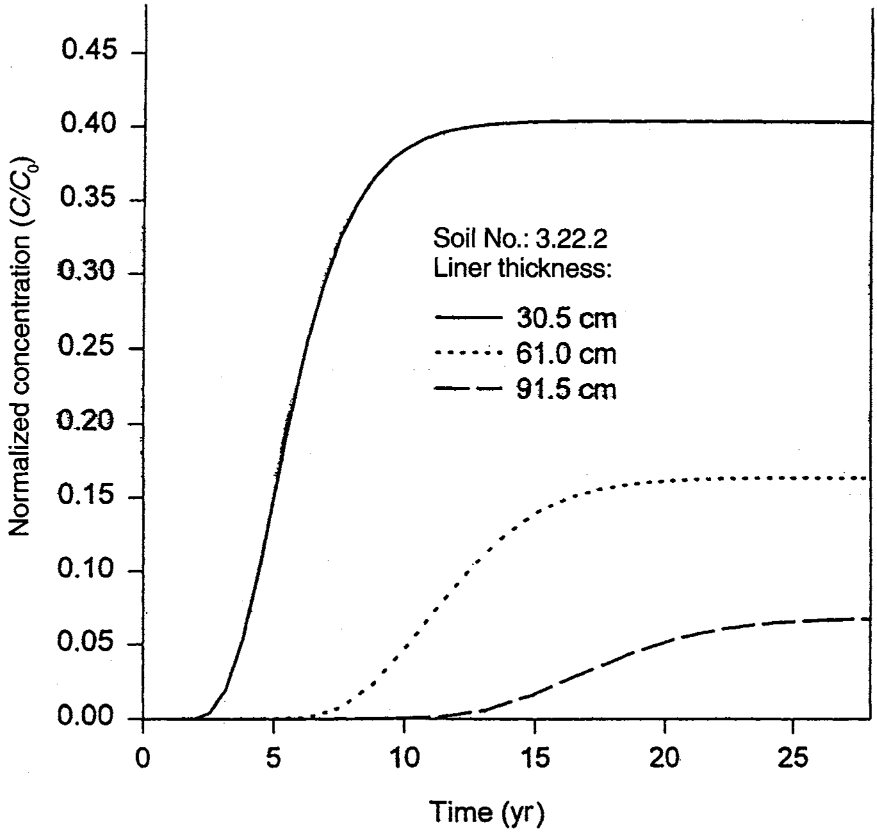


FIGURE 3-12. Breakthrough Curves of NH_4-N through Liners of Three Different Thicknesses.

3.4 NH_4-N MIGRATION BELOW LINER

The impact on groundwater quality of NH_4-N exiting the liner will depend on a host of hydrogeological parameters, such as the depth to the groundwater table, and on physicochemical and biological characteristics of the unsaturated soils underlying the lagoon liner. Although the leachate quantities are far below the Kansas-regulated limit of 0.64 cm/day (0.25 in./day), the high concentrations of NH_4-N (particularly in the case of swine facilities) may not be attenuated enough in the unsaturated domain to bring the concentrations at groundwater level within drinking water quality standards. An upper bound estimate of the impact of NH_4-N transport on groundwater quality could be obtained using numerical simulations with Soil Water Movement and Solute Transport in 2D (SWMS-2D) computer software.

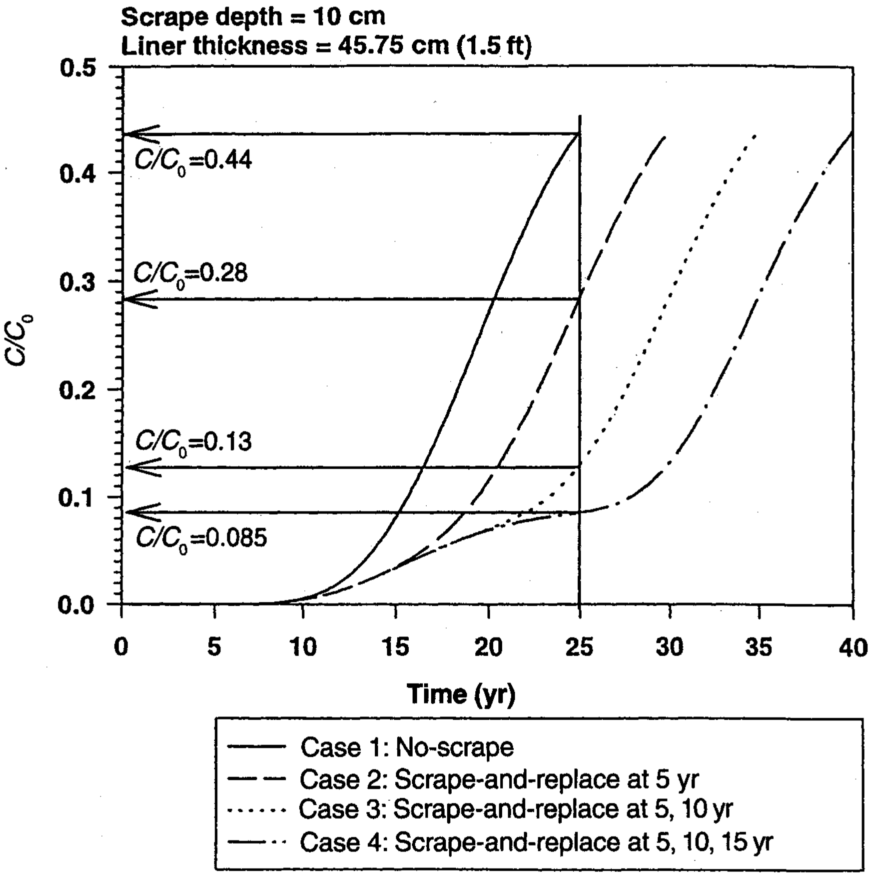


FIGURE 3-13. NH_4-N Breakthrough Curves at the Bottom of the Liner for Various Scrape-and-Replace Scenarios.

The SWMS-2D program is a finite element model (Simunek et al. 1994) developed originally for simulating two-dimensional water and solute movement in variably saturated media. The program solves the Richard's equation for saturated and unsaturated water flow and the Fickian-based advection-dispersion equation for solute transport. The transport equation includes provisions for linear equilibrium adsorption, zero-order production, and first-order degradation. The program may be used to analyze water and solute movement in unsaturated, partially saturated, or fully saturated porous media. The governing equations are solved using a Galerkin-type linear element method applied to a network of triangular elements. Integration in time is achieved using an implicit finite difference scheme for both saturated and unsaturated conditions. The resulting equations are solved in an iterative fashion by linearization.

The numerical model was used to simulate the NH_4-N transport in a two-layer system consisting of the lagoon liner and the underlying natural soils. Only an upper bound estimate of NH_4-N concentrations was sought in terms of the liner properties. For the purpose of this hypothetical simulation, a sandy soil was chosen conducive to faster transport (with a high saturated permeability of 7 m/day and longitudinal dispersivity of 450 cm^2/yr) of NH_4-N with no retardation and decay. The problem is idealized using the following assumptions:

1. The lagoon head is constant throughout the simulation period.
2. Percolation in the unsaturated zone underlying the liner begins under a constant suction head of 5 cm in the domain at time $t = 0$.
3. Initial NH_4-N concentration in the natural soil = 0.
4. The hydrodynamic dispersion-retardation factor and decay coefficients in the liner are constant throughout the simulation.
5. Pressure head in the lagoon liner varies linearly between the suction head of the natural soils at the bottom and the lagoon head at the top at time $t = 0$.
6. Mass transfer mechanisms, nitrification (conversion of NH_4^+ to NO_3^- in particular) are ignored in order to estimate the maximum possible NH_4 concentrations at groundwater table elevation.

The sensitivity analyses conducted using the numerical model have all shown predictable trends as the properties of the two layers are varied. The retarded and decayed transport of NH_4-N is demonstrated in comparison with the transport of a conservative tracer Cl in Figure 3-14. Although an increase in the operating head of the lagoon quickens the arrival of the contaminant at the groundwater table, its effect is not as significant as the effect of increasing the liner thickness. As seen in Figure 3-15, Cl breakthrough for a 0.9-m (3-ft) thick liner subjected to a load of 6.10 m (20 ft) is delayed by 10 yr when compared with the case of a 0.3-m (1-ft) liner subjected to an operating head of 3.05 m (10 ft).

The drastic effect of the engineering properties of liners is demonstrated in Figure 3-16. Sample 3.20.1 has a higher permeability and a lower decay coefficient compared with Sample 3.18.2. Accordingly, at field scale, liners corresponding to Sample 3.20.1 are associated with faster travel times and higher end concentrations, as shown in Figure 3-16. The permeabilities of both 3.18 and 3.20 are such that the seepage rates remain well within the state-stipulated limit of 0.64 cm/day (0.25 in./day) for a maximum operating head of 6.10 m (20 ft) and for liner thicknesses as low as 0.5 m (1.6 ft). However, when the mass transfer characteristics of NH_4-N in the liners are considered, it is obvious from Figure 3-16 that the engineering properties of liners play a larger role in the travel times and end concentrations than in seepage rates. A broad range in the normalized concentrations of 0.7 to 0.025 is possible with liners molded from the same soil type.

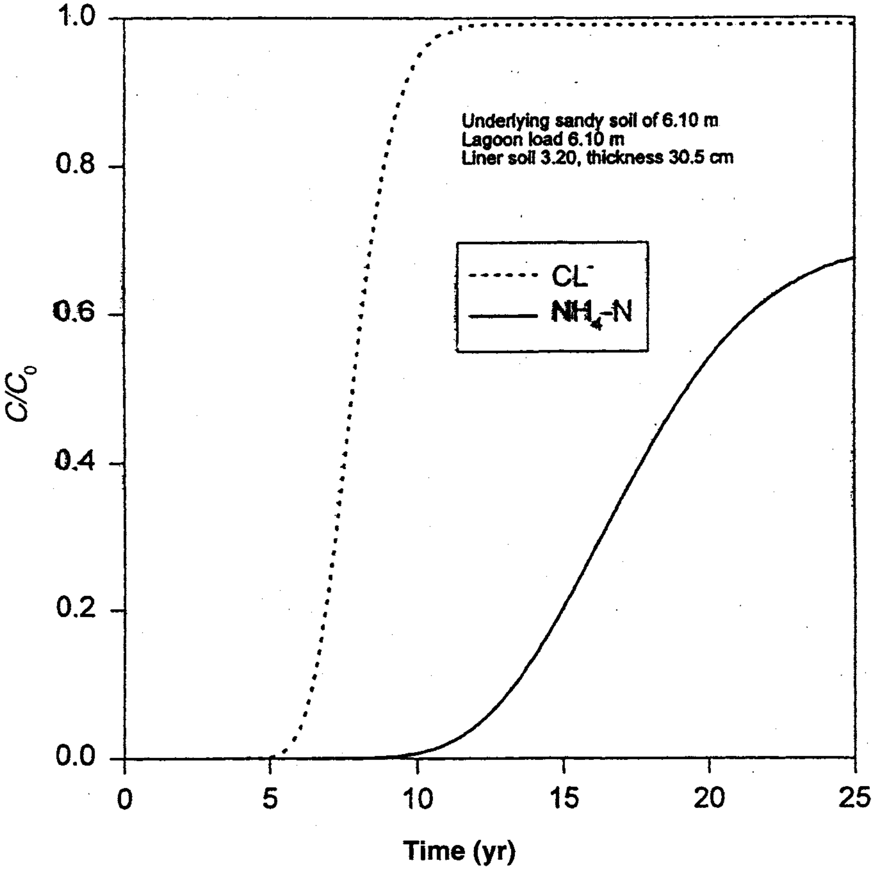


FIGURE 3-14. Comparison of Breakthrough Curves of Cl and NH_4-N Obtained for a Liner–Natural Soil System.

The effect of liner thickness on travel times and final concentrations of NH_4-N is equally significant. Figure 3-17 shows this effect for three thicknesses (0.15, 0.3, and 0.9 m) and for liner material 3.20. The concentrations of NH_4-N, normalized with respect to the influent concentration, have reduced from 0.9 to 0.1, and the travel times have increased from 5 to 65 yr, as the liner thickness is increased from 0.15 to 0.9 m. For clays with high CEC (leading to high retardation and saturation potential), providing thicker liners will yield much larger time periods for safe operation of anaerobic lagoons. The extent of possible retardation, decay, and saturation levels of NH_4-N in clay liners, observed in this study, suggests that properties such as CEC and microbial uptake, which influence mass transfer of NH_4-N, should be given an important consideration in designing liners for animal waste lagoons.

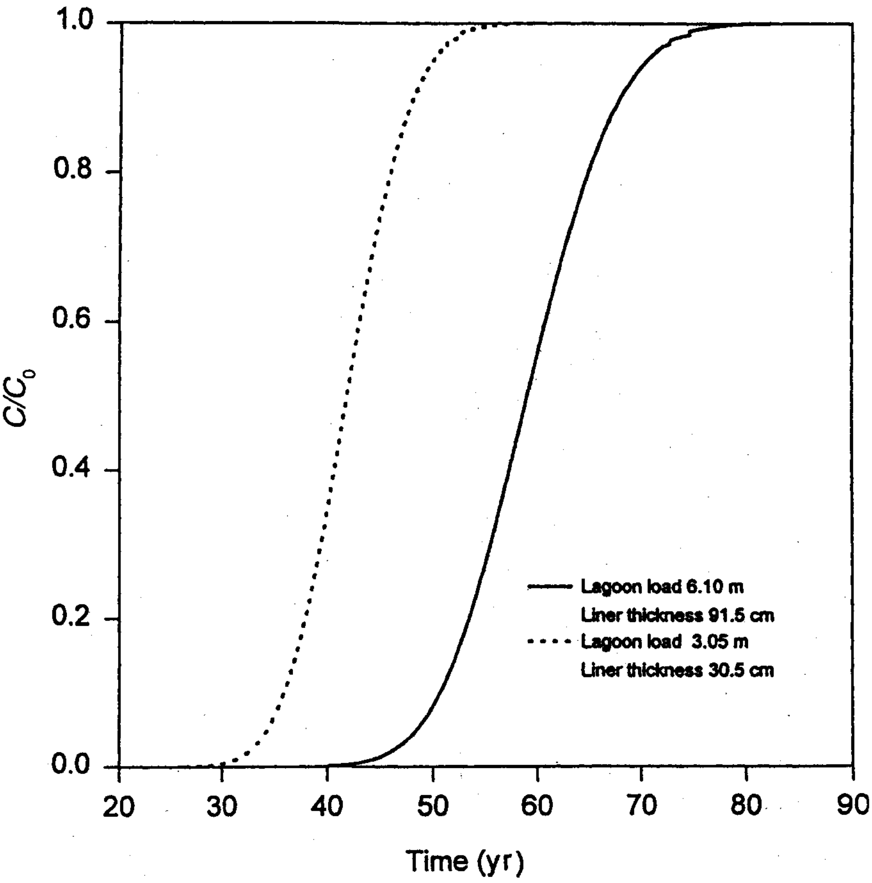


FIGURE 3-15. Effect of Lagoon Load and Liner Thickness on Cl Breakthrough Curve for a Liner–Natural Soil System.

Similar to the transport behavior through the liner, the migration of NH_4-N through the underlying soils will be impacted greatly with the liner maintenance practices. Consider again the case of a liner where the top 10 cm is scraped and replaced with a new layer of the same material at 5-yr intervals. At the end of 25 yr, the relative concentration (C/C_0) would be reduced from 0.15 to 0.05 as a result of this practice. As shown in Figure 3-18, doing more frequent scrape-and-replace further delays NH_4-N breakthrough.

3.5 CONCLUSIONS

In this chapter, the seepage and transport characteristics of lagoon liners are addressed using the soils and hydrogeology of southwest Kansas as examples. In general, the natural soils available in the region under

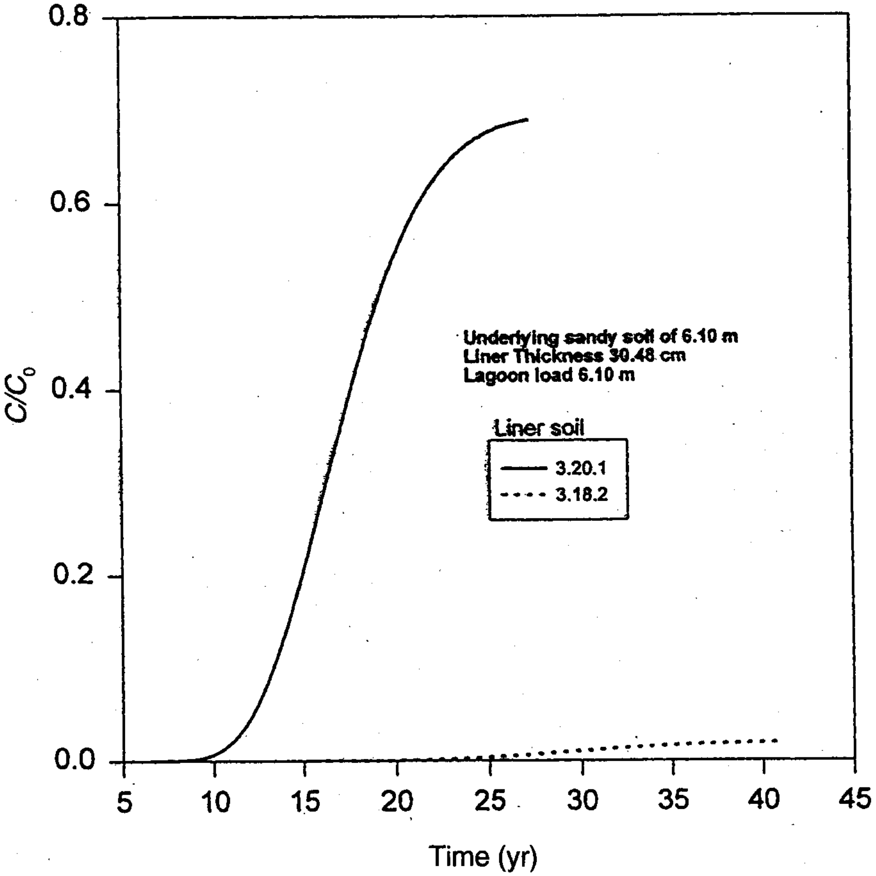


FIGURE 3-16. Effect of Liner Properties on NH_4-N Breakthrough for a Liner–Natural Soil System.

consideration were found to be capable of meeting the KDHE standard of 0.64 cm/day (0.25 in./day). Considering the side liners of lagoon facilities, which may offer no opportunity for particulate clogging or organic sludge formation, it may not be appropriate to assume reductions in permeability. Furthermore, the results indicate that biological clogging may not be prominent during the time period it takes for breakthrough of NH_4-N.

Transport of NH_4-N is associated with significant retardation, decay, and saturation rates in compacted clays. The results indicate significant differences in microbial uptake of NH_4-N among samples of the same soil type. Considering the beneficial effects of a high decay coefficient on groundwater quality, it may be useful for future studies to focus on the effects of soil structure on microbial colony formation and NH_4-N uptake by microorganisms. Potential for excessive sorption of ammonium in the

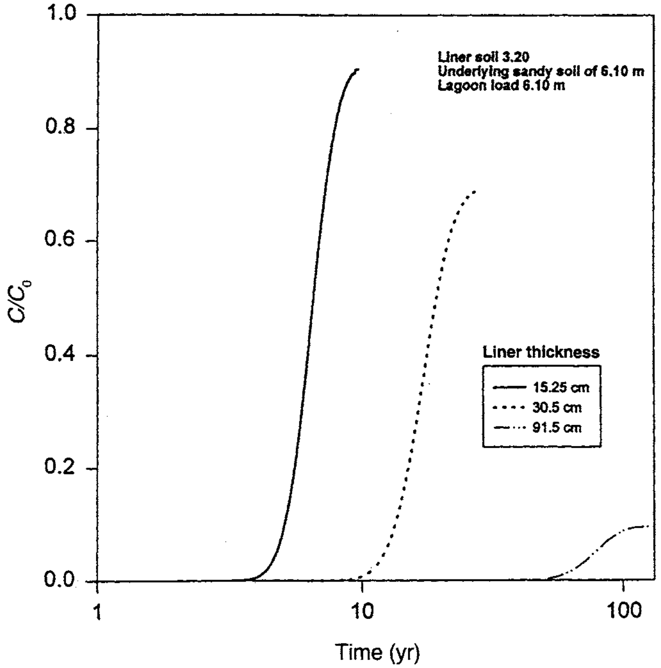


FIGURE 3-17. *Effect of Liner Thickness on NH_4-N Breakthrough for a Liner–Natural Soil System.*

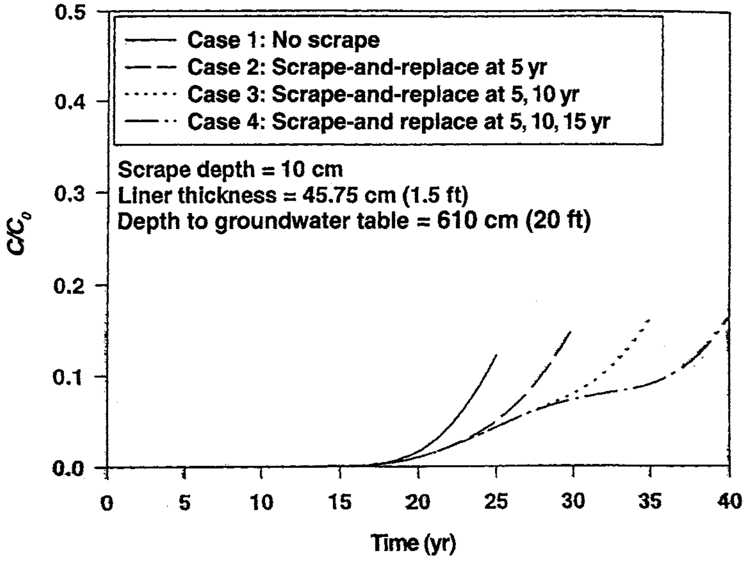


FIGURE 3-18. *NH_4-N Breakthrough Curves at the Groundwater Table for Various Scrape-and-Replace Scenarios.*

liner and its microbial uptake make the thickness of the liner an important variable. Although the seepage quantity may be less than the regulated limit for thinner liners, providing a thicker liner may significantly increase the time periods necessary for periodic cleanup of animal waste lagoon bottoms. In general, mass transfer characteristics of liner material, CEC and microbial uptake in particular, should be important considerations in the design of animal waste lagoon liners. Results also indicate that the ammonium concentrations are significantly reduced in the underlying soil profiles with frequent scrape-and-replace of the top portion of the liner. Higher scraping depth and frequent replacement of the top of the liner significantly reduce the ammonium concentrations leaching out of the liner.

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Chapter 4

CLOGGING OF ANIMAL WASTE LAGOON LINERS: AN OVERVIEW

Mohan V.S. Bonala and Lakshmi N. Reddi

Concentrated animal operations (e.g., dairy, beef, and poultry) produce significant amounts of waste that is rich in suspended solids, ammonium, nitrate, chloride, pathogens, and, to some extent, other chemicals such as potassium, sodium, and sulfates. This waste usually is stored in earthen lagoons that are lined either with a clayey type of material or with geosynthetic materials. The lagoon liner plays an important role in containing the contaminant transport into the subsurface. The extent of contamination caused by the presence of lagoons in any region depends on the subsurface soil properties, contaminant type, and depth of the groundwater table from the bottom of the lagoon. Presence of coarse-grained soils in the subsurface with high hydraulic conductivity is known to contaminate the groundwater to significant levels. Coastal regions, which have shallow water tables in coarse-grained soils, are more susceptible to contamination than any other region (Ritter et al. 1984; Huffman and Westerman 1995).

Seepage from animal waste lagoons is the topic of much debate. In the last 40 yrs, numerous laboratory and field investigations were conducted to quantify the seepage loss from animal waste lagoons. These previous investigations, conducted in different countries, are summarized in Table 4-1. Infiltration rates presented in Table 4-1 were obtained from a range of investigations comprising laboratory- and field-scale tests on different soil textures and using different animal wastes. The table shows that the decrease in infiltration rate ranges from 1 to 4 orders of magnitude. This decrease is attributed to physical, chemical, and biological interactions between the animal waste and the lagoon material. These three mechanisms can act either independently or in combination to clog the fine pores of the lagoon material. For example, solids in the animal waste clog the lagoon material pores physically; chemicals in the waste alter the orientation of soil particles; and the microorganisms and their excreta grow

TABLE 4-1. Summary from the Existing Literature on Clogging of Soil by Animal Waste

	Infiltration Rate		Waste			
Author	Initial (10^{-5} m/s)	Final (10^{-9} m/s)	Head (m)	Type	Soil Texture	Experimental Conditions
Hart and Turner 1965	0.9	620–30	2.2	Poultry, dairy, hog	Sand	Laboratory cores kept outside
Meyer et al. 1972; Oliver et al. 1974	0.7–2.48	12	3	Screened daily	Sand, clay loam	Lagoon
Davis et al. 1973	15	60	3	Screened daily	Clay loam, sand	Lagoon
Robinson 1973	1.3	35	1	Beef	Clay loam, silty clay	Lagoon
Chang et al. 1974	120–0.3	486–11	—	Dairy	Sand, loam, silty clay	Laboratory
Reddi 1998	0.003	1.3	6	Hog	Silty clay	Compacted wet of optimum (3-ft liner)
			NEW ZEALAND			
Hills 1976	0.4–0.8	10	2.5	Dairy	Sand to clay	Laboratory
			CANADA			
Lo 1977	1.16–0.06	3.5–7.75	2.4	Diluted dairy	Sand to clay	Laboratory
Rowsell et al. 1985	0.2	11	1	Beef	Sandy loam	Laboratory
	0.16–0.37	15	5			
	0.16–0.49	70–129	1		Loam	
	0.01–0.22	10–21	5			
	0.06–0.24	25–56	1		Clay	
	0.13–0.68	39–95	5			

within the pores. All of these mechanisms play a significant role in the clogging process. This chapter describes the research addressing these mechanisms.

4.1 PHYSICAL CLOGGING

Physical clogging or sealing occurs mainly due to the sedimentation of suspended solids in the liquid waste. Upon the initial flooding of the lagoon, gravity forces the undigested suspended solids to settle to the bottom of the lagoon, either forming a thin cake of settled solids or filling the pore spaces of the lagoon bottom soil material. The degree and rate of physical clogging depends on the amount of solids present in the liquid waste. Usually, the higher the suspended solids content in the liquid waste, the higher the degree of sealing. Baier et al. (1974) and Meyer et al. (1973) obtained infiltration rates in nine waste ponds using tensiometers placed at various depths in the soil under the lagoon. They observed that ponds containing high solids content in liquid waste had a sharp drop in infiltration rate during the first week of operation, whereas the ponds with a low solids content had rather steady decline in infiltration rate over a long period of time.

DeTar (1979) conducted infiltration studies on four different soils to observe the effect of total solids content on the infiltration rate. The effect of total solids content on the infiltration rate for four soils at 1 week is shown in Figure 4-1 (DeTar 1979). The figure shows that the infiltration rate is decreased as the total solids content is increased. The effect is similar for all the soil materials studied. The amount of total solids content depends on the type of animal waste. Table 4-2 shows the liquid animal waste characteristics for different animal waste types.

The degree and rate of sealing also depends on the type of the lagoon bottom soil. If the lagoon bottom is composed of coarse-grained soils, the lagoon may need longer time to clog. On the other hand, if the lagoon bottom is composed of fine-grained soils, the lagoon may clog in a relatively short period of time. Rowsell et al. (1985) conducted laboratory infiltration studies on loam, sandy loam, and clay soils using cattle manure at 1-m and 5-m hydraulic heads. Infiltration rate versus time is plotted in Figure 4-2 for 5-m hydraulic head; the infiltration rates correspond to 10 min, 1 day, and 30 days. It is evident from this figure that the infiltration rate is decreased by at least one order of magnitude by the end of 30 days. It was concluded that the reduction in infiltration rate is due to the blocking of soil pores by the solids in the waste. To further understand the location of clogging, the cross-sectioned soil samples were observed with reflected-light microscopy after completion of infiltration experiments. A distinct top layer in the sandy loam soil, which is rich in organic matter,

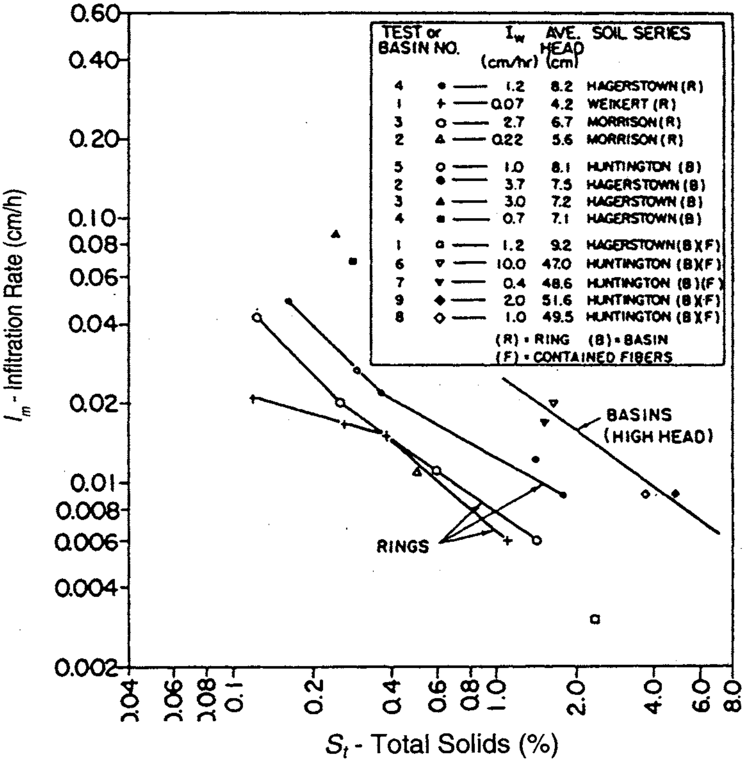


FIGURE 4-1. Infiltration Rates versus Concentration of Liquid Manure for Various Soil Permeabilities at 1 Week (Source: DeTar 1979) (Reprinted by Permission of American Society of Agricultural Engineers).

was observed. They hypothesized that waste particles could have entered the sandy loam soil because of the large pore openings, and they concluded that the soil pore sizes could have a significant effect on the blocking of the pores. Barrington (1987a) supported the conclusion that physical mechanisms play a major role in the clogging process.

4.2 CHEMICAL CLOGGING

Chemical sealing depends on the presence of Na^+ and K^+ ions in the liquid waste and on the type of soil. These ions usually influence the electrical charges of the soil particles and cause them to disperse. In general, soils with high cation exchange capacity (CEC) are more susceptible to chemical reactions. Bentonite, with a very high CEC, can adsorb significant amounts of water and expand, which is very effective in reducing the permeability. This could be a good chemical additive to alter the permeability of soil materials.

TABLE 4-2. Livestock Manure Characteristics

Item	Poultry	Cattle	Swine
Total solids (TS) in feces and urine:*			
(lb/day/animal)	0.066	10.44	0.795
(g/day/animal)	30.0	4,730	360
Volatile solids (VS) (% of TS)	77.5	80.3	78.5
5-day BOD (mg BOD/mg VS)	0.288	0.183	0.320
COD (mg COD/mg VS)	1.11	1.0	1.2
Total nitrogen, as N (% of TS)	5.4	3.7	4.0
NH_4 nitrogen (% of total N)	74	—	75
Phosphorus, as P_2O_5 (% of TS)	4.6	1.1	3.1
Potassium, as K (% of TS)	2.1	3.0	1.4
Volatile acids (as acetic) (% of TS)	5.8	3.2	4.8

Source: Hart and Turner 1965. (Reprinted by permission of Water Environment Federation.)
BOD = Biological oxygen demand; COD = Chemical oxygen demand.
Note:
*Quantity depends on the size of the animal; the above figures are for a 5-lb (2.2-kg) chicken (usual egg-laying size), a 1,000-lb (454-kg) bovine (small dairy cow or marketable beef animal), and a 100-lb (45-kg) swine (half-grown to market size).

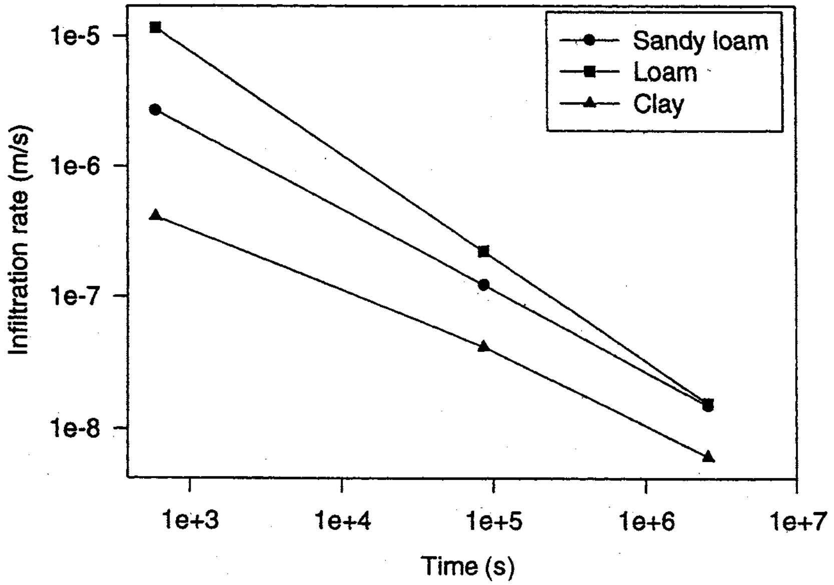


FIGURE 4-2. Infiltration-Time Relationship for Infiltration of Liquid Manure into Three Different Soils at 5-m Head (Source: Rowsell et al. 1985).

Sokolovski (1940) produced a perfect sealing effect by dispersing a layer of clay for a canal liner using NaCl solution. Matthew and Harms (1969) conducted infiltration studies (which did not involve waste) and indicated that dispersion of soil particles by Na^+ affects infiltration rates in soils. They concluded that the higher the sodium adsorption ratio (SAR) of a soil, the lower the permeability. Travis et al. (1971) reported that infiltration of a lagoon-water simulant caused sealing in a silty clay loam soil (300 g/kg clay, 500 g/kg silt), presumably by dispersion, but not in a loam soil (200 g/kg clay, 400 g/kg silt).

Hills (1976) reported that chemical effects were important in soils with more than 75 g/kg of clay and 305 g/kg silt. Mirtskhylara et al. (1972) claimed that a temperature greater than 15°C is required for the chemical mechanisms to be effective. Curry and Beasly (1962) conducted laboratory studies to investigate the decrease in the permeability of carborundum porous media using bentonite particle suspensions in distilled water. They concluded that the zeta potential of the carborundum could be correlated to the degree of sealing. Chang et al. (1974) concluded that chemical effects on soil clogging were insignificant when compared to the biological effects in reducing the hydraulic conductivity of the soils.

4.3 BIOLOGICAL CLOGGING

Though the initial sealing of the lagoon may be caused by physical clogging of the soil pores by waste solids, biological activity within the soil pores can contribute significantly to the long-term sealing. Biological sealing is usually associated with aerobic or anaerobic conditions that cause bacterial growth (Winterer 1922; McCalla 1945; Allison 1947; Bendixon et al. 1962; Preul 1968). Because of the growth of microorganisms, the soil pore diameters reduce, which in turn lowers the infiltration rate (Rice 1974). Biological sealing also occurs as a function of waste solids' biodegradability and ambient temperature. In the absence of oxygen and when a good source is present, anaerobic microorganisms multiply rapidly and fill the soil voids. These bacteria excrete polysaccharides and polyuronides (by-products of metabolism) in the form of waxes and jells which can seal the voids very effectively. Davis et al. (1973) measured the infiltration rate from a lagoon located on a sandy loam soil and concluded that the primary mechanism of sealing was due to biological activity and the presence of high organic carbon.

Chang et al. (1974) concluded that the initial decrease in hydraulic conductivity is due to the physical blocking of the pores of the soil by the suspended solids in the waste, and by the excreta of the microorganisms in the soil pores. Results from this study are shown in Table 4-3, which presents the polysaccharides in four soil types at different depths for different

TABLE 4-3. Polysaccharides (mg of $C_6H_{12}O_6$/g of dried soil) in Recovered Soil Columns

		Days Underneath the Wastewater Pond					
Soil type	Depth (in.)	0	3	7	17	29	64
Sandy	0–2	1.25	1.45	1.65	1.65	1.70	2.10
	2–4	1.10	1.15	1.10	1.25	1.50	2.70
	4–6	1.05	0.85	0.95	1.15	1.45	1.30
	6–8	1.25	1.25	0.85	0.75	1.25	1.20
	8–10	1.15	1.10	1.10	1.10	1.20	1.40
Loamy	0–2	1.1	1.10	1.10	1.25	1.20	3.40
	2–4	1.1	1.60	1.15	1.15	1.10	2.90
	4–6	1.45	0.95	1.15	1.05	1.40	2.80
	6–8	1.25	1.00	1.20	1.25	1.70	1.00
	8–10	1.05	1.25	1.40	1.20	1.20	1.00
Clay	0–2	1.70	1.90	2.00	2.25	2.50	3.00
	2–4	1.65	1.90	1.85	2.10	2.15	2.80
	4–6	2.05	1.80	1.70	2.25	1.95	2.20
	6–8	1.90	1.90	1.70	1.45	1.65	1.50
	8–10	1.90	1.95	1.80	1.40	1.15	1.40
Sand	0–2	0.24	0.25	0.22	0.30	0.41	0.30
	2–4	0.10	0.11	0.16	0.29	0.28	0.40
	4–6	0.02	0.02	0.10	0.26	0.30	0.40
	6–8	0.002	0.03	0.09	0.15	0.23	0.30
	8–10	0.006	0.05	0.03	0.05	0.20	0.20

Source: Chang et al. 1974. (Reprinted by permission of Water Environment Federation.)

times. The table shows that, for any given soil at any depth, the amount of polysaccharides content is increasing with time. Barrington et al. (1987b) claim that biological sealing augments the physical sealing process. Contradicting some of these studies, Hart and Turner (1965) concluded that the biological sealing of the lagoon did not appear to be very effective in their studies on infiltration of liquid waste in a sandy loam lagoon.

Biological clogging mechanisms can be conceptually explained using bacterial growth kinetics (Figure 4-3). The initial phase of the growth cycle, known as the lag phase, shows no detectable increase in population size, and so the rate of population increase (specific growth rate) is zero. However, during this period, the cells manufacture enzymes and other chemicals necessary to adapt their metabolism to the new conditions. During this phase the individual cells grow larger. The acceleration phase marks the beginning of a gradual increase in the specific growth rate.

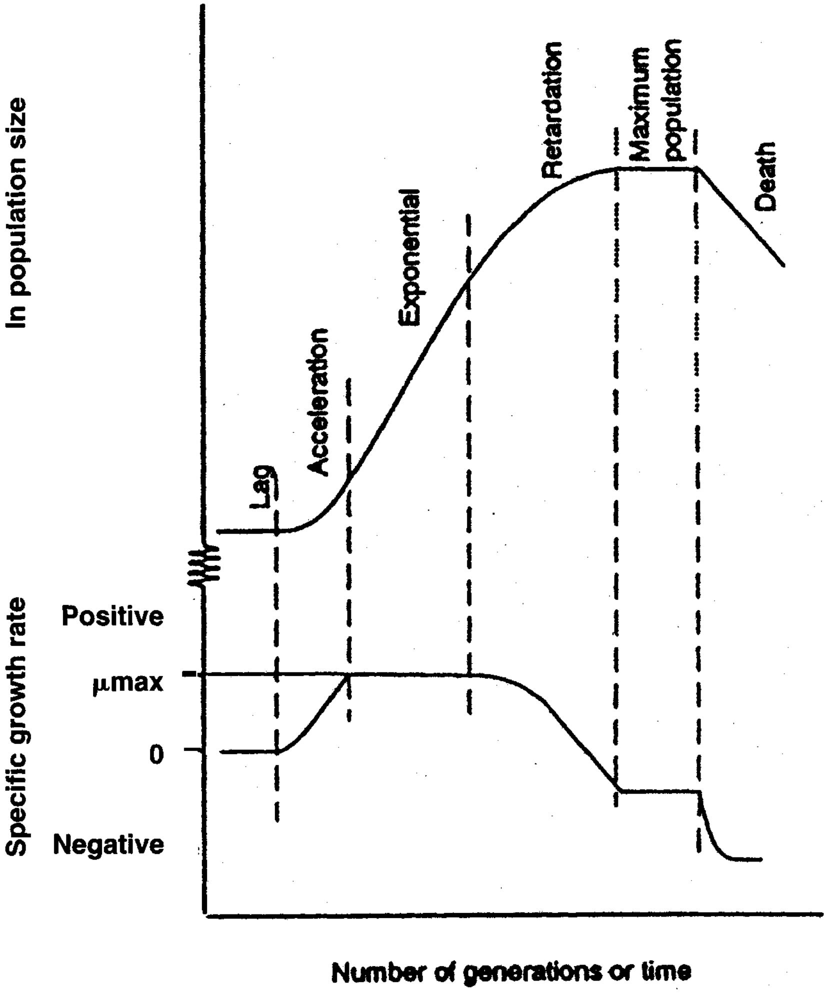


FIGURE 4-3. Various Phases in a Biological Growth Cycle in a Limited Environment.

During the exponential phase, the population grows exponentially and may be described by the basic growth equation (Lynch and Poole 1979):

$$\frac{dp}{dt} = Rp\left(1 - \frac{p}{p_m}\right) \tag{4-1}$$

where

p = population size
R = maximum growth rate (T^{-1})
p_m = population size in the maximum population phase

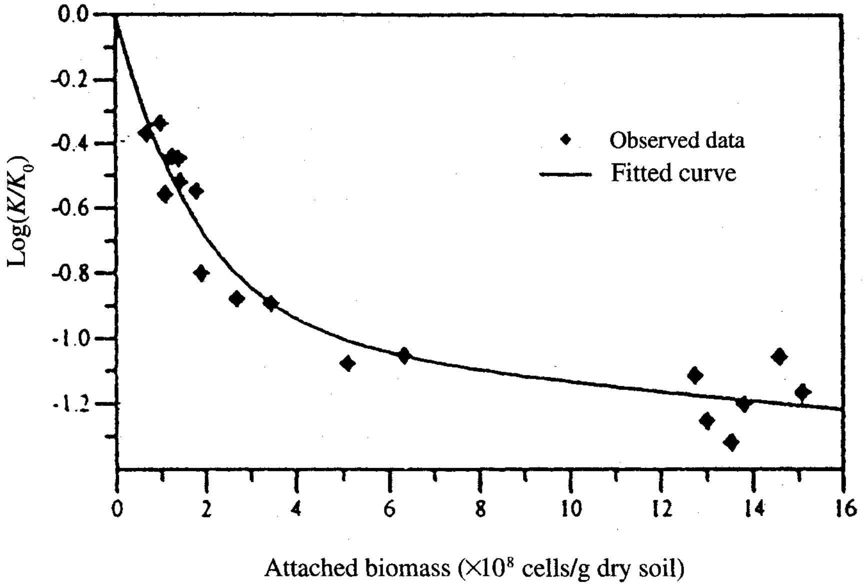


FIGURE 4-4. Relationship between the Logarithmic Ratio of Hydraulic Conductivity and the Content of Attached Biomass (Source: Wu, J., Gui, S., Stahl, P., and Zhang, R. (1997). "Experimental study on the reduction of soil hydraulic conductivity by enhanced biomass growth." *Soil Science,* 162(10): 741–748*) (Reprinted by Permission of Lippincott Williams & Wilkins).*

The retardation phase is established once the specific growth rate begins to decline. In the maximum population phase, the population is metabolically active, but active growth stops. During the death phase, the number of microorganisms declines.

The influence of biomass growth on hydraulic conductivity has been studied extensively (Cunningham et al. 1991; Vandevivere and Baveye 1992; James et al. 1995; Turner 1995). In a recent study, Wu et al. (1997) conducted experiments on soil columns filled with river sand to observe the hydraulic conductivity reduction due to the biomass growth. They supplied dextrose-nutrient solution to stimulate the growth of microbial colonies. The variation of normalized hydraulic conductivity with biomass growth is shown in Figure 4-4. The best-fit curve for the experimental data was expressed as

$$\log_{10}(K/K_0) = e^{-0.538S_b} - 1 - 0.0137S_b \tag{4-2}$$

where S_b is the attached biomass content (10^8 cells per gram of dry soil). They concluded that the significant reduction in hydraulic conductivity is solely due to biomass growth.

4.4 AMMONIUM TRANSPORT SIMULATIONS USING SWMS-2D

In spite of the clogging of the liner material, contaminants will seep into the subsurface to some extent. The chemicals in the waste, including chloride and ammonium, pose serious health hazards to humans. Ammonium—a positive ion—can adsorb onto the negatively charged soil particles. In the presence of oxygen (i.e., in unsaturated conditions, ammonium transforms into nitrate, which, being a cation, can easily transport in the subsurface. To study the impact of clogging on ammonium transport in the subsurface, ammonium transport is modeled.

In addition, with continuous use of a lagoon, significant amounts of settled solids occupy a large volume of it. To reclaim the lost volume, the lagoon is frequently emptied. The settled solids are rich in nutrients and can be applied to fields as organic fertilizers. The top portion of the lagoon liner also is rich in nutrients and can be scraped and used as fertilizer as well. To understand the impact of these actions on ammonium transport in the subsurface, the following scenarios are simulated:

1. Hydraulic conductivity reduction of the liner material due to clogging
2. Scraping the lagoon liner material to reclaim the nutrients
3. Scraping and replacing the top few centimeters of lagoon liner material

The Soil Water Movement and Solute Transport in 2D (SWMS-2D) program is described in Section 3.4. The numerical model was used to simulate the NH_4-N transport in a two-layer system composed of the lagoon liner and the underlying natural soils. For the purpose of the simulations, the properties of southwestern Kansas soils were chosen for liner material, as native soils are used typically to construct liners in that region. Hobson (1991) reported that more than two-thirds of the clay is montmorillonite in Kansas soil. Montmorillonite, with a CEC of 80 to 150 meq/100 g of soil and a specific-surface of 700 to 840 m^2/g (Mitchell 1993), would offer a significant number of adsorption sites for ammonium adsorption. A sandy soil conducive to faster transport of NH_4-N with virtually no retardation and decay was chosen as the underlying unsaturated soil medium. The hydraulic properties and transport characteristics of NH_4-N in the liner and the underlying subsurface soils are adopted from a previous study (Reddi and Davalos 2000) and are presented in Table 4-4. Reddi and Davalos (2000) conducted permeation studies on Kansas soils using dairy and swine waste as permeates for leachate quality and quantity. They obtained the ammonium transport parameters (longitudinal dispersivity, retardation factor, and decay coefficient) by fitting an analytical solution to the effluent ammonium concentrations obtained from the experiments. Other parameters (θ_r, θ_s, α, and n) in Table 4-4 were chosen from the SWMS-2D database.

TABLE 4-4. Hydraulic Properties and Transport Characteristics of NH_4-N in Southwest Kansas (Lagoon Liner) and Sandy (Natural Subsurface) Soils

Property	Soil Type: Southwest Kansas Soils as Liner Materials	Soil Type: Sandy Subsurface Soil
Longitudinal dispersivity, D_h (cm^2/yr)	116	18,250
Retardation factor, R_d	5.6	0
Decay coefficient (yr^{-1})	0.2	0
Saturated hydraulic conductivity, K_s (m/day)	1.12E-5	7.128
Residual water content, θ_r	0.02	0.045
Saturated water content, θ_s	0.4	0.43
α*	.41	1.45
n*	1.964	2.68

Source: Reddi and Davalos 2000.
Note:
*Parameters required for calculating the unsaturated hydraulic conductivity of the soil using van Genuchten expression.

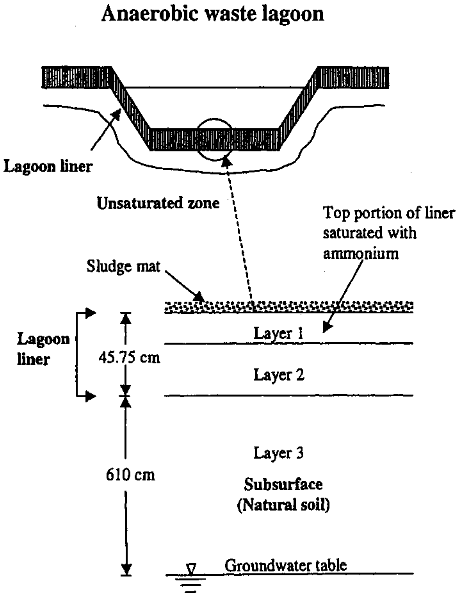
The problem is idealized using assumptions 1 through 6 listed in Section 3.4, as well as the following two additional assumptions:

1. Scraped liner material is replaced immediately, and the freshly applied liner binds uniformly with the existing liner.
2. Secondary porosity features, such as fissures and desiccation cracks, are not present in the lagoon liner.

Figure 4-5 shows the schematic of the anaerobic animal waste lagoon with a liner and the three simulation scenarios studied here. The liner is assumed to be constructed with naturally available Kansas soils represented by the properties in Table 4-4. The three simulation scenarios and the results are discussed in the following sections.

4.4.1 Scenario 1: Hydraulic Conductivity Reduction of the Liner due to Clogging

To simulate this scenario, Phase 1 is carried until the liner is clogged. In Phase 2 simulation, the initial conditions for layers 1 and 2 are taken from the final conditions from Phase 1 simulation, and the saturated hydraulic conductivity of the liner (layers 1 and 2) is reduced to reflect the effect of



Scenario 1: Hydraulic conductivity reduction of the liner

Phase 1
Initial conditions: All layers are clean

Phase 2
Initial conditions:
Layers 1&2: Final conditions from Phase 1 run with reduced hydraulic conductivity
Layer 3: Final conditions from Phase 1 run

Scenario 2: Scraping only

Phase 1
Initial conditions: All layers are clean

Phase 2
Initial conditions:
Layer 1: Scraped
Layers 2&3: Final conditions from Phase 1 run

Scenario 3: Scrape-and-replace

Phase 1
Initial conditions: All layers are clean

Phase 2
Initial conditions:
Layer 1: Clean liner material
Layers 2&3: Final conditions from Phase 1 run

FIGURE 4-5. Anaerobic Animal Waste Lagoon with the Modeling Simulation Scenarios (Source: Bonala and Reddi 2000).

clogging. The initial conditions for layer 3 are taken from the final conditions from Phase 1 simulation.

Figure 4-6 shows the relative NH_4-N concentration profiles in the subsurface at the end of 25 yr. In this scenario, for demonstration purposes the saturated hydraulic conductivity of the liner was chosen to be 1.12e-4 m/day. Case 1 in Figure 4-6 refers to a scenario with no clogging, and hence the hydraulic conductivity was kept constant throughout the simulation period. Cases 2, 3, and 4 refer to the clogging scenarios with hydraulic conductivities reduced 10, 100, and 1,000 times, respectively, at the end of 120 days of simulation period. From Figure 4-6 it can be observed that, at any given depth in the subsurface, the concentration of NH_4-N is lower for lower hydraulic conductivities of the liner. In Cases 3 and 4, NH_4-N leaching out of the liner is almost zero.

Figure 4-7 shows the breakthrough curves at the bottom of the liner for Cases 1, 2, and 3 as described in Figure 4-6. Complete NH_4-N breakthrough occurred at about 2.5 yr of simulation in Case 1. In Cases 2 and 3, the NH_4-N concentrations are 0.48 C_0 and almost zero, respectively, at the end of 25 yr. From Case 3 it can be concluded that, if the lagoon liner hydraulic conductivity reduces by two orders of magnitude in 120 days, then the ammonium

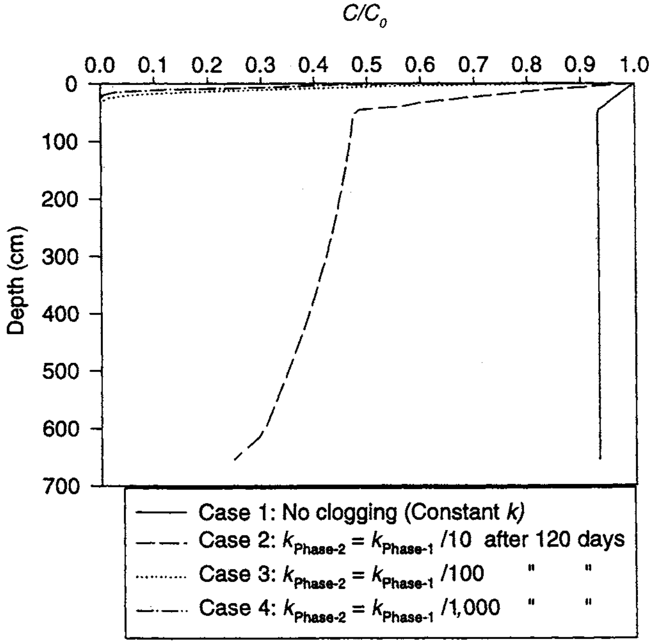


FIGURE 4-6. NH_4-N Concentration Profiles at 25 yr for Different Reduced Hydraulic Conductivities (Source: Bonala and Reddi 2000).

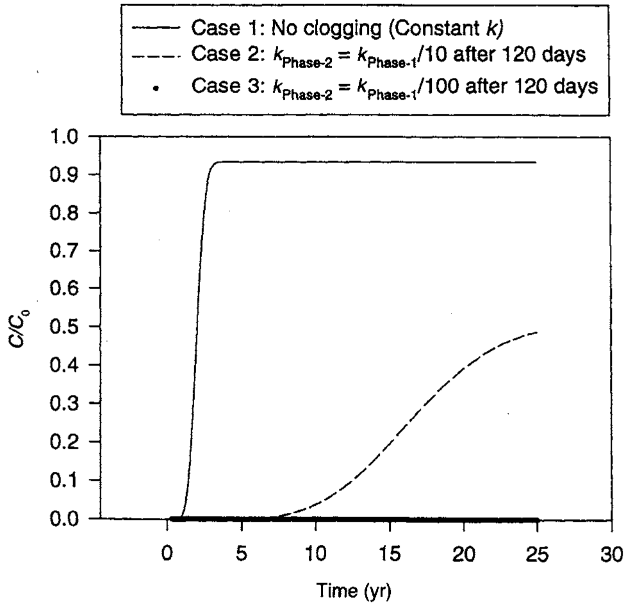


FIGURE 4-7. NH_4-N Breakthrough Curves at the Bottom of the Liner for Different Reduced Hydraulic Conductivities (Source: Bonala and Reddi 2000).

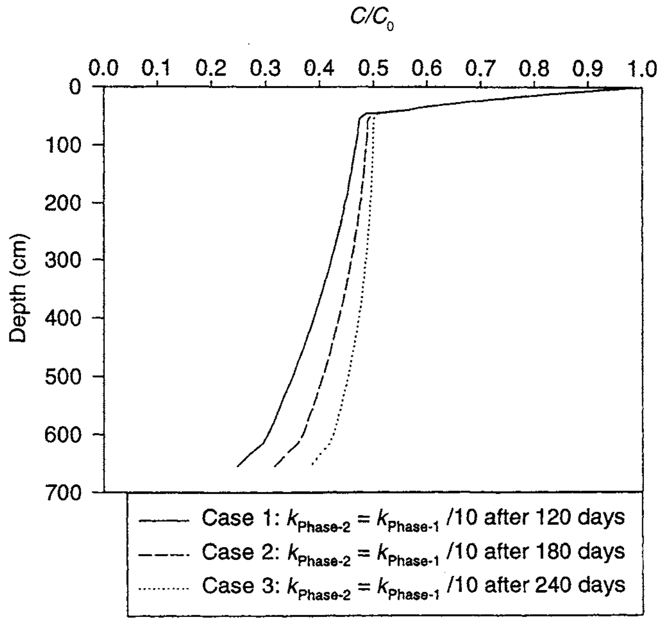


FIGURE 4-8. NH_4-N Concentration Profiles at 25 yr for Different Clogging Times (Source: Bonala and Reddi 2000).

concentrations in the subsurface are close to zero. This conclusion is based on the simulation results only; other important factors (e.g., poor liner construction quality and control, desiccation cracks, and animal burrows) may increase the NH_4-N concentrations in the subsurface.

To study the effect of the time it takes for clogging to occur, simulations were carried out for 120 days, 180 days, and 240 days of clogging times. Figure 4-8 shows the concentration profiles in the subsurface for these clogging times assuming that the hydraulic conductivity reduced by one order. At any given depth, higher NH_4-N concentrations are observed when the clogging time is higher. Figure 4-9 shows the breakthrough curves at the bottom of the liner for the same simulation cases. As the clogging time is doubled from 120 days to 240 days, the concentrations leaching from the liner are increased from 0.2 C_0 to 0.35 C_0. However, the effect of clogging time appears to be not as significant as the magnitude of hydraulic conductivity reduction.

4.4.2 Scenario 2: Scraping the Top Portion of the Lagoon Liner

In this scenario, Phase 1 simulations are conducted until the top portion of the liner is saturated with ammonium (NH_4-N). The saturated liner portion is scraped at the end of this phase. In Phase 2 simulations,

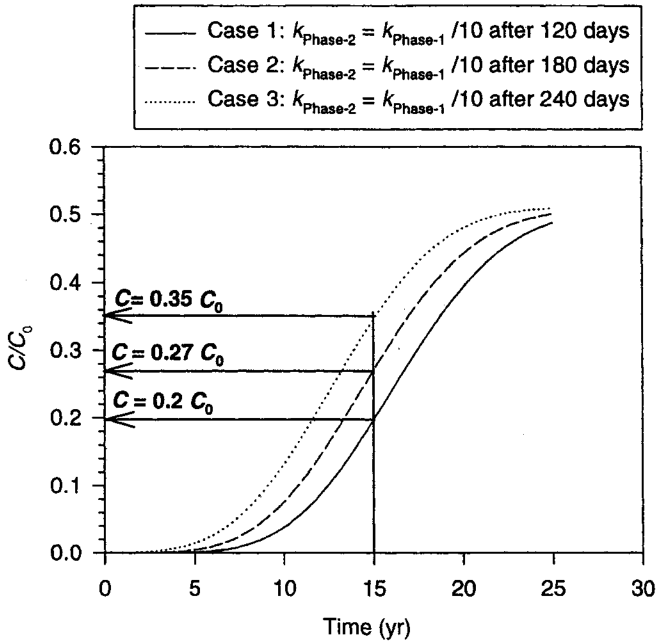


FIGURE 4-9. NH_4-N Breakthrough Curves at the Bottom of the Liner for Different Clogging Times (Source: Bonala and Reddi 2000).

the initial conditions for layers 2 and 3 are taken from the final conditions from Phase 1 simulations. In this study, simulations are carried out only for a scraping depth of 10 cm to demonstrate the effect of scraping.

Figure 4-10 shows the relative NH_4-N concentration profiles in the lagoon liner and in the natural soil at the end of 25 yr. Case 1 in Figure 4-10 refers to the concentration profile of NH_4-N when the liner is not scraped for the entire period of lagoon operation. Cases 2 and 3 represent the relative NH_4-N profiles when the liner is scraped at 5 yr and at 5 and 10 yr, respectively. Figure 4-10 shows that, at any given depth, the concentration of NH_4-N in the subsurface is higher in Cases 2 and 3 than in Case 1. The increase in NH_4-N concentrations in the subsurface in this scenario is clearly due to the decrease in the liner thickness.

Figure 4-11 shows the NH_4-N breakthrough curves at the bottom of the liner for the same cases presented in Figure 4-10. Figure 4-11 shows that Case 1 has not reached the complete breakthrough of NH_4-N at the end of 25 years. Also, at any given time the NH_4-N concentrations are smaller in Case 1 than in Cases 2 and 3. For example, NH_4-N concentrations leaching out of the liner are 0.08 C_0, 0.52 C_0, and 0.78 C_0 for Cases 1, 2, and 3, respectively, at the end of 15 yr. That is, the concentration of NH_4-N leaching out of the liner in Case 3 is about 10 times higher than the concentration leaching out of the liner in Case 1. From Figures 4-10 and 4-11 it can be concluded that

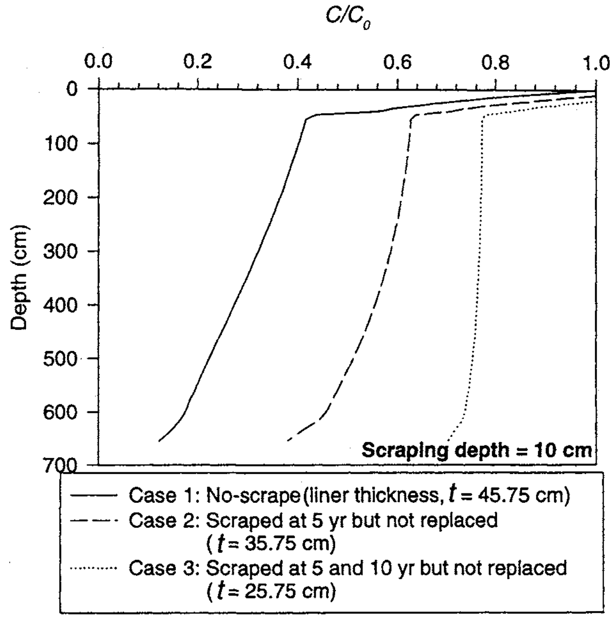


FIGURE 4-10. NH_4-N Concentration Profiles at 25 yr with and without Scraping the Lagoon Liner (Source: Bonala and Reddi 2000).

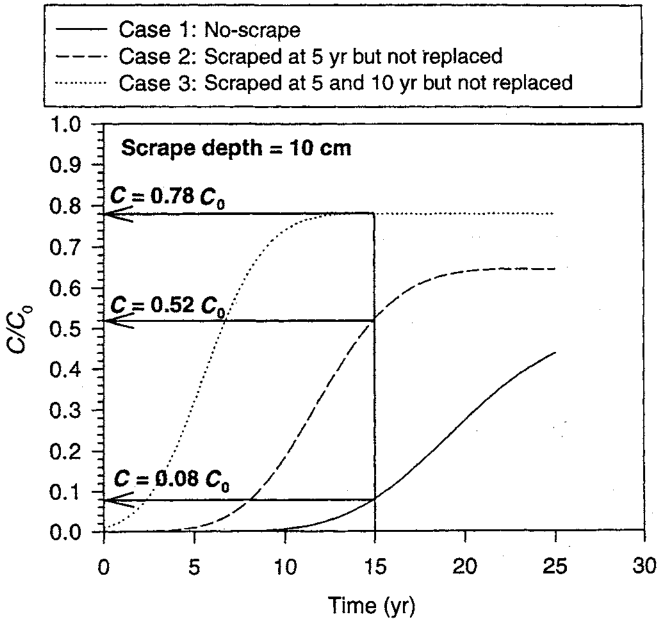


FIGURE 4-11. NH_4-N Breakthrough Curves at the Bottom of the Liner with and without Scraping the Lagoon Liner (Source: Bonala and Reddi 2000).

scraping the lagoon liner will only increase the amount of NH_4-N leaching out of the liner bottom into the subsurface. Therefore, extreme care must be taken while reclaiming the lagoon volume to keep the same liner thickness.

4.4.3 Scenario 3: Scrape-and-Replace the Top Portion of the Lagoon Liner

Scraping the top portion of the liner and replacing it with a fresh layer of liner material can retard the NH_4-N movement in the liner significantly, because the clean liner material has a large number of adsorption sites for NH_4-N. In this scenario, Phase 1 simulations are conducted until the top portion of the liner is saturated with ammonium. The saturated liner portion is scraped and replaced with fresh liner material at the end of this phase. In Phase 2 simulations, the initial conditions for layers 2 and 3 are taken from the final conditions from Phase 1 simulations. The initial conditions for layer 1 are reset to represent clean layers, as the replaced liner material is fresh.

Figure 4-12 shows the relative NH_4-N concentration profiles after a simulation period of 25 yr for a scrape-and-replace depth of 10 cm. Case 1 in Figure 4-12 refers to the relative NH_4-N concentration profile in the subsurface without scraping. Case 2 refers to the relative NH_4-N concen-

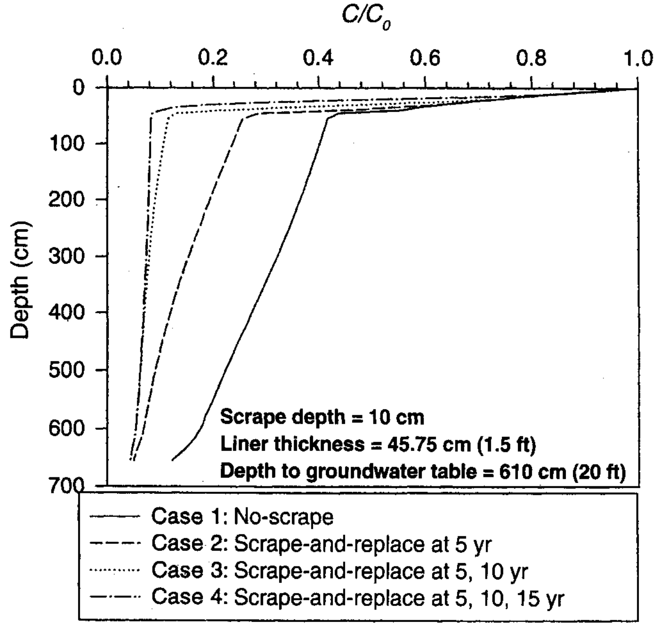


FIGURE 4-12. NH_4-N Concentration Profiles at 25 yr with and without Scrape-and-Replace of the Lagoon Liner (Source: Bonala and Reddi 2000).

tration profile after a single scrape-and-replace at the end of 5 yr. Cases 3 and 4 refer to the scrape-and-replace at the end of 5 and 10 yr, and 5, 10, and 15 yr, respectively. Figure 4-12 shows that the ammonium concentrations in the subsurface are much lower when the scrape-and-replace process continued periodically at the end of every 5 yr. Ammonium concentrations at 655-cm depth are 0.12 C_0 for Case 1 and 0.06 C_0 for the scraped cases. An important observation from Cases 3 and 4 is that the concentration of ammonium leaching out of the liner is very minimal. This suggests that frequent scrape-and-replace guarantees lower ammonium concentrations in the subsurface in addition to conserving the rich nutrients and fertilizing resources.

Figure 4-13 shows the breakthrough curves of NH_4-N at the bottom of the liner (45.75 cm). The relative concentrations leaching out of the liner are 0.44, 0.28, 0.13, and 0.085 for Cases 1, 2, 3, and 4, respectively. Scrape-and-replace at the end of every 5-yr period (Case 4) has reduced the contaminants leaching out of the system by about five times when compared to no-scrape (Case 1). The important observation to be made from this graph is that, as long as the scrape-and-replace continues at

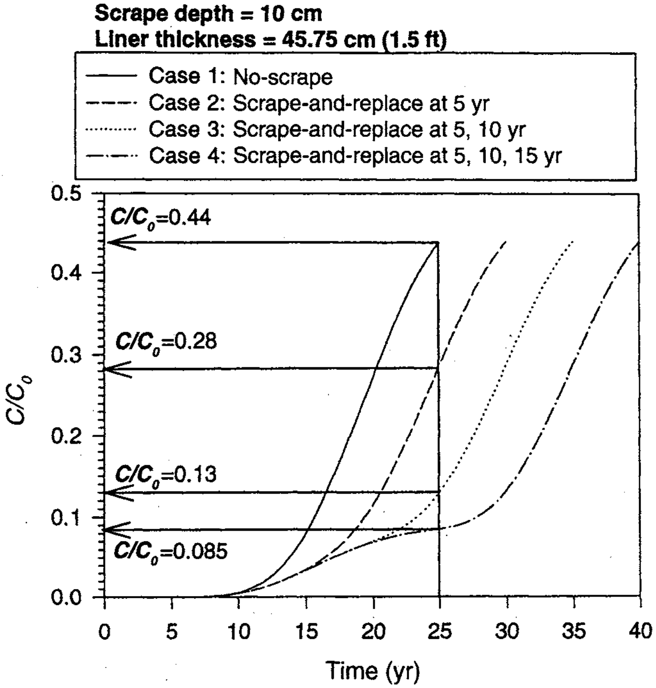


FIGURE 4-13. NH_4-N Breakthrough Curves at the Bottom of the Liner with and without Scrape-and-Replace of the Lagoon Liner (Source: Bonala and Reddi 2000).

regular intervals of time, the relative concentrations are maintained at a low and leveled trend (Case 4). The breakthrough curve shows a steep increase in C/C_0 after the scrape-and-replace is stopped. The presence of montmorillonite and illite in the native Kansas soils, with high CEC and specific-surface, retarded the NH_4-N transport significantly into the subsurface. If the liners are not scraped and replaced at regular intervals of time, then the available cation exchange locations on the soil would have been saturated and the ammonium breakthrough would occur at much earlier times.

To study the effect of scraping depth on ammonium concentrations leaking out of the liner, simulations also were carried out for 20-cm scrape-and-replace liner depth. Figure 4-14 shows that NH_4-N concentrations coming out of the liner are close to zero for a periodic scrape-and-replace depth of 20 cm. Figure 4-15 shows the concentration profiles for no-scrape and for scrape-and-replace depths of 10 cm and 20 cm. The end concentrations mixing with the groundwater with the 20-cm scrape-and-replace depth are reduced by about seven times. This graph also shows that the concentrations leaking from the liner are very minimal and uniform, which is desirable in cases where the water table rises in the rainy seasons.

Figure 4-16 shows the effect of scrape-and-replace time interval on the NH_4-N transport from the liner. The simulations were carried for a period

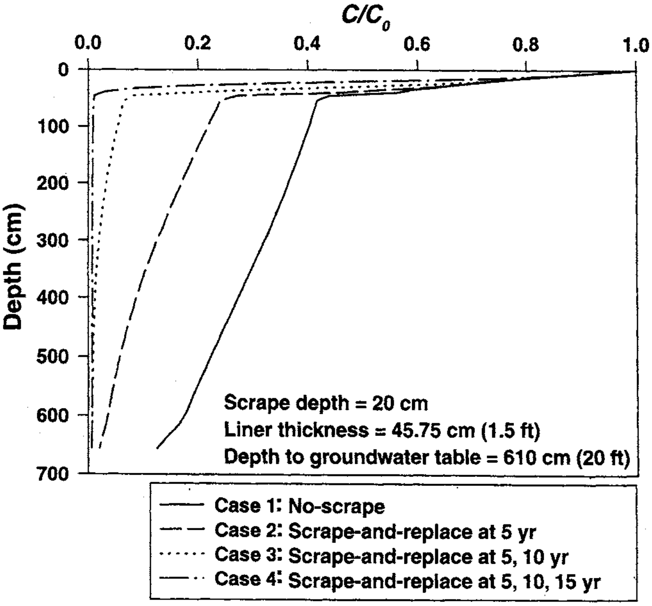


FIGURE 4-14. NH_4-N Concentration Profiles at 25 yr with and without Scrape-and-Replace of the Lagoon Liner (Source: Bonala et al. 2000).

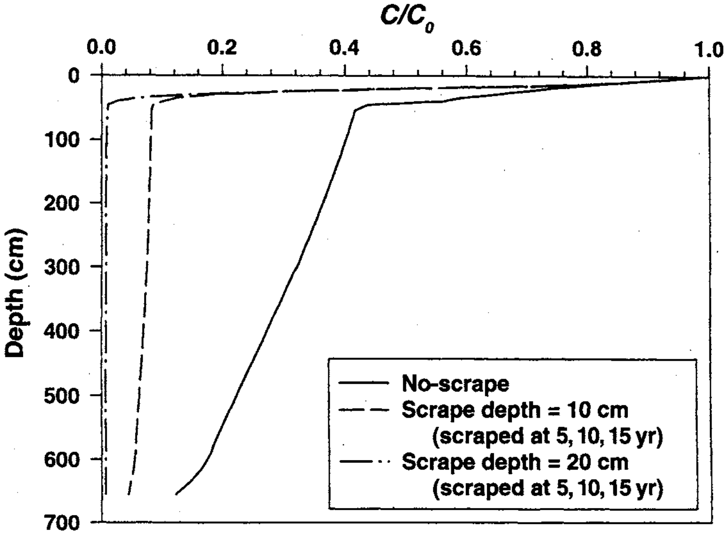


FIGURE 4-15. NH_4-N Concentration Profiles for 10-cm and 20-cm Scraping Depths at 25 yr (Source: Bonala et al. 2000).

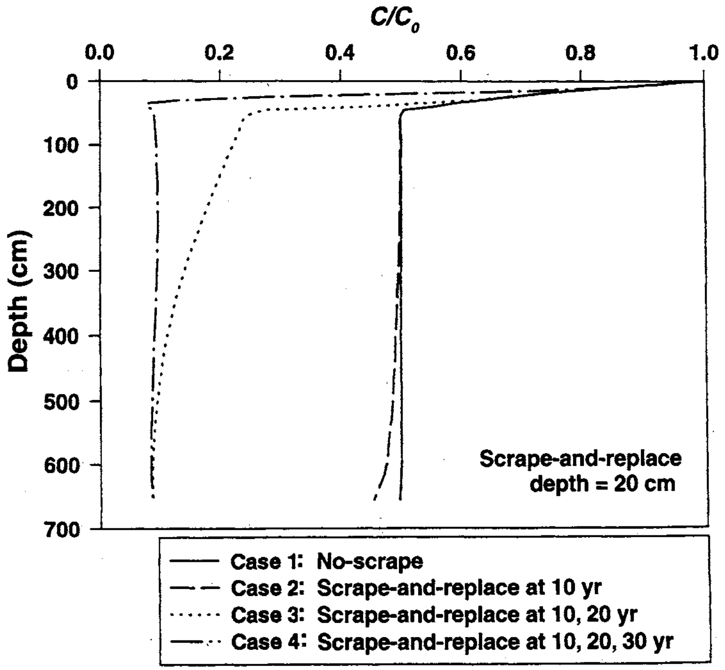


FIGURE 4-16. NH_4-N Concentration Profiles at 40 yr with and without Scrape-and-Replace of the Lagoon Liner (Source: Bonala et al. 2000).

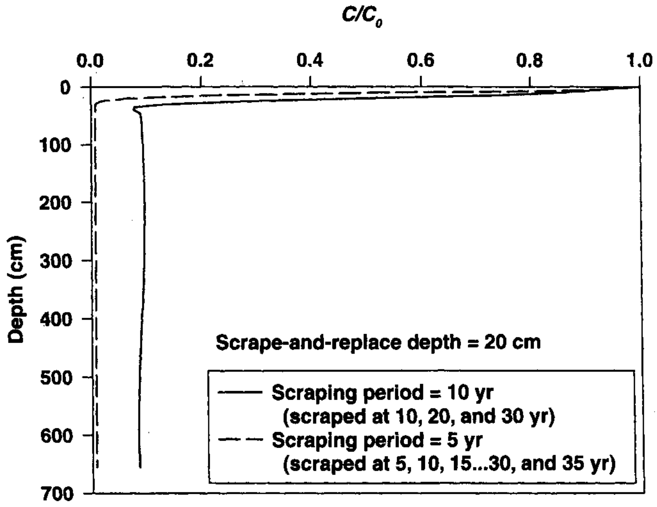


FIGURE 4-17. NH_4-N Concentration Profiles at 40 yr for 5-yr and 10-yr Scrape-and-Replace Intervals (Source: Bonala et al. 2000).

of 30 yr, keeping the scrape-and-replace period as 10 yr and the scrape depth as 20 cm. Similar observations can be made from this graph. Concentration profiles in Figure 4-17 show the effect of scrape-and-replace period on the contaminant transport. The graph shows that shorter scrape-and-replace intervals result in less contaminants leaking from the system.

4.5 CONCLUSIONS

Undigested suspended solids settle to the bottom of the lagoon and physically clog the pores of the liner material. This physical clogging is aided by chemical and biological interactions between the animal waste and lagoon material that further clog the liner material. The clogging of the liner that is caused by these interactions is certainly known to reduce the hydraulic conductivity of the liner, but the decrease in magnitude is difficult to estimate. For this reason, the clogging of liner materials is a debated topic requiring more research to address the reduction of hydraulic conductivity.

Mathematical models like SWMS-2D could be used to model the transport of contaminants in the subsurface for possible scenarios. Simulation results from Scenario 1 indicate that hydraulic conductivity reduction due to clogging has a significant effect on the ammonium concentrations in the subsurface. An important conclusion from this scenario is that, when the hydraulic conductivity is reduced by two orders of magnitude, ammonium breakthrough has not occurred from the liner until 25 yr.

Simulations with shorter clogging times have shown less ammonium concentrations in the subsurface. Results from Scenario 1 simulations should be interpreted with due care because the hydraulic conductivity reduction caused by clogging may be compensated or increased by other factors, such as desiccation cracks and animal burrows.

Results from Scenario 2 simulations indicate that scraping the lagoon liner increased the NH_4-N concentrations in the subsurface significantly, even for a 10-cm scrape depth. Therefore, extreme care must be exercised when reclaiming the lagoon volume to ensure that the lagoon liner is not scraped. On the other hand, results from Scenario 3 simulations indicate that scrape-and-replace of lagoon liners has shown to be a promising method to slow ammonium transport into the subsurface.

Chapter 5

EVALUATING SEEPAGE LOSSES AND LINER PERFORMANCE AT ANIMAL WASTE LAGOONS USING WATER BALANCE METHODS

Jay M. Ham

Anaerobic lagoons have been used effectively for more than 40 yr to collect, store, and treat wastewater from animal feeding operations (AFOs). In 1992, there were more than 6,600 AFOs in the United States that had more than 1,000 animal units, and most of these facilities used lagoons as part of their waste management plan (Kosco and Hall 1999). Conversion of organic matter to methane and carbon dioxide gases (via anaerobic digestion) has proven to be an effective way to remove 35% to 70% of the manure solids that initially enter a lagoon. Furthermore, up to 80% of the nitrogen added to the lagoon is lost from the surface by ammonia volatilization (Koelliker and Miner 1973). The remaining wastewater eventually evaporates or is applied to nearby farmland, whereas undigested organic solids and sediments slowly accumulate on the bottom of the lagoon. Despite the efficiency and convenience of anaerobic lagoons, the residual liquid waste contains nitrogen, salts, bacteria, viruses, pharmaceuticals, and other potential contaminants (Bitton 1999). Thus, under certain conditions, seepage losses from the sides and bottoms of earthen lagoons could pollute groundwater near AFOs.

Because of the potential environmental impact of seepage, most states have set design standards to limit the maximum allowable seepage rate from animal waste lagoons (Parker et al. 1999; Ham and DeSutter 2000). These standards range from 0.5 to 6 mm day^{-1} and may differ among species (e.g., swine, cattle). It is important to recognize that these are "design standards" rather than "performance standards." When applying for a new lagoon permit, the applicant must show that the proposed design, materials, and construction practices are such that the expected "engineered" seepage rate will meet the standard. A critical step in the engineering process is the design of the compacted-soil or clay liner that blankets the bottom and sides of the basin. In many cases, the coefficient of permeability (K_s) of the proposed liner material is measured in a laboratory (e.g., ASTM D 5084-90). Often, this is done using a single disturbed sample collected from the construction site. Once the value for K_s is known, Darcy's law or other simple models are used to design the liner thickness and lagoon depth to theoretically meet the seepage requirements (Rowe et al. 1995). Unfortunately, many factors may cause actual seepage rates from working lagoons to deviate significantly from the "engineered" rate (Daniel 1984). Seepage is affected by construction methods, weathering of side embankments, roots and burrowing pests, sludge deposits, and many other processes (Chang et al. 1974; Hills 1976; Miller et al. 1985; Rowsell et al. 1985; Barrington and Madramootoo 1989; Kim and Daniel 1992; McCurdy and McSweeney 1993; Ham and DeSutter 1999; Maulé et al. 2000). Table 5-1

TABLE 5-1. Processes and Factors that Affect Whole Lagoon Seepage

Factor or Process	Effect*
Macropores, preferential flow paths from debris in liner material, insects, roots	+
Excessive seepage on side embankments caused by freeze/thaw or wet/dry cycles, and erosion	+
Inadequate liner construction techniques and quality control	+
Drying and cracking of compacted liner when dry (when new or during cleaning)	+
Spatial variation of liner compaction and thickness	+
Lagoon design (side:bottom ratio, presence of sedimentation basins or filter strips)	+/−
Rate of waste input, lagoon age, and cleaning frequency	+/−
Type of waste (animal species), feed quality and type	+/−
Thickness of compacted liner	+/−
Waste depth (pressure head and wetted periphery)	+/−
Depth to water table, vadose zone characteristics	+/−
Environmental factors (e.g., temperature effect on waste chemistry and microbes)	+/−
Physical clogging of liner by manure and sediments	−
Chemical defloculation of soil aggregates in liner (i.e., salts)	−
Biological clogging of soil pores by microbial by-products	−

Note:
*Positive and negative signs depict factors that typically increase (+) or decrease (−) seepage rates.

summarizes some of these factors. Clearly, the processes that govern seepage are numerous and complex, and vary with time and management factors. Thus, evaluating lagoon/liner performance in the field requires a "top-down" approach that integrates across all of these factors simultaneously. The most common approach to this problem has been to measure whole-lagoon seepage rates using water balance methods.

Several researchers have estimated seepage from existing animal waste lagoons by performing short-term water balance experiments (Hart and Turner 1965; Davis et al. 1973; Robinson 1973; Clark 1975; Ham 1999, 2002a; Ham and DeSutter 1999; Glanville et al. 2001). Typically, seepage was calculated as the difference between changes in waste level and evaporation when waste inputs and outputs were precluded or quantified. Results showed that seepage rates from lagoons ranged from 0.2 to 5 mm day^{-1}. However, some of this work was conducted on miniature, pilot-scale lagoons and may not have been representative of the large lagoons used at many modern AFOs. Furthermore, the resolution of the techniques used in many early studies was such that water balance studies had to be conducted for long periods (>30 days). Unfortunately, withholding waste inputs for extended periods is not logistically feasible at many commercial AFOs because waste must be flushed from the barns (e.g., swine and dairy) on a routine schedule. Recently, more comprehensive studies conducted in Kansas and Iowa estimated the water balances of commercial lagoons by measuring depth changes and evaporation over very brief measurement cycles (e.g., 1 to 6 days). In the Iowa study, Glanville et al. (2001) found an average seepage rate of 1.2 mm day^{-1} from 12 lagoons and 15 slurry pits when results were adjusted to reflect a waste depth of 1.8 m. However, evaporation was not considered, so seepage may have been slightly overestimated. Ham (2002a) found an average seepage rate of 1.1 mm day^{-1} from 20 lagoons in Kansas that had an average waste depth of 3.0 m. Ham (2002a) coupled seepage results from the Kansas study with data on liner thickness and basin geometry to calculate the apparent insitu permeability of the compacted liners.

Measuring the whole-lagoon seepage rate not only provides information on the performance of the liner but also allows analysis of the chemical flux boundary conditions at the interface between the lagoon liquid and the underlying soil. For example, Ham and DeSutter (1999) combined their seepage results with data on lagoon chemistry to estimate the rate of nitrogen leaching (kg m^{-2} s^{-1}) into the subsoil. Data of this type could greatly enhance the accuracy of models that predict the fate and transport of contaminants in the vadose zone. The rate of input loading of a contaminant (e.g., total maximum daily load) is an essential factor when assessing the risk of groundwater contamination (Nolan et al. 1997). Ham and DeSutter (2000) used a risk-based approach to propose a site-specific logical framework for determining lagoon liner specifications.

This chapter discusses techniques for evaluating seepage losses from existing animal waste lagoons under field conditions. Sections 5.1 through 5.4 describe instrumentation and theory for measuring whole-lagoon seepage using water balance methods. Sections 5.5 through 5.7 show how seepage results can be used in combination with other data to evaluate the performance (i.e., permeability) of compacted-soil liners and to approximate the rate at which soluble chemicals are leaching from lagoons.

5.1 MEASURING THE LAGOON WATER BALANCE AND SEEPAGE RATE

The water balance of an earthen lagoon can be expressed as

$$\Delta D = P + I - E - S - O \tag{5-1}$$

where

ΔD = rate change in lagoon depth
P = precipitation
I = waste input from production barns or runoff
E = evaporation
S = seepage
O = waste removed by pumping

All values are expressed in mm day^{-1}. All variables are considered positive except for ΔD, which is negative when waste levels are declining and positive when they are increasing. In a seepage study, the goal is to eliminate or measure variables in Eq. (5-1), so S can be determined by residual calculation. In most cases, investigators work with the lagoon operator to stop waste inputs and outputs during an experiment. This may require placing caps over inlet pipes or temporarily damming channels and culverts that route waste into the lagoon. In any case, withholding waste inputs into a lagoon for any appreciable period can be difficult, and this should be considered when selecting a study site. Accounting for precipitation that enters a lagoon can be troublesome because of runoff from the side embankments and the surrounding landscape. At cattle feedlots, where lagoons are designed to collect runoff from large areas, even a small rain can cause a large increase in waste depth. Thus, if precipitation occurs during the study, the best choice is to wait for a period of dry weather and restart the study. If I, O, and P can be eliminated from the water balance, then seepage can be calculated as the difference between the change in depth and evaporation as follows:

$$S = -(\Delta D) - E \tag{5-2}$$

Recall that $-(\Delta D)$ usually will be positive, because ΔD is negative when waste levels are declining. In most cases, measurements are performed for a minimum of several days, and S is calculated from total E and ΔD over the period. For example, a 5-day experiment may produce the following results: $\Delta D = 35$ mm, $E = 25$ mm, and $S = 10$ mm or 2 mm day^{-1}. Clearly, the resolution and accuracy of the seepage estimate will be dependent on the measurement of cumulative E and ΔD over the study period. Thus, the duration of the experiment and the magnitudes of E and ΔD may affect the results.

The use of Eqs. (5-1) and (5-2), as described here, assumes that the static water table is lower than the bottom of the lagoon and does not affect waste movement through the liner. For work in regions with shallow water tables, it is imperative to make sure that groundwater is not moving in or out of the lagoon. Drilling a shallow monitoring well may be necessary to verify the thickness of the vadose zone before proceeding with a water balance test.

5.2 MEASURING CHANGES IN WASTE LEVEL

Measuring the cumulative decline in waste level is one of the most import aspects of any water balance experiment. Many automated water-level recorders have been developed and are commercially available. However, in a lagoon experiment, the magnitude of ΔD may be small compared to applications in hydrology or industry (e.g., tide recorders, river level, liquid tank level). As shown in the previous example, total changes in depth over the course of most studies will be less than 50 mm. Thus, instruments are needed with a small full-scale range (e.g., 1 m); high resolution; and high accuracy (e.g., 0.5 mm). Because ΔD is calculated as the difference between two readings, instrument bias (i.e., constant systematic error) is not important. For example, a 1-cm bias error at the start of an experiment will not affect the determination ΔD as long as that error is constant throughout the study. In addition, because the experiment is operated for a long period, instantaneous random errors (electronic noise) that are much smaller than ΔD typically are not important. The most significant problems are caused by calibration error (slope error) as well as by any effects on the sensor response caused by environmental factors, such as waves or temperature fluctuations. A good discussion of the error types and their importance is given by Rabinovich (1995).

The four most common ways to measure waste level changes in lagoons are to use (1) float-based transducers (linear displacement), (2) subsurface pressure transducers, (3) ultrasonic ranging, and (4) siphon methods. Float-based systems consist of a 10- to 15-cm-diameter float that is housed in a wet well made from larger diameter pipe. A wire or beaded chain is connected to the float, then looped over a pulley at the top of the wet well, and finally attached to a counterweight hanging from the opposite side of the pulley. An encoder is used to monitor the float height by sensing the rotation of the pulley. Other variations of float-based recorders include the use of potentiometers with spring-loaded retractable tethers (Ham and DeSutter 1999, 2003) and floats connected to cantilever-beam strain gauges (Yoder et al. 1999). Float-based recorders are easy to use and low-cost and can be customized for installation at lagoons. For example, the recorder developed by Ham and DeSutter (1999) had a resolution of 0.17 mm, a full-scale range of 0.6 m, and could be moved easily and reinstalled at different locations. Because most float systems have rotating mechanisms, precautions must be taken to reduce static friction.

Pressure sensors can measure waste levels by detecting small changes in piezometric head. Most transducers are small, cylinder-shaped probes that are submerged 0.5 to 3 m below the surface. A signal cable, which contains the signal wires as well as a small tube that vents the transducer to the atmosphere, is routed to the surface. The transducer is usually placed in a wet well to minimize wave-induced pressure fluctuations. Pressure transducers have the advantage of being able to operate in slurries and sludge-laden lagoons. In addition, they are not influenced by temperature-induced changes in fluid density. Thus, pressure sensors are the only option when attempting a water balance of a lagoon with an ice-covered surface.

Ultrasonic ranging is emerging as a popular method for measuring water levels in both industry and hydrology. These transducers are mounted above the waste surface and pointed downward. The time required for sound waves to bounce off the surface and return to the transducer is used to calculate distance. One of the main advantages of ultrasonic sensors is that they are not in contact with the waste and are thus easier to deploy. Performance of ultrasonic ranging can be improved by "shooting" the sound waves down a vertical plastic pipe. This improves the acoustic performance of the sensor, and the pipe acts as a wet well to reduce wave effects.

Changes in waste level can be detected gravimetrically by using a siphon hose to hydraulically connect the lagoon to a beaker of fluid sitting on an electronic balance positioned outside the lagoon (Glanville et al. 1999, 2001). As waste levels change, fluid moves in or out of the beaker to maintain equilibrium with the lagoon surface, and changes in beaker mass provide a measurement of ΔD. This method has high resolution (e.g., 0.025 mm) but requires close supervision of the equipment. However, a drawback to this technique is that the configuration of the lagoon and surrounding landscape must be such that a siphon can be established between the waste and the balance-beaker apparatus positioned outside the lagoon. In many situations, ground level adjacent to the lagoon is higher than the waste, thus disallowing the technique.

Several of the instruments that can be used to measure waste level changes are shown in Table 5-2. The option that is best for a given situation is dependent on the duration and desired accuracy associated with each experiment. Instruments with smaller ranges have higher resolution. However, if one intends to leave the equipment unattended for long periods (e.g., several weeks), it is better to use a recorder with greater range to accommodate fluctuations associated with precipitation and waste inputs. Regardless of the technique used, deploying or mounting transducers at a lagoon can be difficult. Many sites have a staff gauge or sight gauge (vertical pipe or post set in concrete) that can be used as an anchor point for the transducers. Ham and DeSutter (1999) and Ham (1999) used existing staff gauges to support their wet well and float recorder at swine waste lagoons. All of the techniques are affected by wind and waves that cause water to "pile up" on the downwind edge of the lagoon. The magnitude of this error is dependent on the size of the lagoon, wind speed, and wind duration. It is common for changes in wind speed or wind direction to cause a 1- or 2-mm change in water level at the site of the recorder (Ham 2002b). An anemometer and wind vane should be deployed at the lagoon to improve later analysis of data. The effect of wind can be minimized by placing the waste-level recorders in the center of the lagoons. In practice, however, many researchers use data from water-level recorders only when wind speeds are low (e.g., <2 m/s^{-1}).

TABLE 5-2. Characteristics of Several Water-Level Recording Instruments that have been used at Animal Waste Lagoons or in Similar Applications

Type	Model	Range (m)	Resolution (mm)
Linear Displace. (float)	Unimeasure, LXPA2.5[a]	0.16	0.03
Linear Displace. (float)	Unimeasure, LXPA25[a]	0.64	0.17
Linear Displace. (float)	Unimeasure, LXPA50[a]	1.27	0.34
Cantilever Beam (float)	Tennessee Recorder[b]	1 or 3	1.0
Ultrasonic Ranging	Lundahl, DCU-7110[c]	4.9	1.3
Pressure	Global Water[d]	0.91	0.30
Siphon Tube (gravimetric)	Iowa State Univ.[e]	0.16	0.03

Notes:
[a] Unimeasure Inc., Corvallis, Ore. (Ham and DeSutter 1999, 2003).
[b] Yoder et al. 1999.
[c] Lundahl Instruments, Inc., Logan, Utah.
[d] Global Water, Gold River, Calif.
[e] Glanville et al. 1999.

These conditions often occur just before dawn, so ΔD is calculated by comparing early morning results on successive days.

Figure 5-1 shows fluctuations in waste level for 17 days at a swine waste lagoon in western Kansas (Ham and DeSutter 1999). This was a "pull-plug" operation, where swine were raised on slatted floors inside production barns, and waste was temporarily collected in shallow concrete pits directly under the animals. The underfloor pits were drained into the lagoon on day 105, causing a 60-mm increase in depth. On days 105 to 107, to control odor, a portion of the lagoon water was pumped back into the barns to create a thin layer of liquid in the bottom of the pits. Several small rain showers also were detected. These data show how a sensitive water-level recorder can easily detect small inputs and outputs to a lagoon. This lagoon was 2.1 ha in size, so each millimeter of waste represented 21 m^3 of volume. The waste inlet tubes to the lagoon were capped between days 109 and 116. During this period, ΔD was 35 mm; thus, seepage plus evaporation was about 5 mm day^{-1}. The design standard for this lagoon at the time of construction was 6.1 mm day^{-1}. Thus, even without accounting for evaporation, this data shows that the lagoon was performing within compliance. Much can be learned about the performance of a lagoon by simply measuring ΔD during periods of low

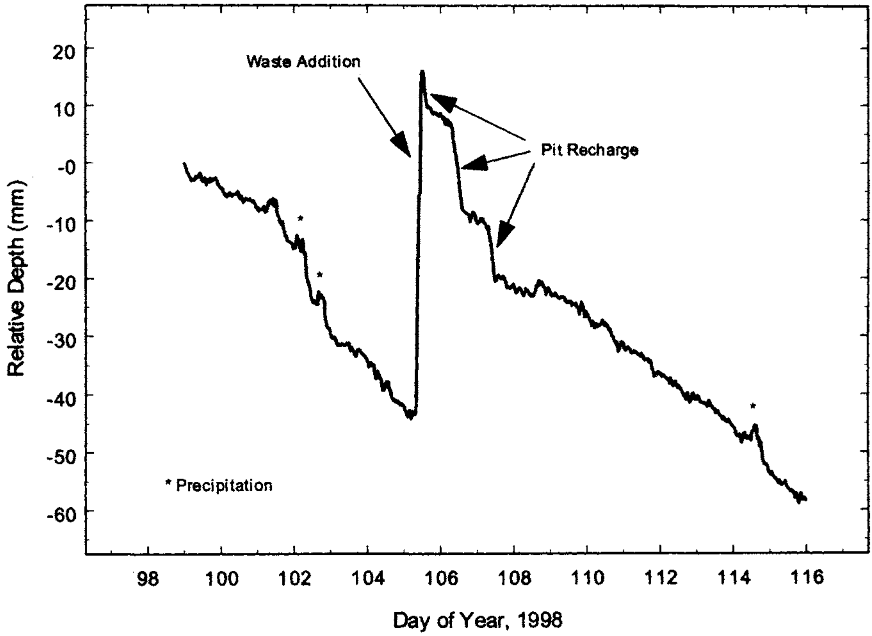


FIGURE 5-1. Fluctuations in Waste Level at a Swine Waste Lagoon over a 17-Day Period.

evaporative demand. Ham and DeSutter (1999) and Glanville et al. (1999) both showed that reasonable estimates of S could be obtained by measuring ΔD over a single winter night and assuming that E was negligible. However, as shown in Section 5.3.3, E from lagoons can be quite large during the night, so ignoring the effect of E must be limited to periods when vapor pressure differences between the surface and the air are very small (e.g., winter nights with low wind speeds).

5.3 ESTIMATING EVAPORATION FROM LAGOONS

Estimating evaporation from lagoons is usually the largest source of uncertainty in a water balance experiment (Ham 2002b). Seepage rates from lagoons are often much smaller than evaporation; thus, a small error in E can cause a large percentage error in S. Evaporation from lagoons can be approximated by measuring water losses from evaporation pans positioned on the lagoon bank or floating on the lagoon surface. Another option is to monitor environmental conditions at the lagoon and calculate the evaporation using a meteorological equation (e.g., Penman or Priestley-Taylor).

5.3.1 Class-A Evaporation Pans

Evaporation pans are traditional tools for approximating reference crop evapotranspiration and estimating E from open water (Brutsaert 1982; Jensen et al. 1989; Jones 1992). Most work has been done using a Class-A pan, which is a cylindrical steel tank (1.21 m in diameter, 0.254 m in height) mounted 0.15 m above the ground on a wooden platform. Water levels are maintained 0.05 m from the top of the pan rim. Class-A pans can be deployed adjacent to lagoons during water balance experiments to estimate E (White and Denmead 1989). Unfortunately, the water temperature and microenvironment of the pan are not the same as those of the surface the lagoon. Horizontal advection, radiation heating of the pan's sidewalls, turbulence on the pan rim, and other factors complicate the energy balance (Jacobs et al. 1998). To estimate actual E from lagoons, measurements of pan evaporation, E_{pan}, are adjusted downward using an empirical pan coefficient, k_p, as follows:

$$E = k_p E_{pan} \tag{5-3}$$

Pan coefficients are affected by the type of pan used, surrounding landscape and vegetation, and environmental conditions. For open water, k_p for a Class-A pan typically ranges from 0.6 to 0.9, with a median value near 0.75 (Brutsaert 1982; Jones 1992). Coefficients tend to be lower in dry, windy climates compared to humid, calm climates. Also, k_p is lower in the winter months and higher in the summer. Several researchers have developed formulas for k_p based on wind run, relative humidity, and surrounding fetch (e.g., Snyder 1993; Pereira et al. 1995; Raghuwanshi and Wallender 1998). Unfortunately, lagoons at AFOs often are situated in unusual locations adjacent to animal pens, barns, and other structures. In many cases, lagoons are in natural depressions or surrounded by steep embankments. These complications make the aerodynamic conditions at a lagoon highly variable and complex. An evaporation pan positioned on the edge of the lagoon will be affected by surroundings unique to that location. In addition, changes in wind direction could affect the boundary layer flowing over the pan and change k_p. Finally, the microclimate (relative humidity, temperature, and wind speed) at the pan may be different from conditions at the surface of the lagoon waste. All of these factors make it almost impossible to estimate k_p for a lagoon experiment using one of the formulas presented in the literature.

Despite the aforementioned problems, the advantages of evaporation pans are simplicity of design and ease of use. A portable Class-A–sized pan was designed that could be deployed on the berm of lagoons (Figure 5-2). The pan was made from a 1.2-m-diameter stock watering tank cut down to a height of 25 cm. A 0.3-m^3 (55-gal.) drum filled with waste from the lagoon was used to feed liquid to the pan. Both the pan and the drum rested on wooden pallets to insulate them from the ground. Waste levels in the pan were maintained 5 cm from the top edge using a remote water-level controller. The rate of change in waste levels inside the 55-gal. drum was monitored using a float-based recorder equipped with an LX-PA50 linear transducer (Table 5-2). Sample calculations showed that measurements of water-level changes inside the drum could measure E_{pan} to within 0.08 mm. A single fill of the drum would supply enough waste to replace 0.18 m of evaporation from the pan, which was sufficient capacity for most water balance experiments lasting less than 2 weeks. Every 1 mm

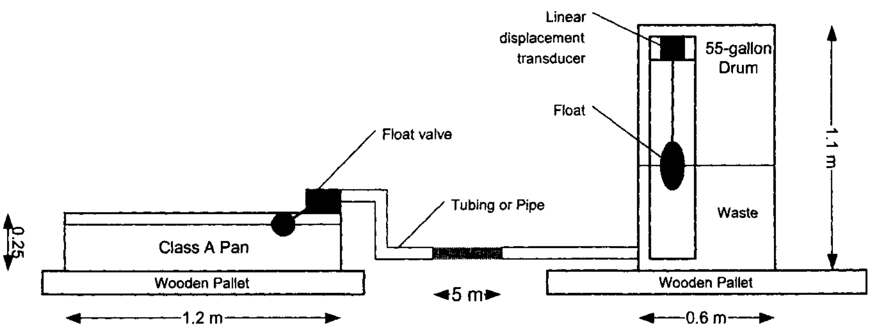


FIGURE 5-2. Class-A Evaporation Pan Configuration for Use on the Embankment of an Animal Waste Lagoon.

in evaporation from the pan caused a 4.7-mm decline in waste levels in the drum. The drum was positioned at least 5 m from the pan to avoid affecting the wind speed profile.

Pan coefficients were determined by comparing pan evaporation to lagoon evaporation measured using the bulk transfer method, a micrometeorological technique recommended by Ham (1999). Results from nine experiments at four locations showed that k_p ranged from 0.35 to 0.89 (Table 5-3). The coefficients were very low in March and early April when the water in the pan was much warmer than that in the lagoon. The highest value of k_p, 0.89, occurred in the summer during hot, windy conditions. Seasonal variation in k_p is very undesirable and makes the use of evaporation pans very tenuous for predicting lagoon evaporation. If the median value for k_p, 0.64, were applied to all nine cases, the pan would have produced relatively good results in five of the nine tests (error <7%). However, very large errors (>50%) would have resulted in three cases. It might be possible to use a Class-A pan if waste temperatures in the pan and lagoon were monitored and k_p adjusted using some type of correction algorithm. However, pans have other potential problems that adversely affect their utility (e.g., inoperable during freezing conditions, dust films

TABLE 5-3. Pan Coefficients, k_p, as Determined for a Class-A Evaporation Pan at Animal Waste Lagoons

Start Date	Duration (Days)	Lagoon Type	Pan Evap. (mm)	Lagoon Evap.[a] (mm)	Pan Coef.
March 30	7	Swine[b]	41.9	14.5	0.35
April 8	7	Swine[b]	81.6	33.2	0.41
April 28	10	Dairy[c]	47.5	29.1	0.61
May 24	21	Swine[b]	133.6	89.0	0.67
May 27	5	Swine[d]	63.4	38.0	0.60
July 16	12	Swine[b]	147.5	101.0	0.68
Aug. 13	8	Swine[b]	117.2	104.0	0.89
Nov. 14	15	Cattle[e]	24.1	15.1	0.64
Nov. 23	9	Swine[b]	23.18	16.3	0.70

Notes:

[a] Evaporation from the lagoons was measured using the bulk transfer method (Eq. (5-5), Figure 5-1). Pan evaporation was measured with the apparatus shown in Figure 5-3 positioned on the berm of each lagoon.

[b] 2.2-ha swine waste lagoon in southwest Kansas.

[c] 0.2-ha first-stage lagoon at a small dairy in eastern Kansas.

[d] 1.2-ha swine waste lagoon in central Oklahoma.

[e] 0.4-ha runoff collection lagoon at a cattle feedlot in central Kansas.

on the surface, and water removal by animals). Variation in k_p and other well-known problems suggest that Class-A evaporation pans are a poor choice for predicting lagoon evaporation in water balance studies.

5.3.2 Floating Evaporation Pans

Ham and DeSutter (1999) and Ham (1999) developed an evaporation pan that floated on the lagoon surface. The idea was that a floating pan would be within the internal boundary layer of the lagoon surface and would experience a more natural microclimate compared to a pan outside the lagoon. In addition, because a floating pan is in direct contact with the lagoon waste, the temperature inside the pan should remain closer to that of the lagoon surface. In theory, k_p would have a smaller range and be closer to unity in a floating pan. The Ham and DeSutter (1999) apparatus consisted of an instrument raft, 1.5-m wide and 2.7-m long, built from two low-profile flotation panels (Figure 5-3). An area between the flotation panels supported a square evaporation pan (1.09 m × 1.09 m × 0.38 m) made from 0.8-mm-thick galvanized sheet metal or aluminum. The floating pan had the same surface area as a Class-A evaporation pan. When installed, the pan held 0.23 m of waste with a 0.15-m rim extending above the waterline. Waste levels in the pans were monitored continuously using a float-based detector housed in a 0.15-m-diameter wet well. Water pumps were mounted inside and outside the pans, so waste could be added or removed by remote control. Four 0.3-m-diameter wheels were mounted on the underside of the floating platform so it could be rolled up or down a lagoon's side embankment to facilitate installation and retrieval. The raft was equipped with a suite of meteorological sensors and a data acquisition system. Studies of the floating pans in western Kansas showed that k_p ranged from 0.69 to 0.94, with a mean of 0.81 (Ham 1999). As in previous work with onshore pans, k_p for the floating platform was affected by wind speed and radiation. Daytime surface temperatures inside the pan were sometimes 4°C higher than those of the lagoon surface, causing E_{pan} to exceed E from the lagoon. Heat exchange on the exposed rim of the pan, coupled with a lack of surface mixing (wave action), appeared to be the main cause of the temperature rise. Nevertheless, once an appropriate k_p had been determined for a site, the floating pans could reliably estimate E to within 0.5 mm day^{-1}. Figure 5-4 shows excellent agreement between actual and pan-derived estimates of E. Data was collected from two floating pans in a 2-ha lagoon lined with 1-mm-thick plastic. Actual E from the lagoon was determined by simply measuring ΔD and assuming that S was zero (no seepage through the plastic liner). Once the pans had been tested, the devices were used to conduct water balance studies at many locations (Ham et al. 1999). Unfortunately,

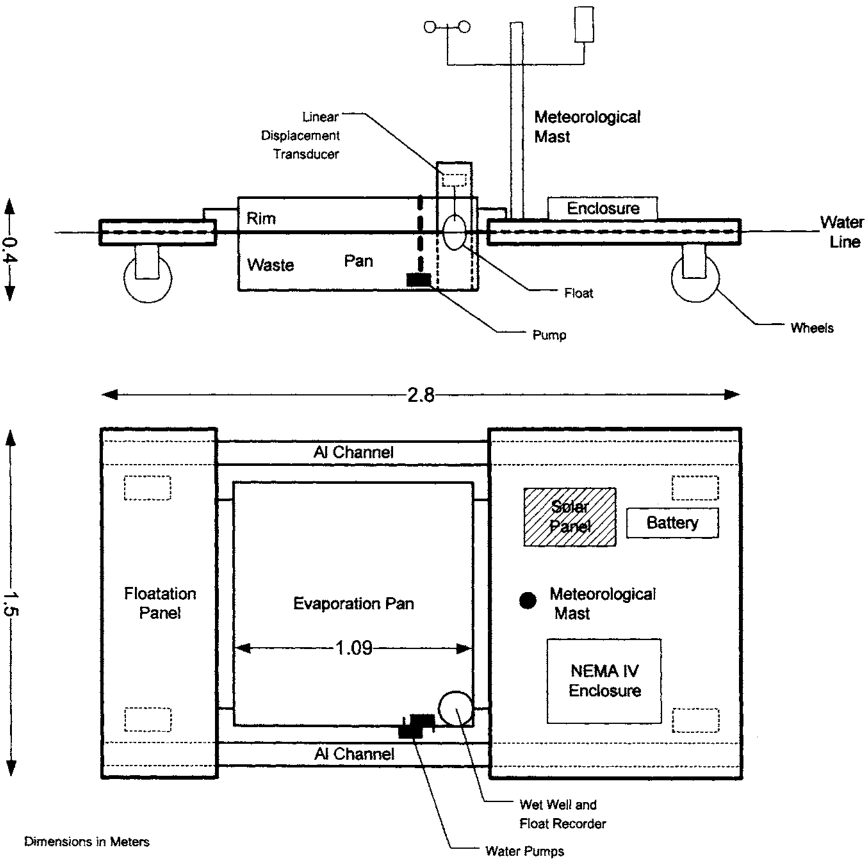


FIGURE 5-3. Floating Evaporation Pan Used to Estimate Evaporation from Lagoons by Ham and DeSutter (1999). A Photograph of the Apparatus is Provided in Ham 1999.

the floating pans were sometimes flooded by large waves when average wind speeds exceeded 10 m s^{-1}. A film of dust sometimes would develop inside the pans, especially at cattle feedlots, which may have affected E_{pan}. Ham (1999) concluded that a floating pan was a viable option for measuring lagoon E. However, variability of k_p, wave effects, and dust accumulation can reduce confidence in the results.

5.3.3 Calculating Evaporation Using Meteorological Formulas

Evaporation from lagoons can be calculated using input data from a weather station or meteorological buoy floating on the surface. An

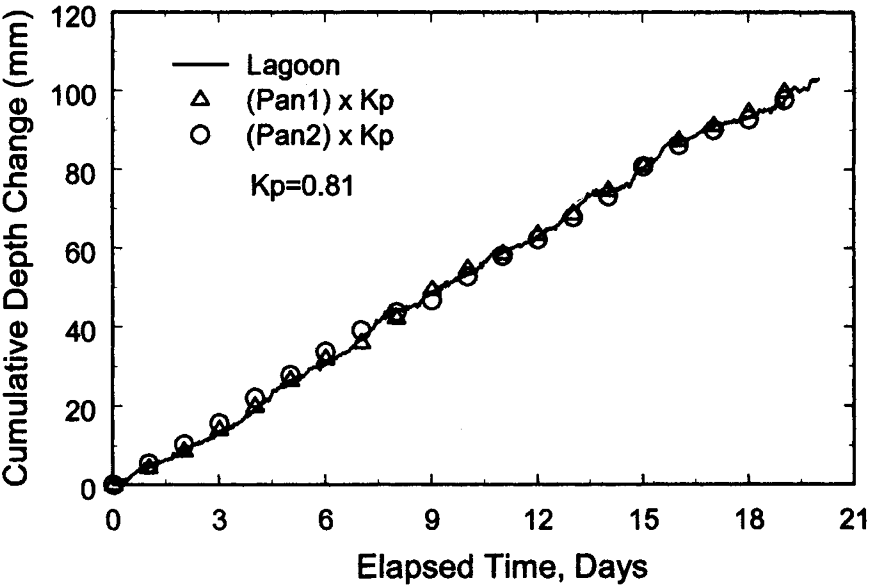


FIGURE 5-4. Comparison of Actual Evaporation to that Approximated with Two Floating Evaporation Pans at a Plastic-Lined Lagoon. A Pan Coefficient, k_p, of 0.81 was Used for the Entire 20-Day Period.

approach that has been used widely to estimate E from open water is the bulk aerodynamic transfer model

$$E = C_e U_r \rho (q_s^* - q_a) \tag{5-4}$$

where

E = evaporation (kg m^{-2} s^{-1})
ρ = air density (kg m^{-3})
q_s^* = saturated specific humidity at water surface temperature (kg kg^{-1})
q_a = specific humidity of the air (kg kg^{-1})
U_r = average wind speed at some reference height (m s^{-1})
C_e = bulk aerodynamic transfer coefficient for vapor transport (dimensionless)

An approximation of Eq. (5-4) expressed in terms of vapor pressure is

$$E = C_e U_r \frac{0.622}{R_d T_s} (e_s^* - e_a) \tag{5-5}$$

where

e_s^* = saturation vapor pressure at the temperature of the water surface (Pa)
e_a = vapor pressure of the air (Pa)
R_d = gas constant (287.04 J kg^{-1} K^{-1})
T_s = temperature of the surface (K)
0.622 = ratio of the molecular weights of water and dry air

Eqs. (5-4) and (5-5) have been used extensively to estimate E from lakes and ponds (Lakshman 1972; Quinn 1979; Bill et al. 1980; White and Denmead 1989). Field studies often are conducted to determine C_e empirically by measuring $(e_s^* - e_a)$ and U_r under conditions where E is measured by some other method. Values of C_e range from 1.0×10^{-3} to 2.0×10^{-3} when U_r and e_a are measured 2 to 3 m above the surface (Brutsaert 1982). However, C_e is dependent on measurement height of U_r and e_a and increases as the reference height decreases. Ham (1999) found a C_e equal to 2.81×10^{-3} at a 2-ha swine waste lagoon when meteorological data was collected 1 m above the waste surface. The bulk transfer approach requires measurements of vapor pressure, wind speed, and surface temperature (i.e., T_s is required to calculate e_s^*). If possible, measurements should be made directly over the lagoon surface (e.g., using a raft or buoy) and recorded at short time intervals (e.g., 30 min).

Penman (1948) introduced a combination model for predicting E from open water that avoided the need for T_s:

$$E = \frac{\Delta}{\Delta + \gamma}\left(\frac{Rn - G}{L}\right) + \frac{\gamma}{\Delta + \gamma} C_e U_r \left(\frac{0.622}{R_d T_a}\right)(e_s^* - e_a) \qquad (5\text{-}6)$$

where

Δ = slope of the saturation vapor pressure curve (Pa K^{-1})
γ = psychrometric constant (Pa K^{-1})
Rn = net radiation (W m^{-2})
G = conduction in the water (W m^{-2})
L = latent heat of vaporization (J kg^{-1})
T_a = air temperature (K)
e_s^* = saturation vapor pressure at T_a (Pa)

A similar model was proposed by Priestley and Taylor (1972) for extended wet surfaces where the internal boundary layer reached equilibrium with the surface:

$$E = \alpha \frac{\Delta}{\Delta + \gamma}\left(\frac{Rn - G}{L}\right) \qquad (5\text{-}7)$$

where α is the empirical Priestley-Taylor coefficient (1.26). Although Eq. (5-7) was developed for expansive surfaces where no advection occurs, Stewart and Rouse (1976) used Eq. (5-7) to approximate E from small ponds. The advantage of the Penman and Priestley-Taylor models is that E can be approximated using basic meteorological data collected in the boundary layer of the lagoon. Both models require a measurement of radiation and T_a, and the Penman model also requires measurements of the vapor pressure deficit and wind speed. The determination of $(Rn - G)$ is perhaps the most important and tedious aspect of using Eqs. (5-6) and (5-7). For lagoons, Rn can be measured using a net radiometer positioned over the surface or approximated as

$$Rn = 0.86Rs - 49 \tag{5-8}$$

where Rs is global shortwave irradiance (W m^{-2}) (Ham 1999). Conduction or heat storage in the lagoon can be estimated by calorimetry from a time series of the water temperature profile. In addition, heat flux between the liquid waste and the underlying soil must be considered in shallow lagoons. Unfortunately, these measurements are difficult and often lead to large errors in G when calculated over short intervals (e.g., hourly) (Webster and Sherman 1995). It is often assumed that G is negligible over a multiday period, and Eqs. (5-6) and (5-7) are solved on a daily basis using 24-h averages of the input parameters. However, Ham (1999) showed that changes in heat storage play an important role in the energy balance of lagoons during a short-term water balance experiment. Neglecting G often caused large errors in E (e.g., 70%) in the spring or fall, when lagoons were rapidly warming or cooling, respectively.

The performances of meteorological models (Eqs. (5-5), (5-6), and (5-7)) were evaluated at a plastic-lined lagoon during Ham's previously mentioned experiment (1999). Results showed that the bulk transfer model provided the best estimate of E (Figure 5-5). However, these experiments were used to determine the optimal C_e for Eq. (5-5); thus, the near-perfect agreement seen in Figure 5-5 was expected. Nevertheless, data shows that C_e must have been relatively constant over the study period. Also, the remarkable agreement in the patterns of actual and calculated E suggests that the physics depicted in Eq. (5-5) accurately represent the evaporative processes from the lagoon. The disadvantage of the bulk transfer model is that results are sensitive to T_s, which usually is measured with an infrared thermometer. Ham (1999) showed that a 1°C error in T_s could cause a 25% error in E. If Eq. (5-5) is used, great lengths must be taken to calibrate the infrared thermometers (e.g., Kalma et al. 1982; Sadler and Van Bavel 1982) and correct data for surface emissivity and downwelling long-wave radiation (e.g., Hipps 1989; Ham and Senock 1994). Attempts to approximate T_s using a thermistor submerged just below the surface have been unsuc-

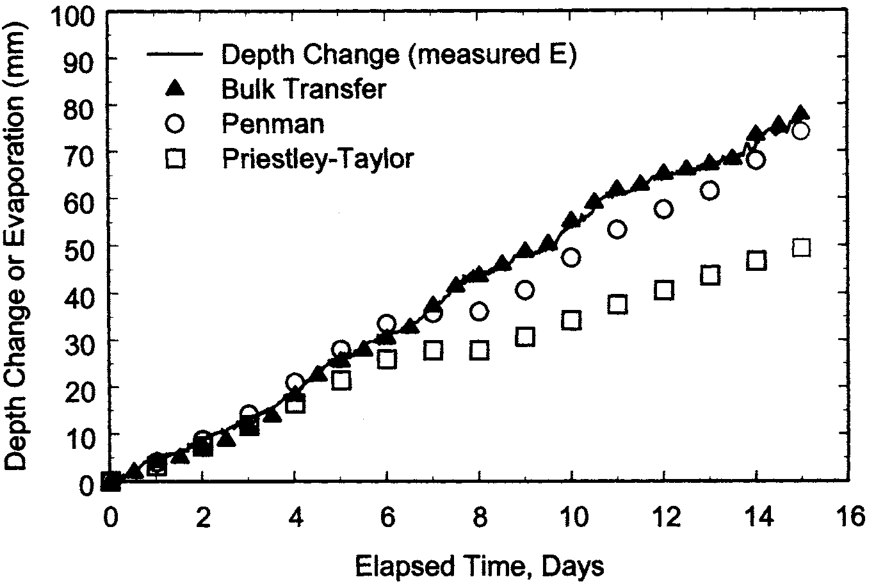


FIGURE 5-5. Comparison of Actual Lagoon Evaporation to that Estimated Using Three Different Meteorological Models [Eqs. (5-4), (5-5), and (5-6)].

cessful. Lagoon waste is turbid, and most radiation is absorbed very close the surface. Because the waste has a rather low thermal diffusivity, strong near-surface temperature gradients are common when net radiation is high. Thus, temperatures at depths as shallow as 50 mm are often 2°C different from T_s.

The Penman and Priestley-Taylor models tended to underestimate E during cloudy weather, and the Penman model also tended to overestimate E during windy conditions (Figure 5-5). Recall that G was neglected, and data was collected in October when heat storage in the lagoon was large (i.e., lagoon temperature was greater than air temperature). During overcast conditions (days 7, 8, and 9), the models failed to account for increases in E caused by large upward conduction of heat, which also contributed to nighttime E. Diurnal patterns in E (e.g., daytime ramps, nighttime plateaus) are not apparent in Figure 5-5, indicating that nighttime losses were comparable to those during the day. Nighttime E is often 40% of the 24-h total in shallow water bodies (White and Denmead 1989; Ham 1999). In Figure 5-5, the Penman model overestimated E at the end of the period, which tended to offset the underestimate that occurred earlier. In the spring, the Penman and Priestley-Taylor models overestimated E (e.g., 70%) because they failed to account for the large fraction of net radiation used to increase the temperature of the waste (data not shown). In summary, accurate measurements of E during short-term water balance experiments can be obtained with Eqs. (5-6) and (5-7) only if G is measured directly.

Salts in the lagoon waste can affect the calculations of e_s^* and Δ used in Eqs. (5-5), (5-6), and (5-7). However, Ham and DeSutter (1999) measured the saturation vapor pressure curve of waste from a swine waste lagoon and found that it was indistinguishable from that of fresh water. Thus, the standard equations used for water apparently are appropriate in most cases [see Appendix A of Ham (1999) for useful equations]. One exception might be lagoons that dispose of waste by evaporation only (i.e., no land application). Salt concentrations would be higher under these circumstances and might affect E by lowering the vapor pressure at the lagoon surface (Turk 1970; Salhotra et al. 1985).

The evaporation pans and the meteorological models presented here do not account for horizontal advection, which can increase E from fetch-limited water bodies like lagoons (Itier et al. 1994; Webster and Sherman 1995). In theory, E should be higher on the upwind edge of the lagoon, where air makes the transition from a dry to a wet surface. However, Condie and Webster (1997) have shown that lower shearing stress (i.e., lower wind speeds, lower C_e) on the leading edge offsets the humidity effect, and there is little variation in E with downwind fetch. Thus, sampling environmental variables at a single point in the center of lagoons is acceptable, provided the surfaces are large enough for a boundary layer to develop at sensor height. Fetch-to-height ratios of 20:1 are usually sufficient for these types of meteorological measurements; thus, lagoons that are 40 m × 40 m or larger in dimension are adequate to minimize sampling errors.

5.4 WATER BALANCE EXAMPLE

Figure 5-6 shows an example of a water balance study conducted at a cattle-feedlot runoff lagoon in western Kansas. Data was collected over an 11-day period in December when evaporative demand was low. The lagoon had been cleaned recently and a new compacted liner constructed. The depth of waste was 2 m. The total change in depth over 11 days was only 22.8 mm; thus, seepage plus evaporation was only 2.1 mm day^{-1}. Estimates of E ranged from 1.5 mm day^{-1} with the Priestley-Taylor equation to 2.6 mm day^{-1} with the Penman equation. The Penman model obviously overestimated E, because E cannot be greater than ΔD without waste additions. One of the evaporation pans and the bulk transfer model produced the same E of 1.9 mm day^{-1}. Assuming that these estimates were the most accurate, the best estimate of S was 0.2 mm day^{-1} (2.1 – 1.9 mm day^{-1}). This example shows how variation in E can affect confidence in S. However, in this case, when E was so small, the measurement of ΔD alone clearly showed that the liner was performing quite well. Even if the smallest esti-

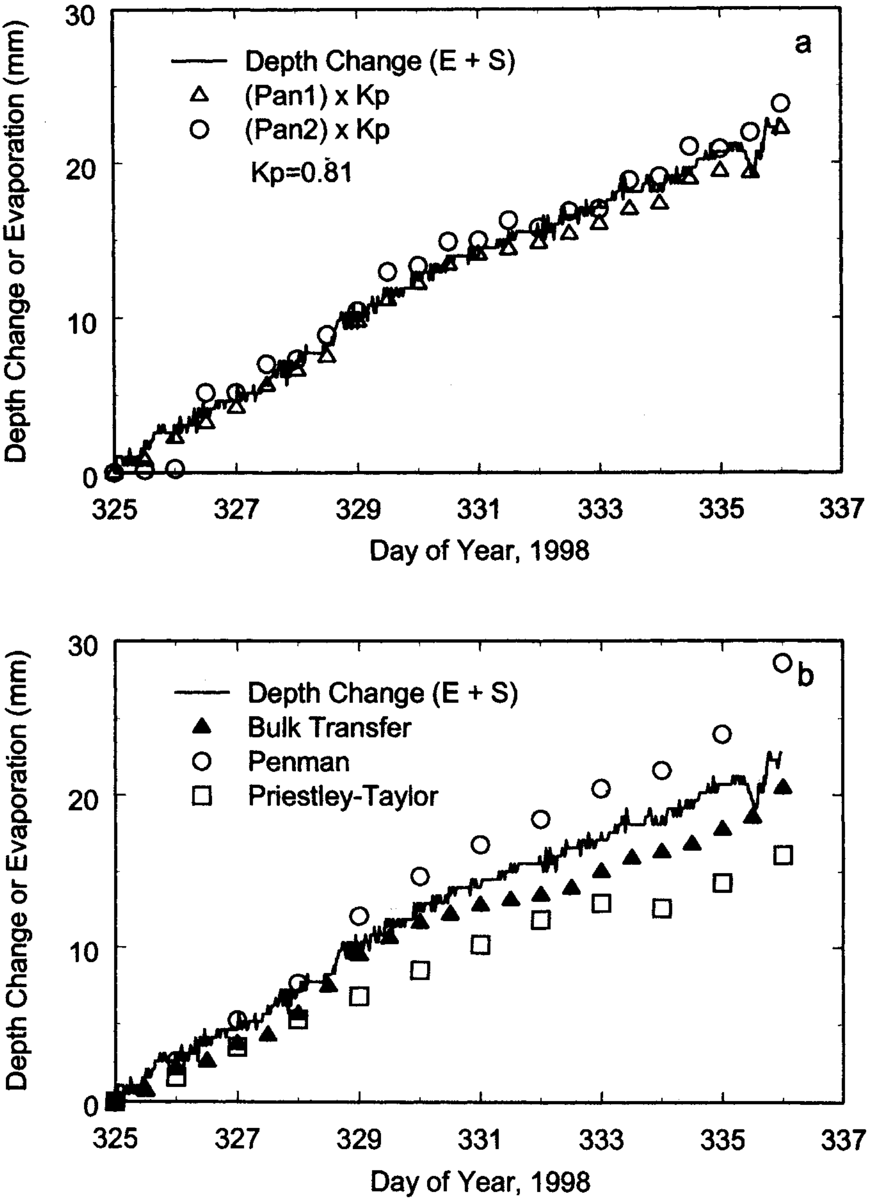


FIGURE 5-6. Cumulative Depth Changes and Evaporation from a Cattle Feedlot Runoff Collection Lagoon. Evaporation was Estimated Using (a) Floating Evaporation Pans and (b) Meteorological Models (Source: Ham 1999).

mate of *E* from the Priestley-Taylor model were used for the calculations, *S* would be only 0.6 mm day^{-1} (2.1 – 1.5 mm day^{-1}). This case is a good example of a study conducted under ideal conditions. An 11-day run was achieved when weather conditions were stable but *E* was small. Although some uncertainty surrounded E, one could confidently conclude that seepage was less than 1 mm day^{-1} and probably less than 0.5 mm day^{-1}.

5.5 RECOMMENDED APPROACH FOR ESTIMATING THE SEEPAGE RATE AND UNCERTAINTY

Ham and DeSutter (2003) proposed a standardized approach for measuring a lagoon water balance using bulk transfer methods for estimating evaporation. The proposed instrumentation consisted of three main components: (1) a meteorological raft or buoy for measuring E (by Eq. (5-5)); (2) a water-level recorder mounted inside a stilling well to monitor ΔD; and (3) a bank station that houses the electronic data collection equipment and tipping-bucket rain gauge (Figure 5-7). The resulting water balance data can be evaluated by plotting ΔD and cumulative E versus time. Figure 5-8(a) shows an example from a swine waste lagoon in December when evaporation rates were low. Note that the sensitivity of the waste-level recorder was so high it detected the diurnal pattern in E even during wintertime conditions. There are several ways to analyze the data from a water balance experiment to arrive at an estimate of S. A good approach, which allows the quality of the test to be judged, is to calculate S at each time period following the designated start time. Basically, a running total of ΔD and E is kept as the test progresses and S is calculated every hour as

$$S(t_i) = \left\{ \frac{[D(t_o) - D(t_i)] - \sum_{t=to}^{ti} E(t)}{t_i - t_o} \right\} \tag{5-9}$$

where

$S(t_i)$ = average seepage rate (mm day^{-1}) from the start time (t_o) to the current time (t_i)
$D(t_o) - D(t_i)$ = total depth change (mm)
$\Sigma E(t)$ = cumulative evaporation
$t_i - t_o$ = elapsed time (days)

Figure 5-8(b) shows an example of the running seepage calculation for the data in Figure 5-8(a). The actual seepage rate from a lagoon should not change with time during a short 5- to 7-day test. Thus, if all measurements were perfect, S calculated using Eq. (5-9) would appear as a flat line. However, random errors cause fluctuations in the calculation of S at the start of the study and then are statistically moderated as the test progresses.

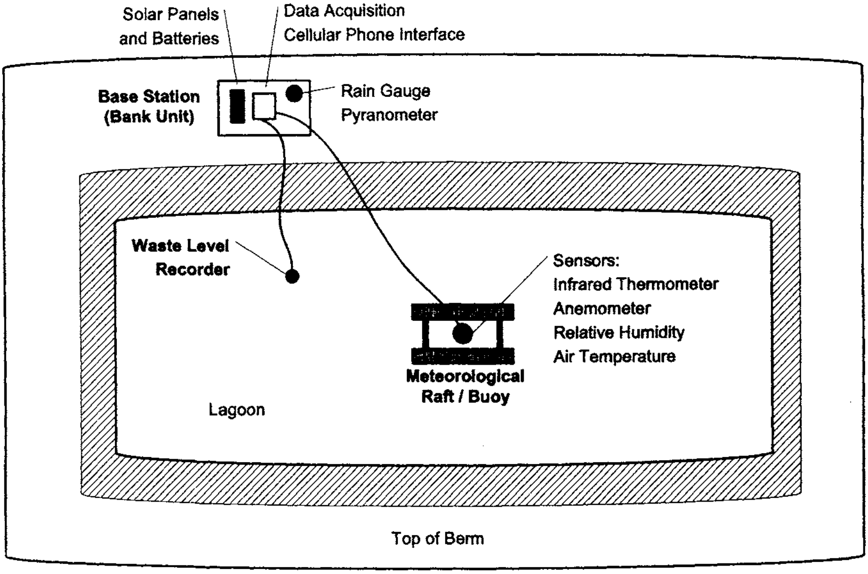


FIGURE 5-7. The Equipment Used to Measure the Lagoon Water Balance (not to scale) (Source: Ham and DeSutter 2003).

This gives the appearance of the test "homing in" on the correct answer. A 24-h moving average showed no trends with time, and at the end of the test the average was 0.65 mm day^{-1}. The graphical analysis presented in Figure 5-8 is an excellent way to judge data quality.

Estimating uncertainty is an important aspect of any seepage test. Ham (2002b) used a first-order error analysis to approximate the most probable uncertainty in S as estimated using water balance techniques of Ham (1999). It is possible to perform a site-specific uncertainty analysis for any test. However, Ham (2002b) showed that uncertainty in S is mainly a function of the average evaporation rate. The 95% confidence interval on S can be approximated as [uncertainty in S (mm day^{-1}) $=$ 0.08 * evaporation (mm day^{-1}) $+$ 0.14].

5.6 ALTERNATIVE TECHNIQUES FOR MEASURING WATER BALANCE AND SEEPAGE

The data shows that accuracy of the S determination often hinges on the measurement of E. In some cases, E might be determined using other options. If a plastic-lined lagoon is located near the study site, measuring ΔD from that facility might provide a good estimate of E for all lagoons in the area (assuming the plastic liner is not leaking). This would be analogous

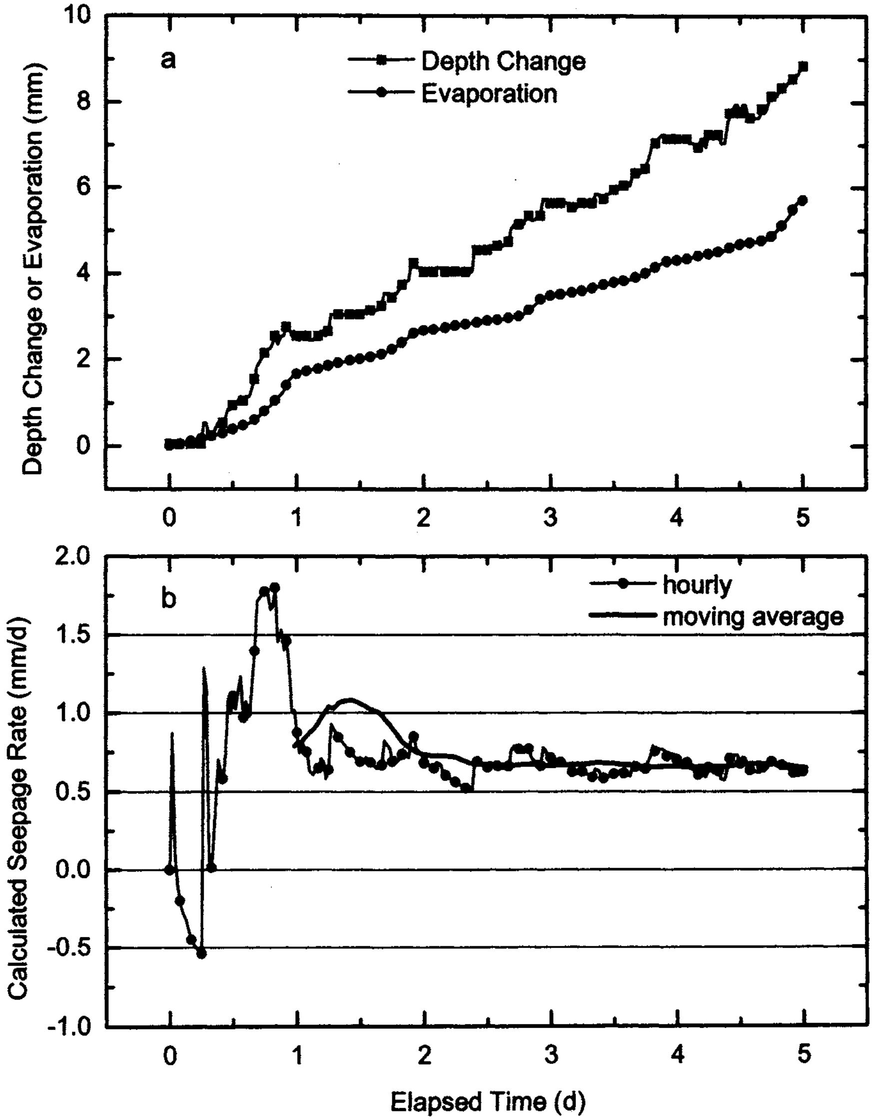


FIGURE 5-8. Water Balance and Calculated Seepage Rate for a Swine Waste Lagoon. Shown are (a) The Total Depth Change and Cumulative Evaporation for a 5-day Test, and (b) The Calculation of the Apparent Seepage Rate as Time Progressed [Seepage Rate = (Total Depth Change – Cumulative Evaporation)/ Elapsed Time]. (Source: Adapted from Ham and DeSutter 2003).

to using the plastic-lined lagoon like a big evaporation pan. If the lagoon being studied were small, it might also be possible to stop evaporation by covering the lagoon surface with plastic or biodegradable foam. In this case, measurements of ΔD would be equal to S. Finally, in some cases, E

might be measured using micrometeorological methods like the Bowen Ratio energy balance technique, eddy correlation, or conditional sampling. Unfortunately, all of these methods have extensive fetch requirements and can be used only on large lagoons. Even if instruments were positioned close to the surface (e.g., 1.5 m) on the downwind edge of a lagoon, upwind fetch would need to be greater than 50 m during neutral and unstable boundary conditions and much larger than 50 m in stable conditions (which are common over lagoons). Furthermore, complex airflow patterns over embankments and nearby structures can further complicate the use of micrometeorological methods.

Some researchers have attempted to measure S more directly using point seepage meters (e.g., Lee 1977; White and Denmead 1989). Seepage is quantified by measuring the water loss rate from a container driven into the bottom of the lagoon or pond. One of the main problems with this method is dealing with the tremendous amount of methane and carbon dioxide gas bubbles that evolve from the bottom sludge in an anaerobic lagoon. This makes it difficult to use pressure transducers or capillary scales to quantify the small volume of water actually seeping through the liner. Furthermore, the logistics of installing these containers and getting a good seal at the liner can be problematic. This technique is used mainly in lakes and oceans where an organic sludge layer is not present.

5.7 EVALUATING THE IN SITU PERFORMANCE OF A COMPACTED LINER AND CHEMICAL EXPORT

Once whole-lagoon seepage has been measured in the field, the results can be used to evaluate the performance of the compacted liner and determine if it is meeting the design criteria. Coupling seepage results to liner hydraulic properties requires assumptions regarding saturated flow. If vertical, steady-state flow and zero pressure potential on the bottom of the liner are assumed (Kisch 1959; Rowe et al. 1995), then flow at some horizontal position, x,y, in the lagoon can be described with Darcy's law:

$$q(x,y) = -K_s\left(\frac{H}{L}+1\right) \tag{5-10}$$

where

q = flux (m s^{-1})
K_s = coefficient of permeability (m s^{-1})
H = depth of waste (m)
L = thickness of the compacted liner (m)

All values are at position x,y. The seepage rate from the whole lagoon is related to the total flux across the liner:

$$Q = S'A \iint q(x,y)dxdy \tag{5-11}$$

where

Q = whole-lagoon seepage rate ($m^3\ s^{-1}$)
S' = seepage rate expressed as a rate change in depth ($m\ s^{-1}$)
A = area of the waste surface (m^2)

Assuming K_s is homogeneous and constant, then it can be removed from the integral, and the right-hand side of Eq. (5-11) is determined essentially by the geometry of the lagoon basin and depth of waste. Calculating the solution to the right-hand side of Eq. (5-11) can be cumbersome if the lagoon has a strange shape or unknown features. However, for most lagoons, the critical step is accounting for expected flux through the side embankments. Many modern lagoons are built with 3:1 or 4:1 side slopes to reduce the effect of water and wave erosion on the shoreline. Because of these small slopes, the side embankments may constitute a large portion of the submerged area. Ham et al. (1999) found that side walls accounted for 78% to 40% of the total area in a group of lagoons recently constructed in Kansas. Eq. (5-9) can be expanded for any lagoon shape to form an expression for flow through each sidewall of the facility plus the bottom. For a simple rectangular lagoon with a flat bottom and four trapezoid-shaped, planer side panels, Eq. (5-11) can be reduced to a simple form:

$$S' = \frac{K_s}{A}\left[A_s\left(\frac{H'}{2L}+1\right)+A_b\left(\frac{H'}{L}+1\right)\right] \tag{5-12}$$

where

H' = waste depth above the bottom of the lagoon (deepest point)
A_s, A_b = areal areas of the submerged side embankments and flat bottom, respectively

Eq. (5-12) assumes that the vertical path lengths through the liner are the same on the sides and bottom, and that waste depth integrated over the side panels is equal to $H/2$. Once Eq. (5-11) has been customized for a given site, one can (1) input S', A_s, A_b, H', and L and solve for the apparent K_s of the whole lagoon, or (2) input a laboratory-derived value of K_s and solve for the expected S'. In addition, once an apparent K_s for a lagoon is obtained from the seepage data, a formula like Eq. (5-12) can be used to predict how S would be affected by changes in waste depth. This may be important, because regulations may stipulate that S should be less than some mandated value when a lagoon is filled to capacity. However, most whole-lagoon seepage tests are not performed at maximum capacity; thus, the determination of the in-situ K_s allows the results from the field test to be scaled to that expected at maximum depth. Additionally, forms of Eq. (5-12) can be used to partition seepage losses between the sidewalls and bottom of the lagoon.

Table 5-4 shows an example of measured and predicted seepage losses from five swine waste lagoons in Kansas (Ham et al. 1999). All the lagoons were less than 5 yr old and had liners between 0.3 and 0.45 m thick. In some cases, the laboratory-derived K_s were obtained from intact cores collected after construction but before the addition of waste. In other cases, K_s was measured from a disturbed, recompacted sample of the liner material. Results show that in the three lagoons with the highest laboratory K_s (1, 2, and 5), the measured whole-lagoon S values were significantly lower than those predicted by a formula (Eq. (5-12)). Conversely, in the two lagoons with a very low laboratory K_s, measured S values were greater than predicted. These results parallel laboratory studies of Hills (1976) and Maulé et al. (2000), who showed that clogging of soil pores by

TABLE 5-4. Properties of Compacted Soil Liners at Five Swine Waste Lagoons, and Comparison of the Predicted and Measured Seepage Rates from Soil Cores Collected Prior to the Addition of Waste

	Soil Texture			CEC[a]	K_s[b]	Seepage (mm/day^{-1})	
Lagoon	% Sand	% Silt	% Clay	(cmol/kg^{-1})	(cm/s^{-1})	Predicted	Measured
1	40.0	30.0	30.0	32	7.6E-7[c]	5.4	1.4
2	52.5	21.3	26.3	26	3.7E-7[d]	5.2	2.0
3	58	20	22[e]	15	4.4E-8[c]	0.4	0.8
4	52.5	21.3	26.3	26	<1.0E-8[d]	0.1	0.9
5	75	11.3	13.8	15	5.0E-7[d]	3.8	1.5

Notes:

[a] CEC = Cation exchange capacity.

[b] K_s = Coefficient of permeability.

[c] Average of four to six soil samples collected from the liner after lagoon construction but prior to the addition of waste.

[d] Measured from a single recompacted core of soil slated for use during construction (not sampled from the actual compacted liner).

[e] Liner was 0.46 m thick, but the lowermost 0.15-m layer was augmented with bentonite. Soil used for texture and CEC analysis was collected from the uppermost 0.38 m and may not accurately reflect the effect of the bentonite.

organic sludge tends to be more pronounced in coarser textured soils but has little effect on liners with an initially low K_s. The higher-than-expected seepage in lagoons 3 and 4 may have been caused by increased liner permeability along the side embankments (e.g., erosion, wetting and drying). However, a weakness of this type of analysis is that it does not consider how positional variation in K_s may be affecting the results. Nevertheless, Table 5-4 shows how seepage measurements can be combined with other data to evaluate the in-situ performance of the liner. Using this approach, Ham (2002a) found that the average compacted liner K_s at 20 animal waste lagoons in Kansas was 1.8×10^{-7} cm s^{-1} and that K_s followed a log-normal distribution among sites.

Measurements of S can be combined with analysis of waste chemistry to estimate the rate at which soluble nutrients and salts are lost from the lagoon. The chemical export rate is essentially the product of the seepage rate (i.e., q in m s^{-1}) and solute concentrations in the effluent (kg m^{-3}). Table 5-5 shows the ammonium-nitrogen ($NH_4^+ - N$) export rate from the five lagoons described in Table 5-4. Values ranged from 0.2 to 0.5 kg $NH_4^+ - N$ m^{-2} yr^{-1}, reflecting the combined effect of site-to-site variations in waste chemistry and S. Export rates can be calculated on an instantaneous basis (kg m^{-2} s^{-1}) or extrapolated to longer periods (e.g., kg ha^{-1}/yr^{-1}). When attempting to approximate annual losses, seasonal variations in chemistry as well as depth changes in the lagoon should be considered. Concentrations of many compounds are affected by climate and temperature-induced changes in microbial activity. Thus, waste should be analyzed at least four times a year, and samples should be taken at multiple depths to check for vertical stratification. Fluctuations in lagoon depth also will affect chemical export by influencing S and A. Again, a one-time field measurement of the apparent K_s (via a seepage

TABLE 5-5. Rate of Subsurface NH_4-N Export into the Underlying Soil

Lagoon	Type	NH_4-N conc. ($mg\ L^{-1}$)	NH_4-N Export Rate[a] kg/m^{-2}/yr^{-1}	lb/$acre^{-1}$/yr^{-1b}
1	Swine, nursery	565	0.289	2,574
2	Swine, sow	685	0.500	4,453
3	Swine, finishing	824	0.241	2,143
4	Swine, sow	685	0.225	2,004
5	Swine, finishing	824	0.451	4,018

Notes:

[a] Export rates were computed based on the measured seepage rates (Table 5-4) and NH_4-N concentrations in the effluent.

[b] Shown in units consistent with conventional application of agricultural fertilizer.

experiment) can be used with Eq. (5-11) and long-term records of lagoon depth (i.e., H') to approximate seasonal variations in S. Chemical export then can be calculated by season and summed over the year to approximate annual losses. Although assumptions are required to approximate chemical export, this data is invaluable when attempting to model how lagoons affect groundwater quality (Ham and DeSutter 1999). Ham (2002a) collected soil cores beneath animal waste lagoons and provides a description of the observed chemical profiles.

5.8 CONCLUSIONS

The use of anaerobic lagoons and their potential impact on groundwater quality are likely to be areas of concern as long as AFOs use earthen structures to store manure waste. It follows that techniques are needed to measure the in-situ performance of existing lagoons rather than assume that they are meeting the engineered seepage rate. Water balance methods appear to be a viable way to assess the overall performance of a lagoon in the field. Results show that seepage can be estimated to within about ± 0.2 mm day^{-1} by measuring evaporation and depth changes over brief measurement cycles (e.g., 5 to 10 days). In many cases, the accuracy of the seepage estimate will depend on the techniques used to measure E as well as its magnitude. Clearly, periods of low evaporative demand ($E < 5$ mm day^{-1}) are the best times to attempt water balance experiments. Despite some uncertainty, a standard protocol for measuring S could be very useful in both research and industry (Ham and DeSutter 2003). For example, seepage data could be used by consulting engineers to verify that a lagoon is operating within compliance or by a research laboratory to test the performance of liner construction methods. Accurate estimates of seepage can be combined with information on lagoon chemistry to approximate the rate of chemical movement into the underlying soil.

This chapter was originally written in 1999. Since then, there have been many new developments in lagoon research, and a few key areas in the text have been updated. However, the reader is encouraged to review more recent papers on lagoon seepage and its impact on groundwater quality. Recommended reading includes the following: Parker et al. 1999; Ham and DeSutter 2000, 2003; Maulé et al. 2000; Glanville et al. 2001; and Ham 2002a, 2002b.

Chapter 6

USE OF COAL COMBUSTION BY-PRODUCTS AS LOW-PERMEABILITY LINERS FOR MANURE STORAGE FACILITIES

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The construction of animal manure storage facilities typically requires the installation of a low-permeability liner to separate the liquid manure from the surrounding environment. Liners for manure storage facilities in the United States have generally been constructed of compacted onsite clay. However, in areas where abundant onsite natural clay in not available, livestock producers must have additional options. The producer can construct the liner using borrow clay, concrete, geomembranes, or geosynthetic clay liners, all of which have a significant cost. An alternative economical option that may be available to producers is the use of flue gas desulfurization (FGD) material in the construction of compacted liners. Research conducted at The Ohio State University (Wolfe et al. 2000) has shown that FGD material can be used as a low-cost alternative in place of synthetic clay, geomembrane, and concrete liners.

This chapter presents an overview of the use of compacted FGD material as a low-permeability liner. It discusses the engineering characteristics of FGD materials and the effect of freeze-thaw cycling on engineering properties. The design, construction, and monitoring of a full-scale FGD-lined demonstration facility is described. The facility held water for the first year, after which swine manure was added. Results of the monitoring program (permeability and water quality testing) for the facility are presented. The advantages and disadvantages of using FGD material for constructing low-permeability liners are discussed.

6.1 WHAT ARE COAL COMBUSTION BY-PRODUCTS?

More than half of the electricity generated in the United States is produced by coal-fired facilities. The Clean Air Act Amendments of 1990 require many utilities that burn high-sulfur bituminous coal to reduce sulfur dioxide emissions. This has resulted in the generation of large amounts of coal combustion by-products (CCBs). These by-products include fly ash, bottom ash, boiler slag, and FGD material. In 2001, more than 1,000 million tons of coal was burned in the United States to produce electricity. This led to the generation of more than 120 million tons of CCBs (American Coal Ash Association 2002).

The different types of CCBs generated from the combustion of coal and their characteristics, texture, production rate, and major constituents are shown in Table 6-1. Fly ash is the noncombustible particulate matter that is removed from the stack gases (typically 160 lb/ton of coal). Fly ash is a

TABLE 6-1. Types of Coal Combustion By-Products and Their Characteristics

CCB Type	Characteristics	Texture	Amount Typically Generated per ton of Coal Burned (lb)	Major Constituents
Fly ash	Noncombustible particulate matter removed from stack gases	Powdery, silt-like	160	Si, Al, Fe, Ca
Bottom ash	Material collected in dry-bottom boilers, heavier than fly ash	Sand-like	40	Si, Al, Fe, Ca
Boiler slag	Material collected in wet-bottom boilers or cyclone units	Glassy, angular particles	100	Si, Al, Fe, Ca
FGD material	Solid/semi-solid material obtained from flue gas scrubbers	Fine to coarse (dry or wet)	350	Ca, S, Si, Fe, Al*

Source: Butalia et al. 1999.

Note:

*Stabilized FGD is a mixture of filter cake (Ca, S), fly ash (Si, Fe, Al), lime, and water. Major constituents of gypsum-quality FGD are Ca and S.

pozzolanic material, i.e., in the presence of lime and water it forms calcium silicate cementious compounds. Fly ash can be classified into two categories: Class C and Class F. Class F fly ash typically has a calcium oxide content less than 10% and almost no free lime, while Class C fly ash has a calcium oxide content greater than 20%, contains 1% to 3% free lime, and is self-cementing in the presence of water. Class C fly ash is generally obtained by combustion of western sub-bituminous and lignite coals, while Class F fly ash is typically generated from eastern bituminous and anthracite coals. Bottom ash refers to the heavier ash particles collected at the bottom of dry-bottom boilers. Typically, about 40 lb of bottom ash is generated per ton of coal from dry-bottom pulverized coal boilers. The glassy, angular ash particles collected at the bottom of wet-bottom boilers or cyclone units are referred to as boiler slag.

A large amount of FGD material is generated during the removal of sulfur dioxide from flue gases during the scrubbing process. Utilities in the United States generate approximately 25 to 30 million tons of FGD material annually. The type of FGD material generated from a coal-fired scrubber depends on several factors, including the type of coal and boiler, reagent used for desulfurization, and FGD process. FGD by-product may be dry or wet, depending on the desulfurization process. At present, dry FGD processes are more commonly used in smaller electricity-generating facilities. The wet scrubbing process, which is commonly used by large electric utilities, involves the injection of a reagent (typically hydrated quicklime, limestone, or dolomite) into the flue gases. The wet filter cake generated is a mixture of sulfites and sulfates of the reagent, unreacted reagent, and water. Calcium sulfite content is typically greater than 70%, while the calcium sulfate content is approximately 10% to 15%, unless the filter cake is oxidized. Fly ash and additional quicklime can be added to stabilize the FGD filter cake. The amount of fly ash and lime to be added to the FGD filter cake depends on the particular end use intended for the stabilized FGD material. Increasing the fly ash and lime content results in increased strength and durability of the stabilized FGD material. Typically, stabilized FGD material is gray in color and looks like silty clay. The typical chemical composition of stabilized FGD material is shown in Table 6-2. Stabilized wet FGD material mainly consists of oxides of calcium, sulfur, silica, and aluminum. Typical results of toxicity characteristic leaching procedure (TCLP) tests, conducted in accordance with U.S. Environmental Protection Agency (USEPA) Method 1311 on wet stabilized FGD by-products, are shown in Table 6-3. Also presented in Table 6-3 are the national primary and secondary (non–health-related) drinking water standards. It can be observed that the leachate from FGD material meets all the primary drinking water standard regulated elements. With the exception of pH and total dissolved solids, FGD leachate also meets the secondary drinking water standards. Secondary standards are unen-

TABLE 6-2. Typical Chemical Composition of Stabilized Wet FGD By-Product

Constituent	Percentage by Weight
Silica (SiO_2)	20.3
Aluminum oxide (Al_2O_3)	12.4
Iron oxide (Fe_2O_3)	7.6
Titanium oxide (TiO_2)	0.7
Calcium oxide (CaO)	25.6
Magnesium oxide (MgO)	1.7
Sodium oxide (Na_2O)	0.3
Potassium oxide (K_2O)	0.9
Sulfur trioxide (SO_3)	30.0
Phosphorus pentoxide (P_2O_5)	0.1
Barium oxide (BaO)	<0.1
Manganese oxide (Mn_2O_3)	<0.1
Strontium oxide (SrO)	<0.1
Total carbon (C)	0.4
Total	**100.00**

forceable federal guidelines regarding non–health-related concerns (e.g., taste, odor, color, and certain other nonaesthetic effects of drinking water). Thus, it can be safely concluded that the leachate from wet fixated FGD materials generally meets national drinking water standards and that the FGD by-product is "non-toxic."

6.2 ENGINEERING CHARACTERISTICS OF FGD MATERIALS

Several researchers at The Ohio State University participated in a long-term study aimed at characterizing the physical, chemical, mineralogical, and engineering properties of FGD material and its land application (Stehouwer et al. 1995, 1998; Dick et al. 1998, 1999). Representative samples of FGD were collected from 13 different coal-fired boilers for detailed analysis. The technologies included lime injection multistage burners (LIMB), pressurized fluidized bed combustion (PFBC), spray dryer, and duct injection. The engineering properties of compacted FGD that were studied included ASTM D 698, optimum moisture content and maximum dry density (ASTM 1998a); ASTM D 2166, unconfined compressive

TABLE 6-3. Typical TCLP Results for Stabilized Wet FGD By-Product

Constituent	Drinking Water Standard	Range	Average
Acidity (mg/L)	N/A[a]	<1–<1	<1
Alkalinity (mg/L)	N/A[a]	85–811	471
Aluminum (mg/L)	.05–.2[c]	.12–2.25	.87
Arsenic (mg/L)	.05[b]	.008–.033	.016
Barium (mg/L)	2.0[b]	.044–.436	.271
Cadmium (mg/L)	.005[b]	<.0005–<.0005	<.0005
Chloride (mg/L)	250[c]	51–67	59
Chromium (mg/L)	.1[b]	<.002–.007	.004
Conductivity (μmhos/cm)	N/A[a]	603–3480	2333
Copper (mg/L)	1.0[c]	.001–.002	.001
Fluoride (mg/L)	2.0[c]	.1–.8	.6
Iron (mg/L)	.3[c]	<.01–.05	.02
Lead (mg/L)	.015[d]	<.002–.002	.001
Manganese (mg/L)	.05[c]	<.01–<.01	<.01
Mercury (mg/L)	.002[b]	<.0002–.0004	.0002
pH	6.5–8.5[c]	11.22–11.70	11.45
Selenium (mg/L)	.05[b]	.005–.026	.011
Silver (mg/L)	.1[c]	<.0002–.001	.0004
Sodium (mg/L)	N/A[a]	5.0–7.4	6.0
Sulfate (mg/L)	250[c]	34–539	156
Total dissolved solids (mg/L)	500[c]	454–881	623
Vanadium (mg/L)	N/A[a]	.006–.024	.013
Zinc (mg/L)	5.0[c]	.008–.031	.018

Notes:
[a] N/A = Not applicable.
[b] Primary drinking water standard.
[c] Secondary (non–health-related) drinking water standard.
[d] Action level.

strength (ASTM 1998b); ASTM D 2435, compressibility (ASTM 1996a); permeability (Lin and Evett 1990); and ASTM D3877, swelling potential (ASTM 1996b).

Table 6-4 presents the range of engineering properties for the dry FGD materials investigated by Stehouwer et al. (1995). For all FGD samples, it

TABLE 6-4. Range of Engineering Properties of Dry FGD By-Products

Property	Range
Optimum moisture content	18%–68 %
Maximum dry density	0.7–1.9 g/cm^3
Unconfined compressive strength (at 28 days)	30–1,530 psi
Compressibility	C_c = 0.01–0.05 C_r = 0.002–0.005
Permeability (at 28 days)	10^{-6}–10^{-10} cm/s
Swell	0.6%–70%

Source: Stehouwer et al. 1995.

was observed that the slopes of moisture-density curves were small, indicating that the range of moistures over which optimum density can be obtained is fairly broad. Thus, the maximum dry density is not particularly sensitive to slight variations in the water content in the vicinity of the optimum. The FGD samples (except those obtained from the PFBC process) exhibited compacted maximum dry densities lower than a typical natural soil. The unconfined compressive strength of compacted FGD samples increased with curing time. As a class, compacted FGD materials possess higher strengths compared to compacted clay. A limited number of one-dimensional compression tests on FGD samples indicated that FGD materials are much less compressible than typical natural materials, such as soft and stiff clays. The permeability of compacted FGD samples decreased with time. The permeability of FGD samples at 28 days of curing ranged from 10^{-6} cm/s (low permeability) to 10^{-10} cm/s (practically impervious). The percentage swell for FGD samples was observed to be 0.6% to 70%. Two distinct swelling episodes were apparent. The first episode occurred immediately after water was supplied to the specimen. This corresponded with the time during which naturally occurring soils typically experience the greatest volume increase. The second episode of swelling began after 10 or more days had elapsed. FGD materials with low permeability, high strength, and low swell potential are ideally suited for liner applications.

Laboratory evaluation of the hydraulic conductivity and strength characteristics of lime-enriched, stabilized wet FGD materials have been presented by Butalia and Wolfe 1998. Table 6-5 shows some of the laboratory test results presented by Butalia and Wolfe and some additional tests that were conducted on compacted sulfite-rich FGD samples. Two laboratory samples (66-34-5 and 66-34-8) were prepared in the laboratory by mixing fly ash (FA) and filter cake (FC) in approximately 2:1 ratio (dry

TABLE 6-5. Laboratory-Compacted FGD Material Samples

Sample	Moisture Content (%)	Dry Density (kN/m^3)	Coefficient of Permeability (cm/s) 7-Day	28-Day	60-Day	90-Day	q_u (psi) 90 Day
66-34-5 (FA-FC-L)	27	12.4	3.6×10^{-5}	3.2×10^{-6}	1.5×10^{-6}	1.4×10^{-6}	242
66-34-8 (FA-FC-L)	32	12.2	1.4×10^{-5}	1.2×10^{-6}	1.3×10^{-7}	4.8×10^{-8}	343
CON(AEP)-5%L	48	11.1	2.6×10^{-6}	3.6×10^{-7}	3.2×10^{-7}	2.6×10^{-7}	240
CON(AEP)-8%L	42	11.9	1.3×10^{-6}	6.1×10^{-8}	3.4×10^{-8}	3.0×10^{-8}	629
GAV(AEP)-4%L	51	10.8	2.2×10^{-6}	2.2×10^{-7}	1.5×10^{-7}	1.7×10^{-7}	371
GAV(AEP)-8%L	39	12.5	1.6×10^{-6}	1.2×10^{-7}	2.0×10^{-8}	1.0×10^{-8}	607

Notes:
AEP = American Electric Power; CON = Conesville plant; FA = Fly ash; FC = Filter cake; GAV = Gavin plant; L = Lime.

weight basis). Samples 66-34-5 and 66-34-8 had lime contents (dry weight basis) of 5% and 8%, respectively. The moisture contents listed for the laboratory-mixed samples are the optimum moisture contents so as to achieve maximum dry density (as per ASTM D 698). The CON(AEP)-5%L and CON(AEP)-8%L samples were obtained from American Electric Power's (AEP) Conesville (CON) power plant near Coshocton, Ohio, while the GAV(AEP)-4%L and GAV(AEP)-8%L samples were obtained from AEP's Gavin (GAV) plant near Gallipolis, Ohio. These samples were prepared at the respective power plants instead of being mixed in the laboratory. The lime percentages of 4%L, 5%L, and 8%L are as estimated on a dry weight basis by the plant operators. The CON and GAV samples were compacted using Standard Proctor test guidelines at as-received moisture contents. Table 6-5 shows that moisture contents of the samples received from the power plants were higher than the optimum moisture contents obtained in the laboratory. Consequently, the dry densities obtained by compacting these samples were lower than the maximum dry densities obtained from the laboratory-mixed samples. However, the coefficient of permeability, which was measured as a function of curing time (7, 28, 60, and 90 days) using a falling-head test (Lin and Evett 1990) is lower for the plant-mixed samples than for the laboratory samples. For the plant mixed samples, the permeability values are in the range of 10^{-7} to 10^{-8} cm/s at 28 days of curing. Samples with higher lime contents resulted in lower coefficients of permeability, as well as higher unconfined compressive strengths. Table 6-5 shows that the permeability and strength characteristics of FGD materials generated at the Conesville and Gavin plants were similar. The 8% lime samples have the lowest permeability values that come close to 10^{-8} cm/s. It can be concluded from Table 6-5 that FGD material can be compacted in the laboratory using standard soil testing procedures to obtain permeability coefficients that are in the range of 10^{-7} to 10^{-8} cm/s, which are lower than the 10^{-6} to 10^{-7} cm/s value typically required for constructing liners for manure storage facilities.

Liners constructed in many parts of the United States experience weather-related freezing and thawing. This can have a detrimental effect on the integrity of the liner. Hargraves (1994) and Chen et al. (1997) investigated the effect of thermal cycling on the strength of compacted wet FGD material. Higher water content samples exhibited greater reduction in compressive strength due to freeze-thaw cycling. It was observed that high strengths could be maintained under freeze-thaw cycling if at least 5% lime (dry weight basis) was added to the FGD material before compaction and the material was allowed to cure for 60 days before being exposed to freeze-thaw. These general recommendations are now used as a general rule for the structural use of FGD material in Ohio.

6.3 FULL-SCALE FGD-MATERIAL–LINED TEST FACILITY

Permeability of a field-compacted FGD structure is likely to be a function of the construction process, and hence the field validation of the properties obtained in the laboratory is an important part of the documentation process. This section presents the design, construction, and monitoring of a full-scale testing facility to evaluate the performance of a field-compacted FGD material liner. The full-scale facility was constructed to address two critical issues about the behavior of stabilized FGD products constructed in the field: (1) the permeability of a compacted engineered liner of known thickness and density, and (2) the quality of the water that flows through the FGD material liner.

6.3.1 Design of Facility

The full-scale facility was designed and constructed at The Ohio State University's Ohio Agricultural Research and Development Center (OARDC) Western Branch in South Charleston (Clark County), Ohio. This site was chosen over other university sites because it had an abundance of clay onsite that was suitable for use as a secondary or outer liner to contain the primary FGD liner. The OARDC Western Branch facility is a swine and agronomic research facility and, hence, it was decided to build a livestock manure storage facility that could be used by the center for storing swine manure after the completion of this research. The facility was designed for a capacity of approximately 1 million gal. (150,000 ft^3) to provide 6 months of storage for all liquid wastes from the swine onsite. A double-layered design was chosen with compacted stabilized FGD material as the primary inner liner and the onsite clay (about 80 ft of gray glacial till) as the secondary outer liner. A leachate system was designed to be placed between the primary FGD material liner and secondary clay liner to collect in a sump any water passing through the FGD material fill. The sump was designed so that it could be used to collect leachate samples with ease and to conduct field permeability tests on the pond liner.

The facility is essentially rectangular in shape with overall dimensions of approximately 150 ft × 250 ft (including 8-ft-wide berms), as shown in Figure 6-1. Three sides of the pond were constructed at 3:1 slope and the fourth (east) side at 7:1 slope. The east-side slope was designed to be less steep to allow for easy access to the pond bottom during and after construction. Cross-sections AA and BB, which are shown in Figures 6-2 and 6-3, respectively, show the final elevations of the facility. As illustrated, the pond is 9 ft deep with a liquid freeboard of 2 ft. A berm of minimum 8-ft top width was added around the periphery of the pond to minimize the inflow of surface water. The natural clay at the site provided an outer liner that was at least 5 ft thick. The leachate collection system, which

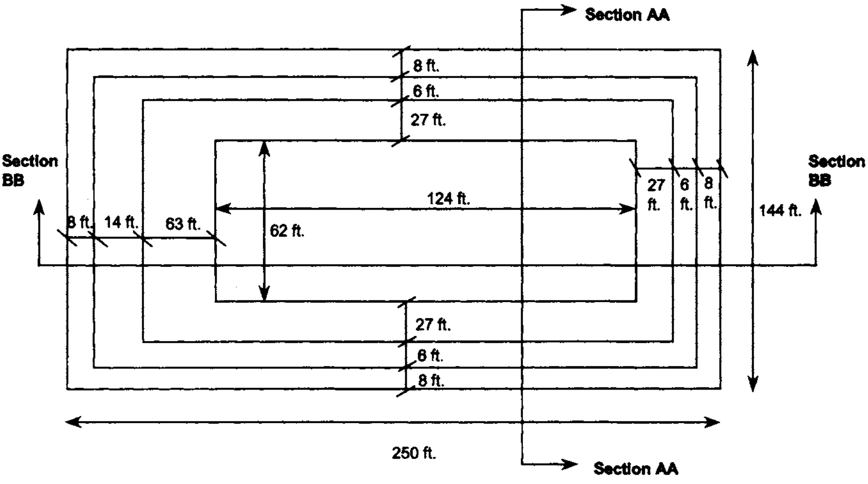


FIGURE 6-1. Plan View of Facility (not to scale).

consisted of corrugated high-density polyethylene (HDPE) perforated pipes (with socks) and protected against crushing using #57 washed river gravel, was placed over the recompacted clay. The bottom of the pond was then covered with 9 in. of sand. On top of the sand layer, an 18-in.-thick layer of compacted FGD material was placed. A plan view of the leachate collection system is shown in Figure 6-4. A typical detail of the perforated pipe embedded in the sand layer is shown in Figure 6-5. The proposed design of the facility was submitted to Ohio Environmental Protection Agency (EPA) for review and approval. Prior to project construction, a Permit to Install was issued by Ohio EPA for the installation of the demonstration facility (Ohio EPA 1997).

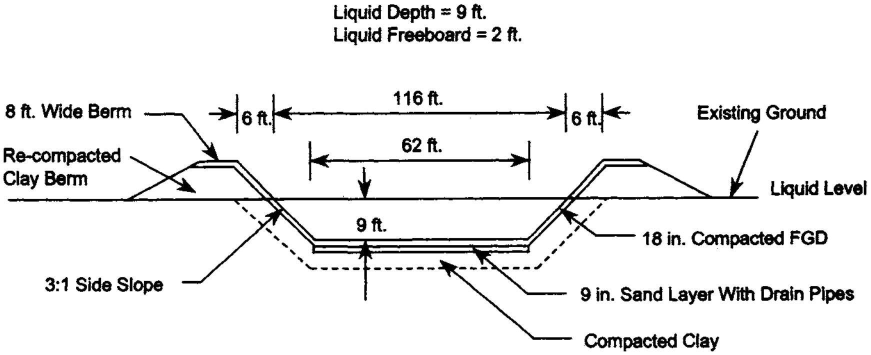


FIGURE 6-2. Section AA of Facility (not to scale).

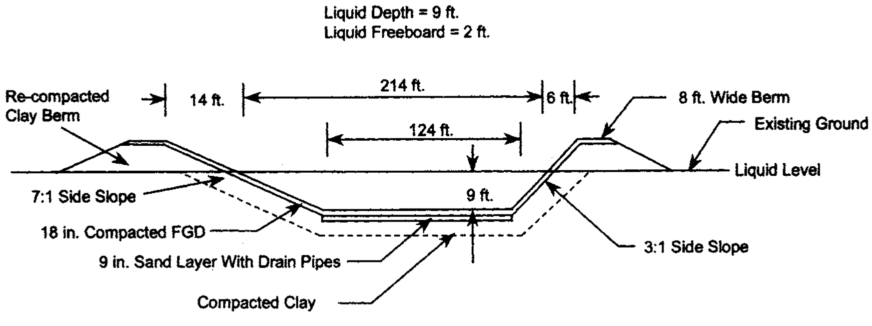


FIGURE 6-3. Section BB of Facility (not to scale).

6.3.2 Construction of Facility

Excavation of the site began on July 30, 1997. The top layer of soil containing organic matter was removed and hauled away from the site. The recompaction of onsite glacial till to form the secondary clay liner was completed on August 7, 1997. The onsite glacial till clay had an average moisture content of 11.6% and maximum proctor dry density of 18.6 kN/m^3 (118 lb/ft^3). A sheepsfoot roller (with vibration) was used to compact the onsite clay (see Figure 6-6) to 99.4% of the proctor density. The laboratory permeability of the onsite compacted clay was measured to range between 3.04×10^{-7} cm/s (9.97×10^{-9} ft/s) and 7.24×10^{-8} cm/s

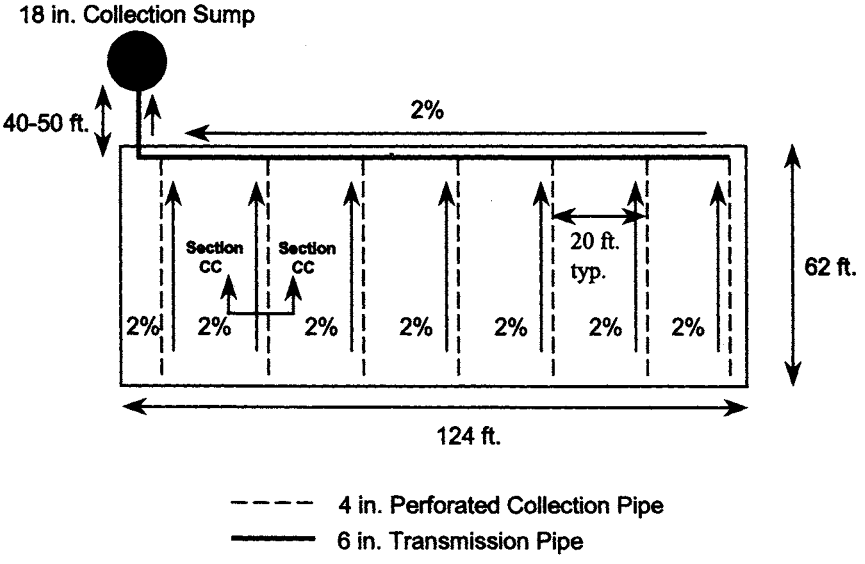


FIGURE 6-4. Leachate Collection System Layout (not to scale).

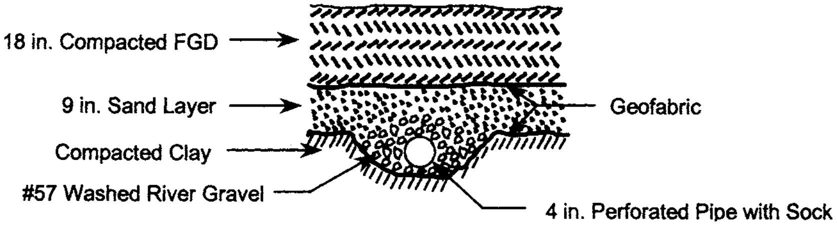


FIGURE 6-5. Section CC of Facility (not to scale).

(2.37×10^{-9} ft/s). The locations of the leachate collection system pipes were marked and excavated to a trench depth of 6 to 8 in. A geofabric (to separate the secondary clay liner and sand particles) was spread over the clay with at least 1 ft of overlap at the geofabric joints. The leachate system pipes were then placed over the geofabric (see Figure 6-7). The leachate collection system was connected to an 18-in.-diameter vertical sump (20 ft height). About 50 tons of crushed #57 washed river gravel was then placed around and on top of the leachate collection system pipes to avoid crushing due to later compaction of FGD on top of it. The crushed river gravel can be seen on the right-hand side of Figure 6-8. The bottom of the facility was then covered with approximately 300 tons of silicious round

FIGURE 6-6. Compaction of Onsite Clay.

FIGURE 6-7. Typical Leachate System Collection.

FIGURE 6-8. Spreading of Sand.

natural fine sand (Figure 6-8). The permeability of the sand used in the drainage layer was evaluated in the laboratory to be 3.12×10^{-3} cm/s (1.02×10^{-4} ft/s). A layer of geofabric was laid over the sand layer.

Lime– and fly-ash–enriched stabilized FGD material was delivered to the site by truck (starting August 11, 1997) from AEP's Conesville station near Coshocton, Ohio (Figure 6-9). Placement and compaction of FGD in 4- to 6-in. lifts on top of the geofabric layer were accomplished using two dozers and one sheepsfoot roller (see Figures 6-10 and 6-11). Approximately 2,700 tons of lime-enriched stabilized FGD material was used in the construction of the primary liner. The ratio of fly ash to filter cake of the FGD material ranged from 1.48:1 to 2.40:1, with an average ratio of 1.81:1. The lime content varied from 6.79% to 8.44%, with an average lime content of 7.98%. The moisture content of the FGD material received at the site during construction ranged from 49% to 62%, while the proctor dry density varied between 9.6 kN/m^3 (61 lb/ft^3) and 11.6 kN/m^3 (74 lb/ft^3). Wet weather during the liner placement resulted in several delays, but construction at the site was completed by August 26, 1997. The site was smooth-rolled before completion of the project (Figure 6-12). Filling of the pond with water from an existing nearby pond began on September 12, 1997, and was completed on September 23, 1997. Figure 6-13 is a photograph of the partially filled facility. The pond was filled with water up to a depth of approximately 9 ft, as shown in Figure 6-14.

FIGURE 6-9. Truck Unloading FGD Material.

FIGURE 6-10. Spreading the FGD Material.

To avoid coring holes—to be used to obtain permeability samples—in the full-scale FGD material liner, several FGD material test pads (TP) were constructed in the vicinity of the full-scale facility. Four rectangular test pads (approximately 15 ft wide, 25 to 30 ft long, and 3 to 4 ft deep) were installed. Each test pad was initially backfilled with 6 in. of sand to provide a permeable layer for drainage. The remainder of the excavation was filled with stabilized compacted FGD material. Due to the small size

FIGURE 6-11. Compacting FGD Material on a Side Slope.

FIGURE 6-12. *Final Smooth Rolling of FGD Material.*

of the test pads compared to the large dimensions of the sheepsfoot roller, adequate compaction of the FGD material in the test pads could not be obtained. Three of the test pads (TP1, TP2, and TP3) were constructed using the 1.25:1 (FA:FC) and 8% lime mix, which was the typical mix used in the construction of the full-scale facility. TP4 was constructed using a

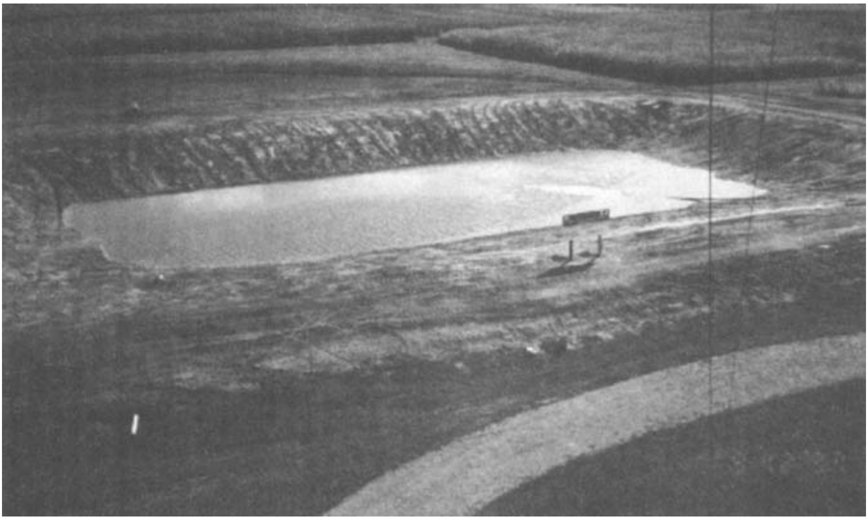


FIGURE 6-13. *Facility Being Filled with Water.*

FIGURE 6-14. Facility Filled with Water.

0.8:1 (FA:FC) and 4% lime FGD material mix, which is the typical material generated at the power plant. This weaker mix was not used in the construction of the full-scale facility. An outline of the test pads constructed is shown in Table 6-6. Each of the test pads was instrumented for Boutwell field permeability testing apparatuses (Boutwell 1992).

The actual cash cost of constructing the full-scale facility was estimated to be $46,623.50. The FGD material was made available at the site free of

TABLE 6-6. FGD Material Test Pads Constructed

		Stabilized FGD Material			
Test Pad Number	Thickness of Sand (in.)	Thickness (in.)	Approx. FA:FC Ratio	Approx. Lime Content (%)	Moisture Content (%)
TP1	6	27	1.25:1	8	62
TP2	6	36	1.25:1	8	69
TP3	6	30	1.25:1	8	58
TP4	6	36	0.8:1	4	84

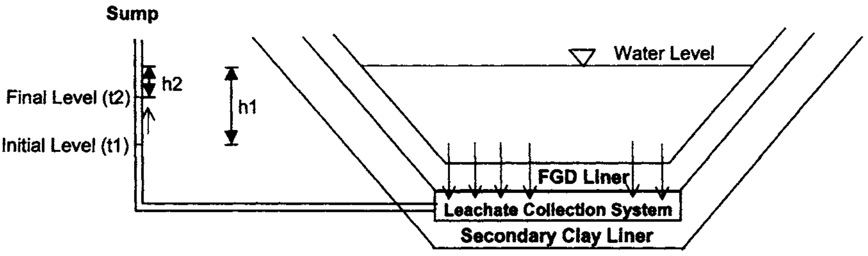
Notes:
FA = Fly ash; FC = Filter cake; TP = Test pad.

charge by AEP's Conesville power plant. Thus, the actual cash cost does not include the cost of adding additional lime to the material (approximately $13,500) and the associated transportation costs (estimated to be $25,000) for transporting the FGD material from Coshocton to the South Charleston project site. The cash cost of sand, gravel, drainage pipes, geofabric, and other miscellaneous materials was $4,567.50. The cash cost of construction of the facility was $42,056. The construction cost included equipment and operator costs for a backhoe (Kebelco 300), two bulldozers (Kamatsu D-6 and John Deere J 450), roller (sheepsfoot/smooth), trenching machine, Bobcat loader, two dump trucks, rotovator with farm tractor, and farm tractor with front-end loader.

6.3.3 Monitoring of Facility

The facility was used to store water for the first year. In early September 1998, some of the water was replaced with swine manure. Since then, swine manure has continued to be added and removed from the facility on a regular basis, depending on the manure storage versus field-spreading needs of the research farm. Monitoring of the site for field permeability and water quality was carried out for a period of 2 yr and 5 months (September 1997 through January 2000). The average monitoring frequency was 3 months. The monitoring program consisted of two main activities:

1. *Field Permeability Testing:* Full-scale falling head permeability tests on the facility were conducted by lowering the water level in the sump to create a head difference across the FGD material liner. The amount of time taken to increase the water in the sump to specific levels was observed. Knowing the thickness of the FGD material liner and its plan view area, the effective permeability of the field-compacted FGD-material–lined facility was calculated (refer to Figure 6-15).
2. *Water Quality Monitoring Program:* Testing of water samples from the pond, the sump, and an existing farm well about 1,000 ft from the site was carried out on a regular basis. Samples were collected in 250-ml polyethylene bottles that had been prewashed with acid and triple-rinsed with deionized water. Acidified samples were obtained by adding 2.5% nitric acid by volume to the samples in the field. Filtered samples were passed through a 0.45-μm filter. The water quality analysis was performed by the Star Laboratory of The Ohio State University's School of Natural Resources at OARDC, Wooster. Tests conducted on the water samples included pH, electrical conductivity, alkalinity, acidity, total dissolved solids, 24 elements by inductively coupled plasma (ICP) emission spectrometry mineral analysis, four anions using ion chromatography (IC) analysis, and ammonia, as well as nitrogen by Micro-Kjeldahl Distillation.



$$k = \frac{L}{(t2 - t1)} \frac{a}{A} \ln\left(\frac{h1}{h2}\right)$$

L = Thickness of FGD liner
a = Area of sump
A = Effective area of FGD liner

FIGURE 6-15. Full-Scale Permeability Test (not to scale).

Although the vicinity well was about 1,000 ft from the site and not necessarily hydrologically connected to the site, the well samples were investigated so that potential contamination of the farm water supply from the FGD-material–lined facility could be detected.

6.3.4 Monitoring Observations and Discussion

6.3.4.1 Permeability. After filling the full-scale facility with water, the actual field permeability of the FGD material liner was measured by lowering the water level in the sump and taking readings of the water level rising in the sump at various time intervals (refer to Figure 6-15). The permeability coefficients were calculated using the bottom area of the pond as the effective leaching area for the FGD material liner. Table 6-7 shows the effective coefficients of permeability obtained from full-scale permeability tests conducted on the pond facility. The permeability coefficient values listed in Table 6-7 are the average of several test readings that were measured at each curing time. The full-scale permeability of the facility was evaluated to be 9.1×10^{-7} cm/s (2.9×10^{-8} ft/s) at a curing time of 1 month. The permeability coefficient continued to reduce over time (due to curing of FGD material) and has stabilized at approximately 4×10^{-7} cm/s (1×10^{-8} ft/s). The range of FGD material permeability coefficient data obtained from the full-scale tests is comparable to typical clays used in the construction of compacted liners. The data presented in Table 6-7 includes the effect of freeze-thaw cycling on the actual permeability of the FGD material liner. The actual area over which water flows through the FGD material liner is greater than the bottom area of the pond (i.e., a

TABLE 6-7. Full-Scale Permeability Test Results

Curing Time (days)	Coefficient of Permeability* (cm/s)
31	9.1×10^{-7}
63	6.8×10^{-7}
153	4.1×10^{-7}
202	4.3×10^{-7}
317	3.8×10^{-7}
402	4.2×10^{-7}
456	3.9×10^{-7}
567	4.0×10^{-7}
693	3.8×10^{-7}
869	4.3×10^{-7}

1 cm/s = 0.0328 ft/s.

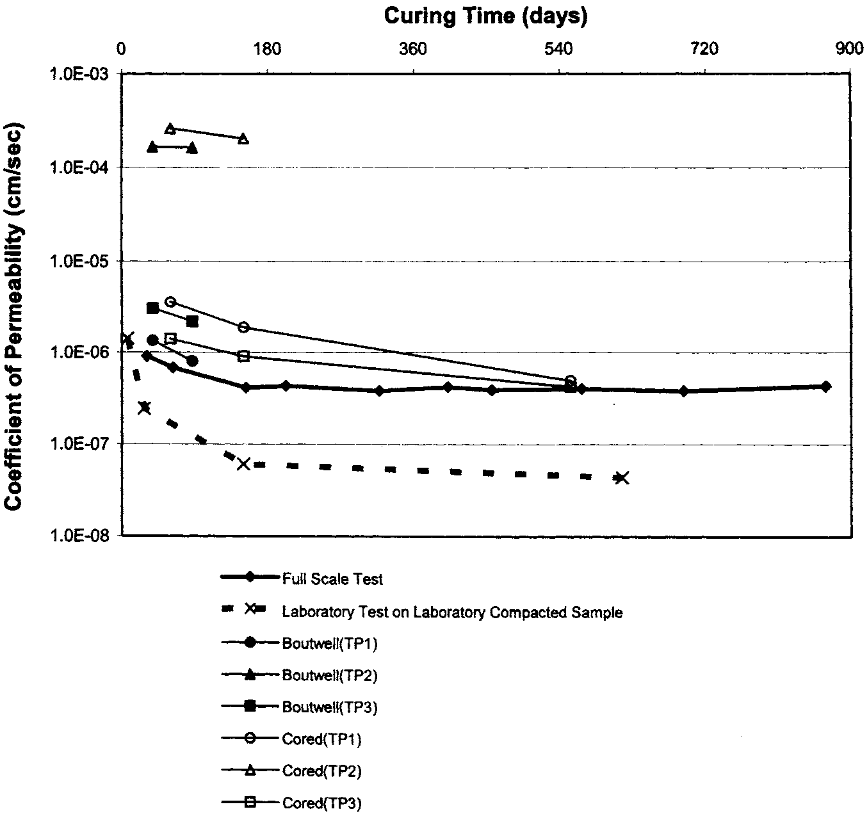
Notes:

*Effective area of FGD liner = Bottom area of pond facility. Swine manure was added to the facility 365 days after it was constructed.

significant amount of water may flow through the sides of the pond). Hence, the full-scale permeability values presented in Table 6-7 should be taken to be an upper bound to the actual permeability of the field-compacted FGD material liner. The addition of swine manure to the facility (at 365 days of curing and thereafter) did not affect the actual permeability of the liner.

The specific seepage rate (seepage volume below liner per unit liner area per unit time) for the full-scale facility can be obtained by multiplying the actual average coefficient of permeability with the hydraulic gradient across the liner. If no secondary liner were present at the site, the specific seepage rate for the 9-ft-deep manure storage pond having an 18-in.-thick FGD material liner was calculated to be 0.0952 $\text{in.}^3/\text{in.}^2/\text{day}$. Although Ohio currently does not provide guidance on maximum design seepage rate from lagoons, many states do specify maximum seepage rate values. For example, Kansas state regulations (Ham et al. 1998, 1999, 2000) allow for a maximum seepage rate of 0.25 $\text{in.}^3/\text{in.}^2/\text{day}$. Specific seepage rates less than 0.1 $\text{in.}^3/\text{in.}^2/\text{day}$ are considered to be very low seepage values. The value for the full-scale FGD-material–lined pond facility is 0.0952 $\text{in.}^3/\text{in.}^2/\text{day}$. Hence, it can be concluded that if there was no secondary clay liner constructed for the facility (as would be the norm for typical FGD-lined pond and manure storage facilities), the seepage rate loss from such FGD facilities would be very low.

The actual field permeability data obtained from the full-scale pond tests was compared with (1) laboratory tests conducted on several laboratory-compacted samples collected during pond construction, (2) laboratory tests conducted on field-compacted samples cored from test pads installed at the site, and (3) field permeability tests (Boutwell) conducted on test pads TP1, TP2, and TP3. TP4 was not considered for comparison because this test pad was constructed using FGD material that was lower in lime content and had a lower ratio of fly ash to filter cake compared to the material used in the construction of the full-scale facility. Figure 6-16 shows the time history comparison of the full-scale permeability test values with averaged permeability coefficients obtained from (1) laboratory tests on laboratory-compacted samples, (2) field tests (Boutwell) conducted on test pads, and (3) laboratory tests conducted on samples cored from test pads. All the test procedures showed decreasing permeability coeffi-



1 cm/sec = 0.0328 ft/sec

FIGURE 6-16. Comparison of Permeability Test Methods.

cient with increasing curing time. It was observed that the laboratory-compacted samples had permeability coefficients that were an order of magnitude lower than the full-scale testing values. Permeability values obtained from Boutwell tests and cored samples tested in the laboratory were in close agreement with each other but were one to three orders of magnitude higher than the full-scale tests. The test pad sample permeability values (Boutwell tests and cored sample testing) indicated a large scatter in the data. The permeability coefficients varied from 10^{-4} cm/s (3×10^{-6} ft/s) to 10^{-7} cm/s (3×10^{-9} ft/s), with average permeability values in the range of 10^{-5} cm/s (3×10^{-7} ft/s). This may be due to the unsuitable compaction achieved for the test pads. Furthermore, the Boutwell test procedure relates infiltration rate with permeability coefficient by assuming certain direction and boundary conditions of flow, which are nearly impossible to control in the field. However, it is important to note that the actual field-measured permeability values of the full-scale FGD material liner are an order of magnitude higher than laboratory-measured values.

6.3.4.2 Water quality. Water quality monitoring of the site was conducted by collecting water samples from the pond, the sump (leachate), and an existing farm vicinity well. The first baseline water samples were collected on September 12, 1997, before any water was added to the facility. Only well and sump samples were collected. After the pond had been filled with water on September 23, 1997, water samples were collected from the pond, sump, and well on a regular basis. The water sampling program was conducted while the pond held water (until September 1998) and beyond, at which time swine manure was added to the facility on a regular basis. It needs to be noted that the addition and removal of swine manure from the FGD-lined facility continued throughout the duration of the project. Swine manure was added to the facility in summer and winter and partially removed in fall for spreading on farm fields as per the agronomic needs of crops.

All samples were tested for several constituents and properties, including pH, electrical conductivity, alkalinity, acidity, total dissolved solids, aluminum, arsenic, boron, barium, calcium, cadmium, chromium, copper, iron, potassium, magnesium, manganese, sodium, nickel, phosphorus, lead, sulfur, selenium, silica, silver, vanadium, zinc, chloride, phosphate, sulfate, nitrate, ammonia, and nitrogen.

The time history for various water quality measurement parameters is shown in Figure 6-17 through Figure 6-51. The concentration level of various analytes in the sump (leachate) suddenly increased following the filling of the facility with water. Thereafter, the concentration levels reduced rapidly. The leachate from the FGD-material–lined facility needs to meet

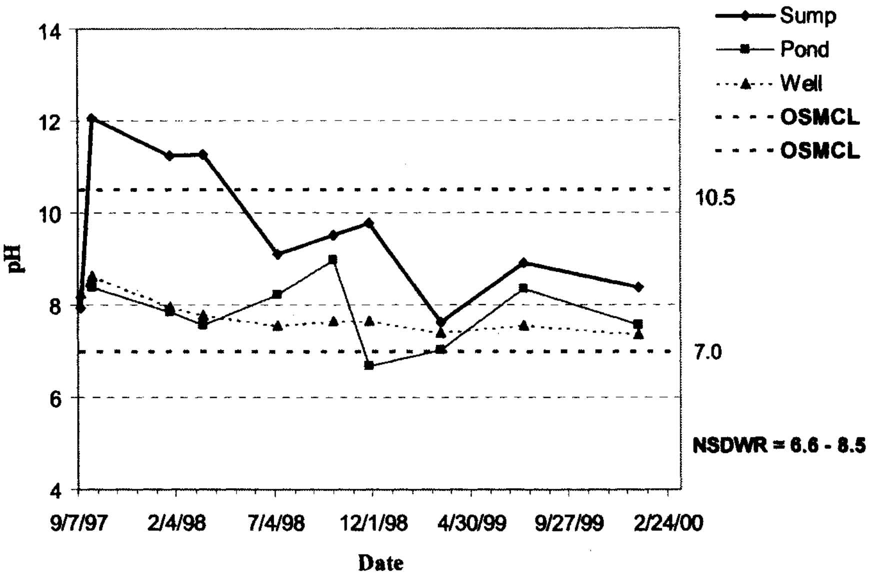


FIGURE 6-17. pH of Sump, Pond, and Well Samples.

the Ohio EPA's non-toxic criteria (Ohio EPA 1994). All the non-toxic parameters (arsenic, barium, cadmium, chromium, lead, and selenium) measured for the FGD material facility, while holding water and swine manure, were much lower than the Ohio EPA's non-toxic criteria. For

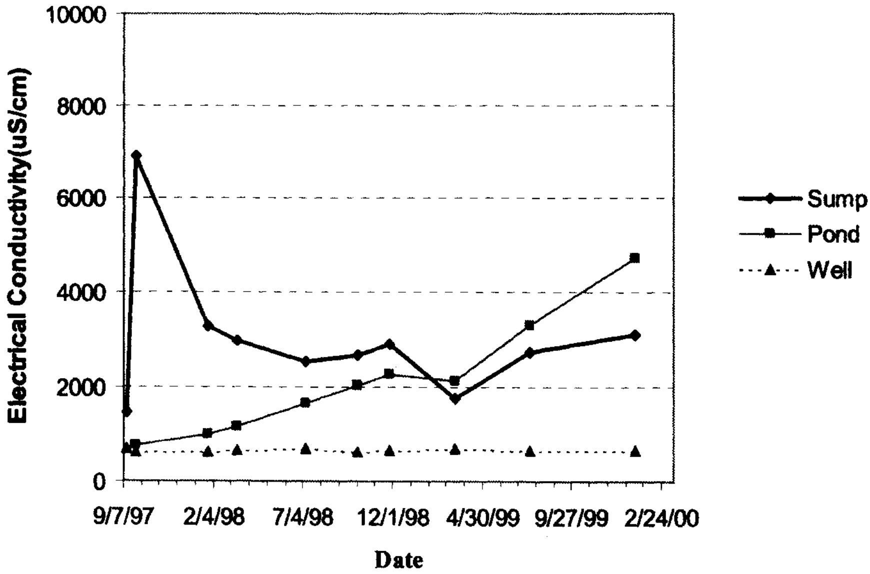


FIGURE 6-18. Electrical Conductivity of Sump, Pond, and Well Samples.

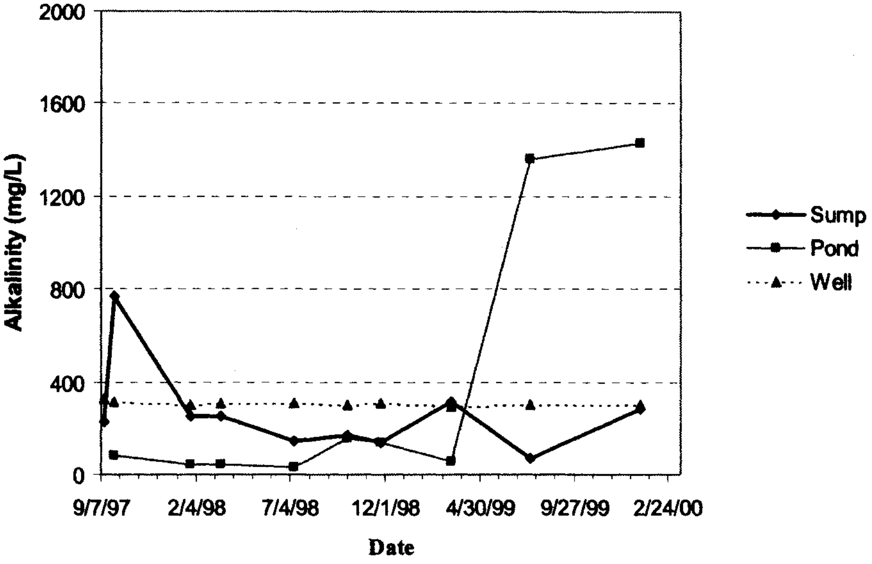


FIGURE 6-19. Alkalinity of Sump, Pond, and Well Samples.

many elements measured (barium, cadmium, chromium, and nitrate), the leachate concentration levels were generally lower than the National Primary Drinking Water Regulations (NPDWR).

Arsenic concentration levels for the pond and sump samples were very low for the first year (Figure 6-23). Upon addition of swine manure to the

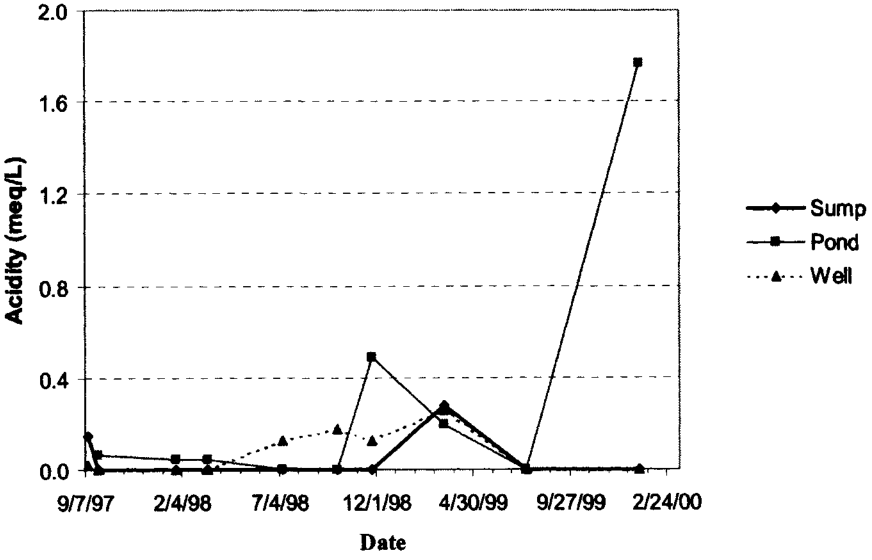


FIGURE 6-20. Acidity of Sump, Pond, and Well Samples.

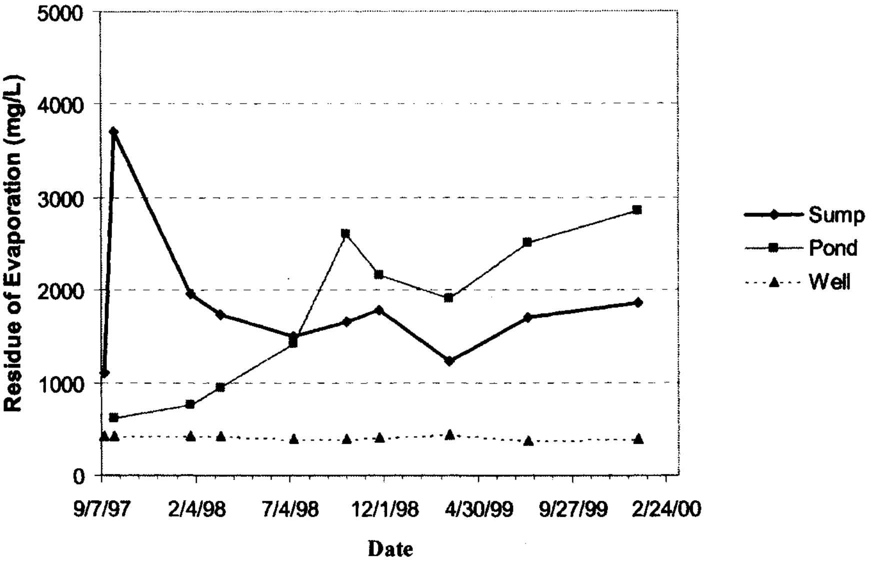


FIGURE 6-21. Residue of Evaporation of Sump, Pond, and Well Samples.

facility (in early September 1998), the concentration level of arsenic in the pond has increased. However, the sump samples indicate little increase in arsenic concentrations (which are slightly higher than the NPDWR level of 0.05 μg/ml). Arsenic levels in the sump and pond have always been

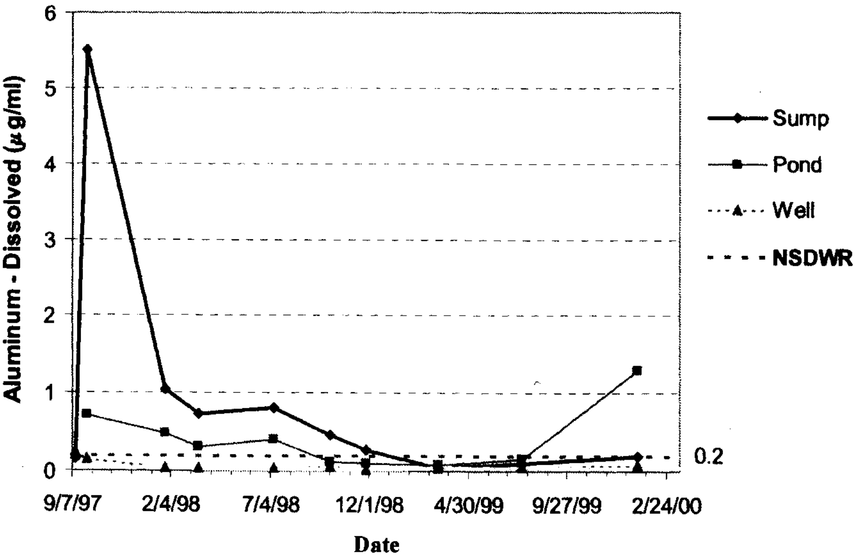


FIGURE 6-22. Concentration of Dissolved Aluminum for Sump, Pond, and Well Samples.

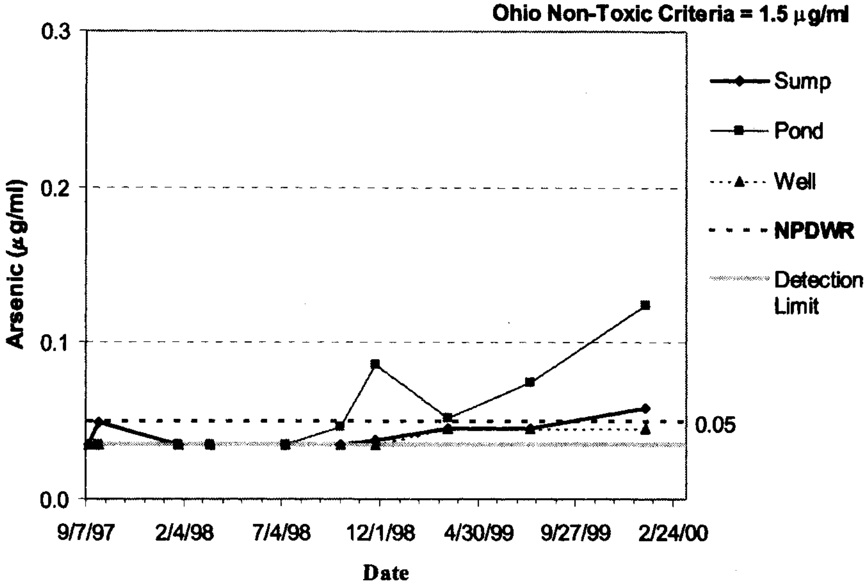


FIGURE 6-23. Concentration of Arsenic for Sump, Pond, and Well Samples.

lower than Ohio EPA's non-toxic regulatory value of 1.5 µg/ml. Barium concentrations were significantly lower than the primary drinking water standards (Figure 6-25). Cadmium concentrations (Figure 6-27) have generally been at the detection limit and much lower than the NPDWR for the

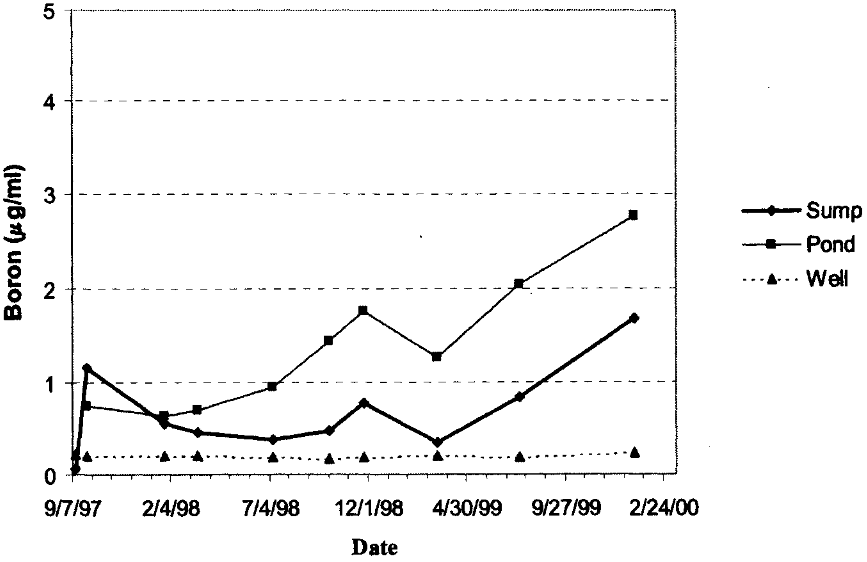


FIGURE 6-24. Concentration of Boron for Sump, Pond, and Well Samples.

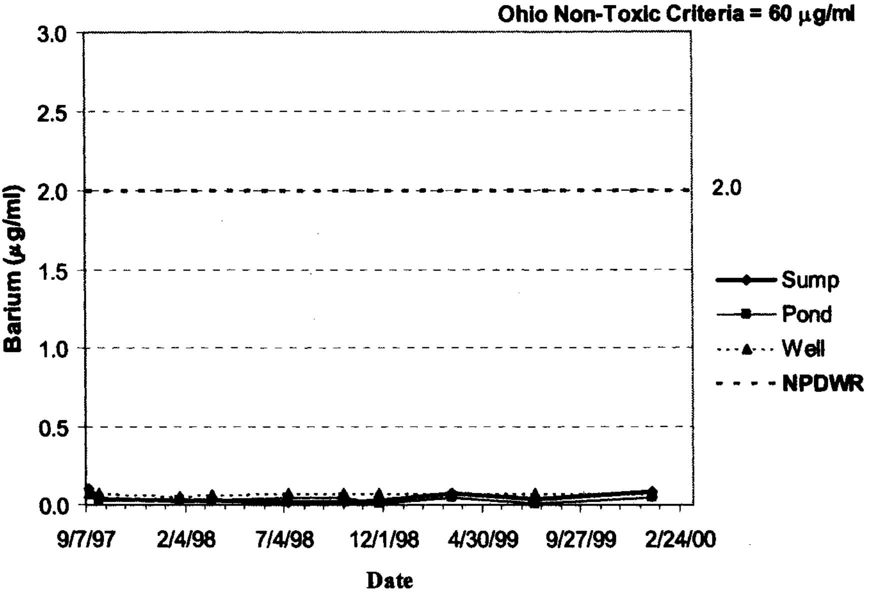


FIGURE 6-25. Concentration of Barium for Sump, Pond, and Well Samples.

monitoring duration. The sudden increase in all cadmium concentration values for October 2, 1998, seems to be erroneous. Chromium concentration in sump and pond samples has been lower than NPDWR except when the facility was just filled with water (Figure 6-28). Copper concen-

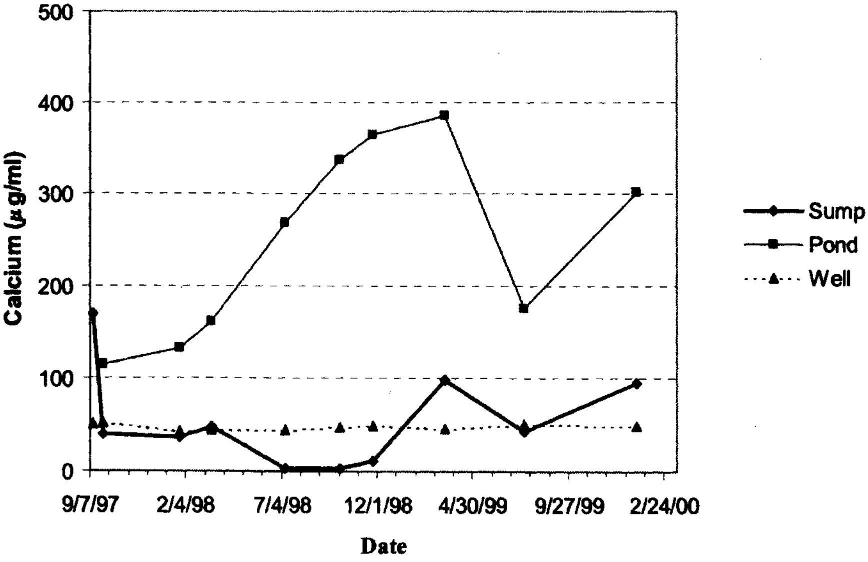


FIGURE 6-26. Concentration of Calcium for Sump, Pond, and Well Samples.

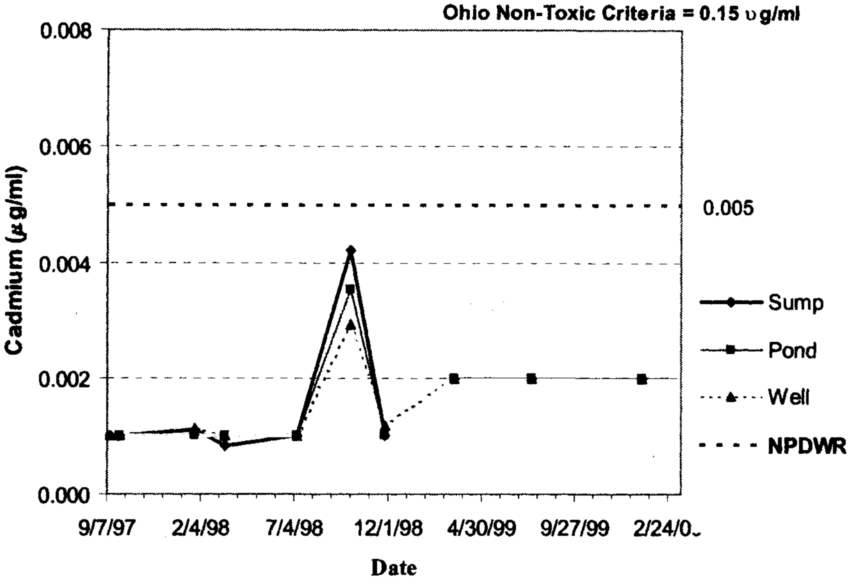


FIGURE 6-27. Concentration of Cadmium for Sump, Pond, and Well Samples.

trations have been significantly lower than the USEPA Action Level of 1.3 μg/ml and NSDWR limit of 1.0 μg/ml (Figure 6-29). Lead concentration levels for sump and pond have been at the detection limit (Figure 6-39). The lead level in well samples is elevated due to the use of lead pipes to

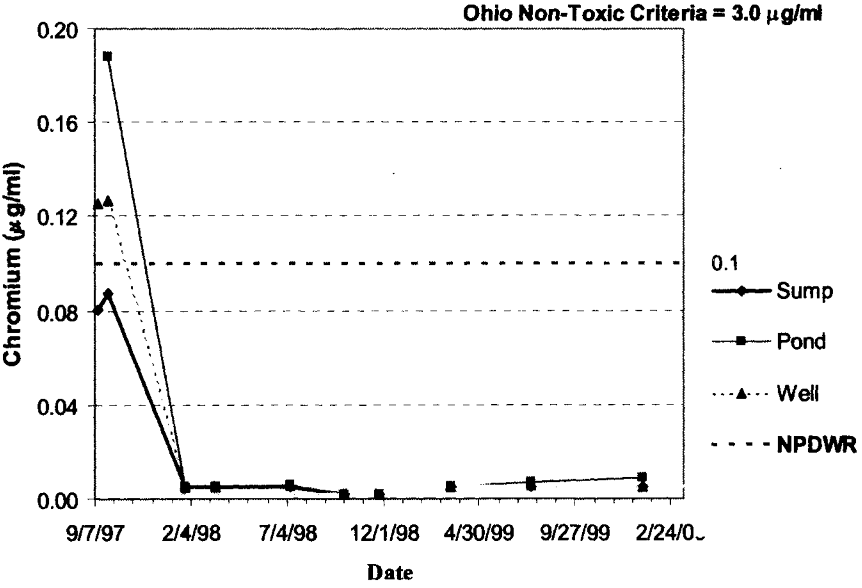


FIGURE 6-28. Concentration of Chromium for Sump, Pond, and Well Samples.

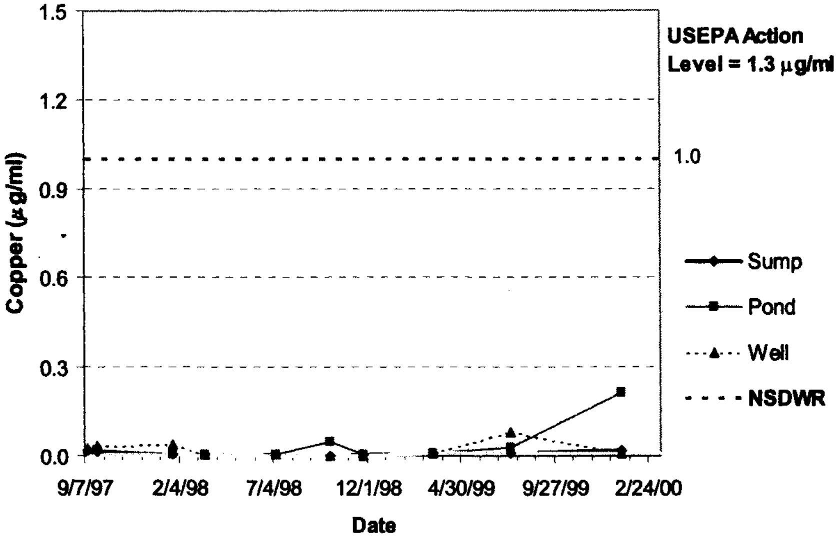


FIGURE 6-29. *Concentration of Copper for Sump, Pond, and Well Samples.*

convey water from the bottom of the well to the sampling point. Selenium concentration levels have generally been slightly above the NPDWR value of 0.05 μg/ml (Figure 6-41). A sudden increase in all selenium measurements for March 16, 1999, seems erroneous. Nitrate, a highly

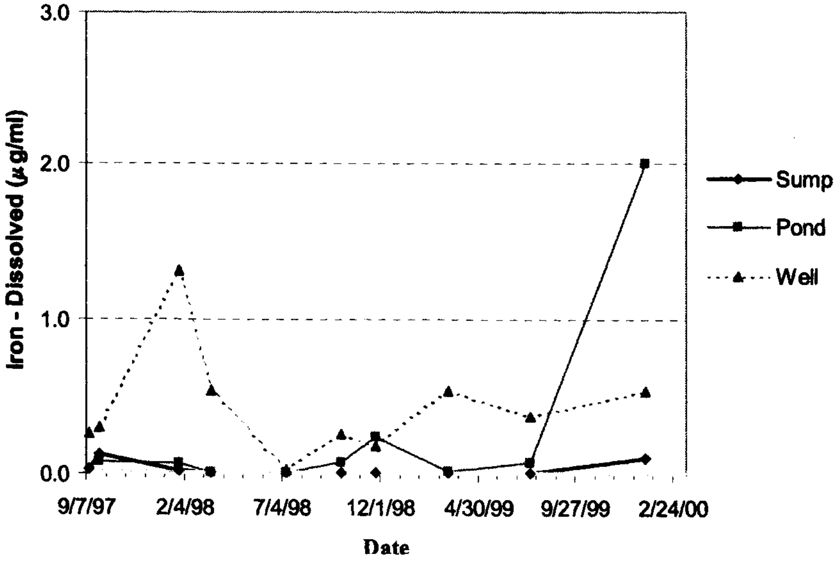


FIGURE 6-30. *Concentration of Dissolved Iron for Sump, Pond, and Well Samples.*

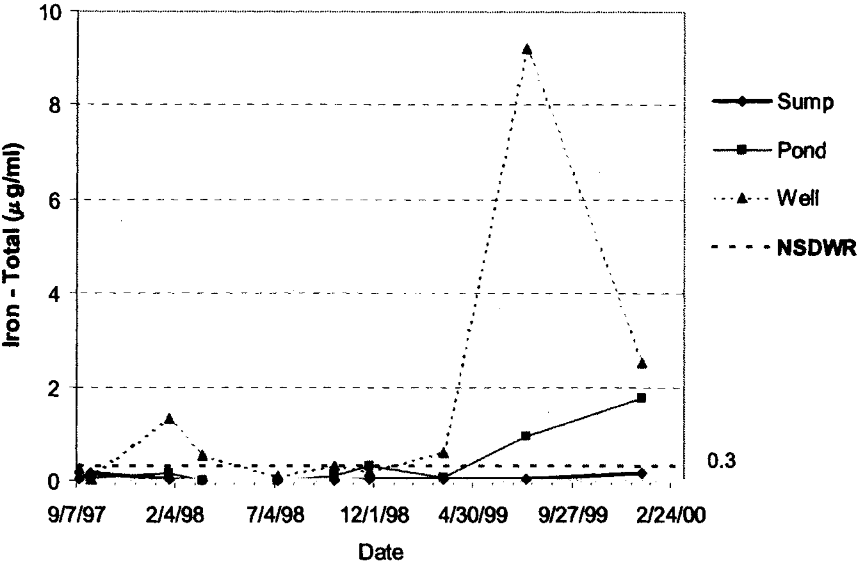


FIGURE 6-31. Concentration of Total Iron for Sump, Pond, and Well Samples.

soluble form of nitrogen, had low concentration for pond and sump samples, except when the facility was filled with water (Figure 6-49). In general, nitrate levels in the leachate are extremely low compared to the NPDWR value of 10 μg/ml.

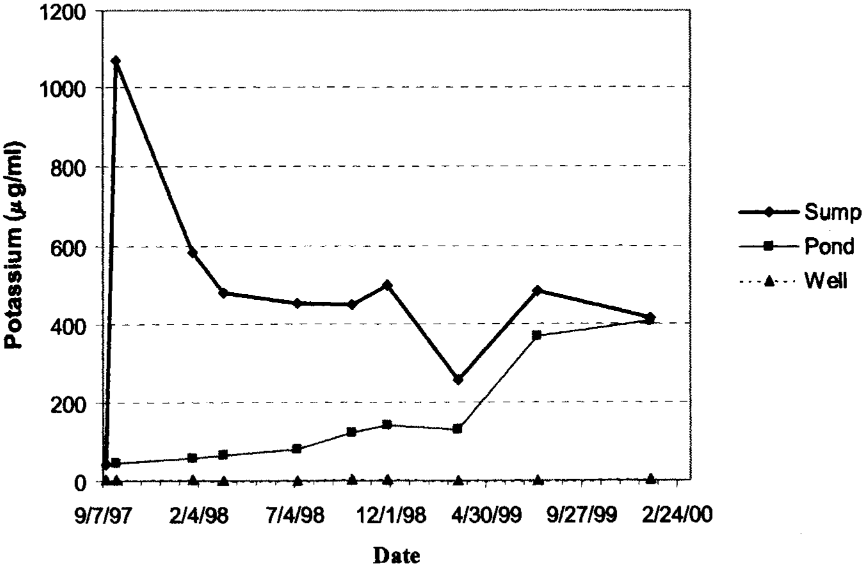


FIGURE 6-32. Concentration of Potassium for Sump, Pond, and Well Samples.

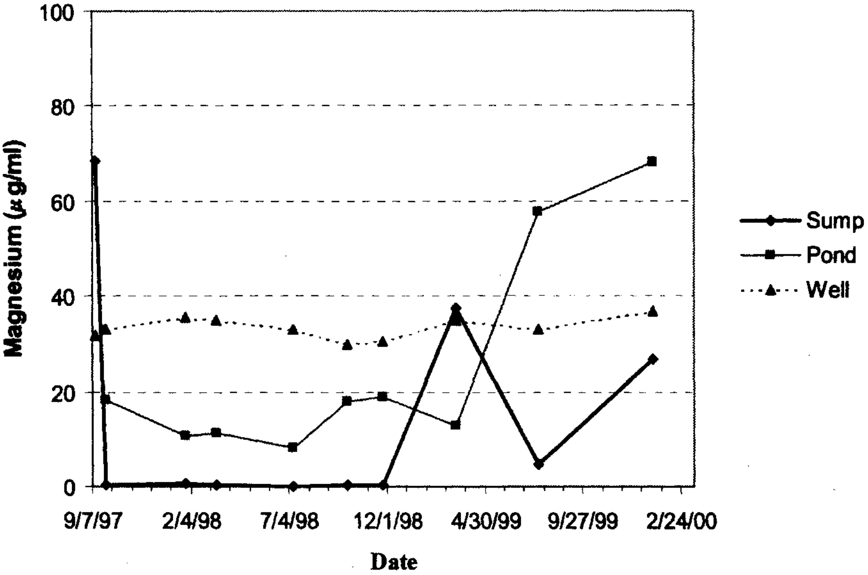


FIGURE 6-33. Concentration of Magnesium for Sump, Pond, and Well Samples.

It can be observed from Figure 6-17 that the pH of the well sample has been decreasing slightly according to seasonal groundwater variations. The pH of the pond sample was within the Ohio Secondary Maximum Contaminant Level (OSMCL). The pH of the sump water rose sharply to

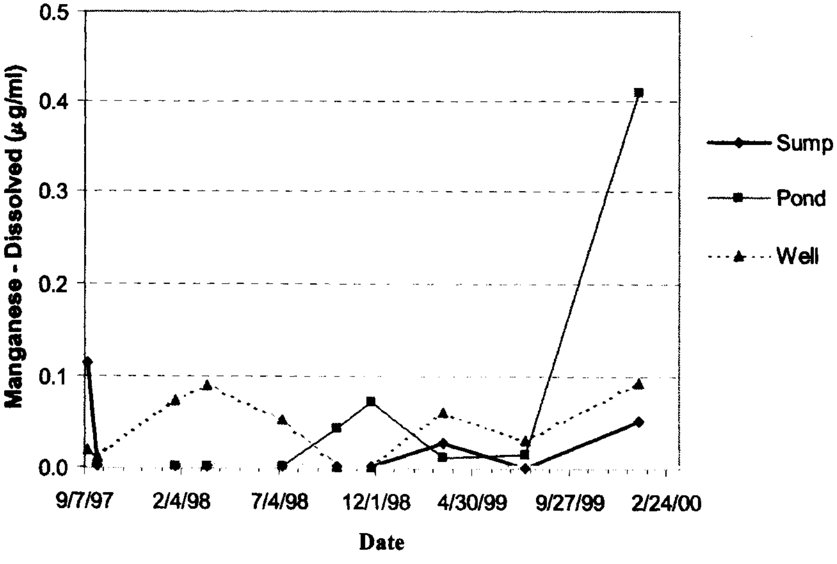


FIGURE 6-34. Concentration of Dissolved Manganese for Sump, Pond, and Well Samples.

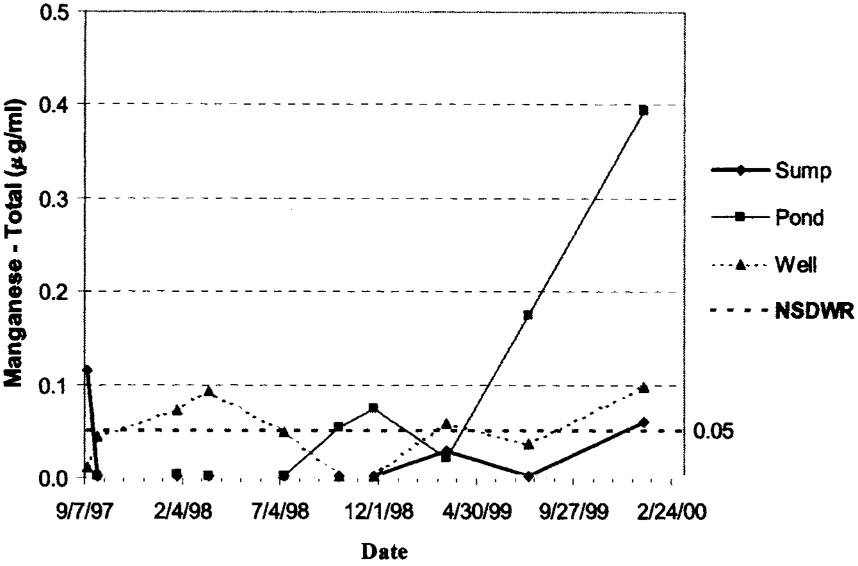


FIGURE 6-35. Concentration of Total Manganese for Sump, Pond, and Well Samples.

12 on filling the facility with water and has been dropping since then. The last pH-level reading for the sump was 8.4, which is within the OSMCL range of 7.0 to 10.5 and slightly lower than the National Secondary Drinking Water Regulations (NSDWR) upper limit of 8.5. The dissolved

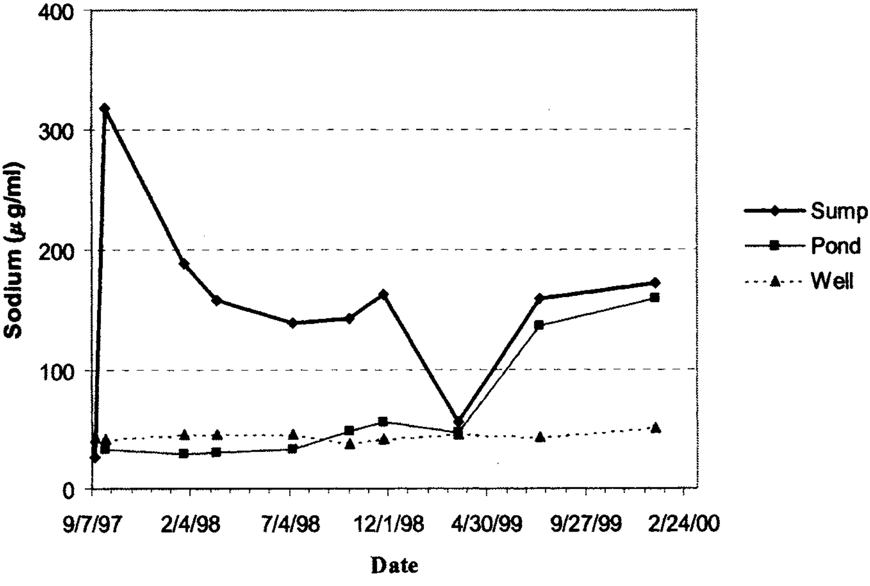


FIGURE 6-36. Concentration of Sodium for Sump, Pond, and Well Samples.

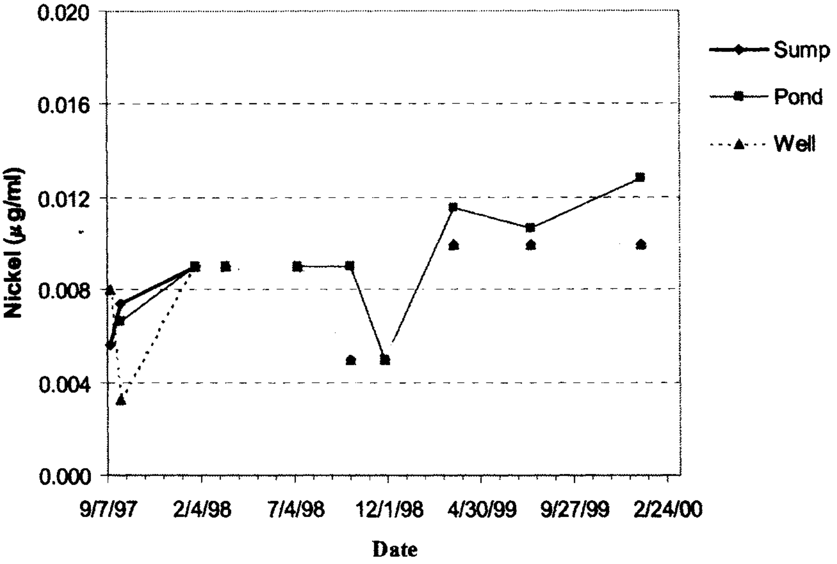


FIGURE 6-37. Concentration of Nickel for Sump, Pond, and Well Samples.

aluminum concentrations in the sump samples increased significantly during the filling of the pond (Figure 6-22). However, soon after filling the facility, the sump aluminum concentrations dropped significantly and have generally been around the NSDWR limit. Total iron levels for the

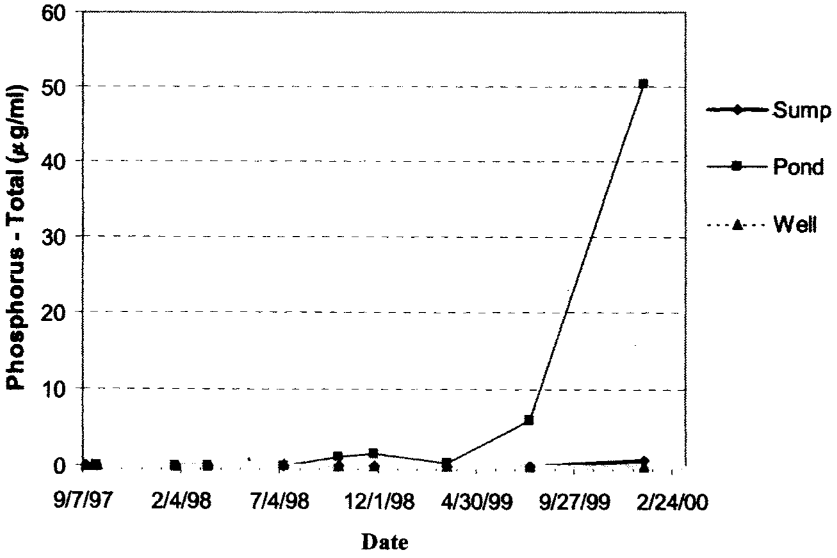


FIGURE 6-38. Concentration of Total Phosphorus for Sump, Pond, and Well Samples.

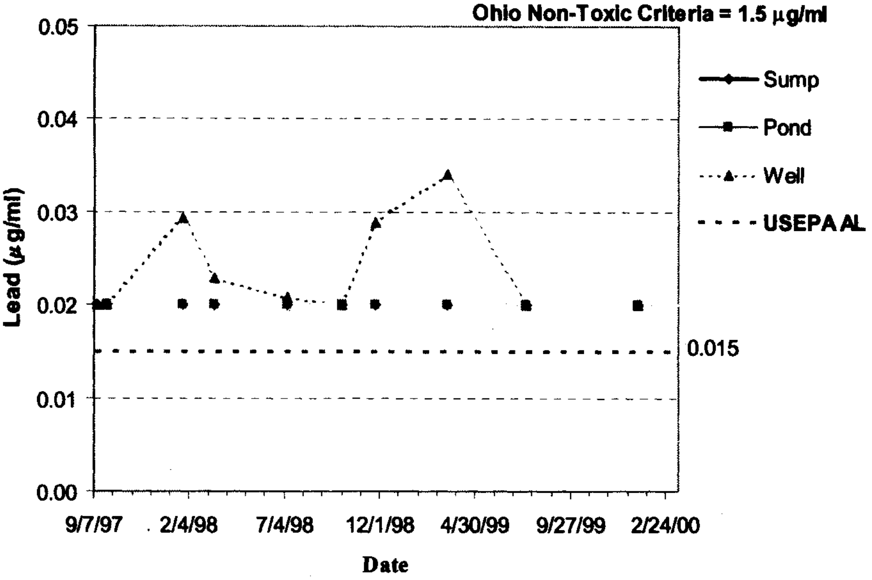


FIGURE 6-39. Concentration of Lead for Sump, Pond, and Well Samples.

pond and sump samples are quite low (Figure 6-31). Total manganese level for the sump is lower than or comparable to the NSDWR limit value (Figure 6-35). However, pond samples indicate manganese concentrations steadily increased due to the addition of swine manure to the pond.

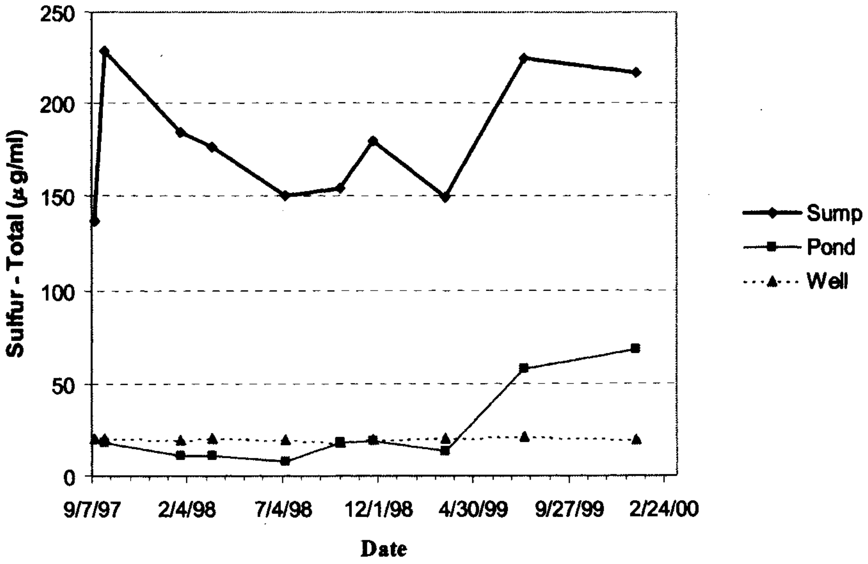


FIGURE 6-40. Concentration of Total Sulfur for Sump, Pond, and Well Samples.

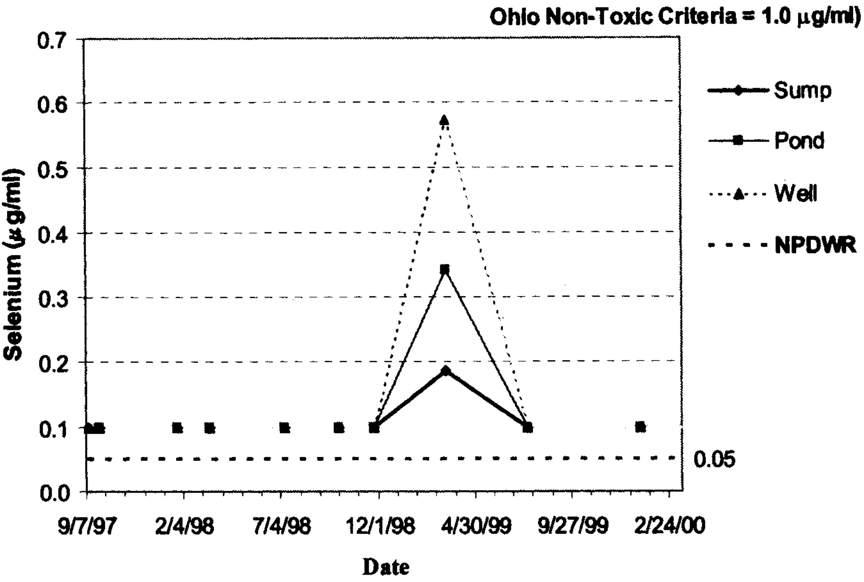


FIGURE 6-41. Concentration of Selenium for Sump, Pond, and Well Samples.

Silver concentrations reduced significantly after filling of the facility with water and for the last year or so have been at the detection limit (Figure 6-43). The concentration level of zinc in the sump and pond was much lower than the NSDWR limit for the complete duration of project moni-

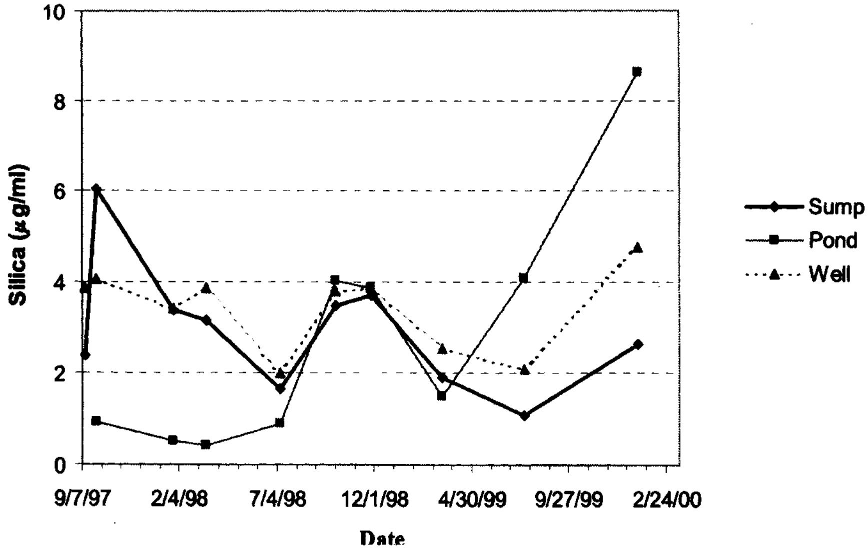


FIGURE 6-42. Concentration of Silica for Sump, Pond, and Well Samples.

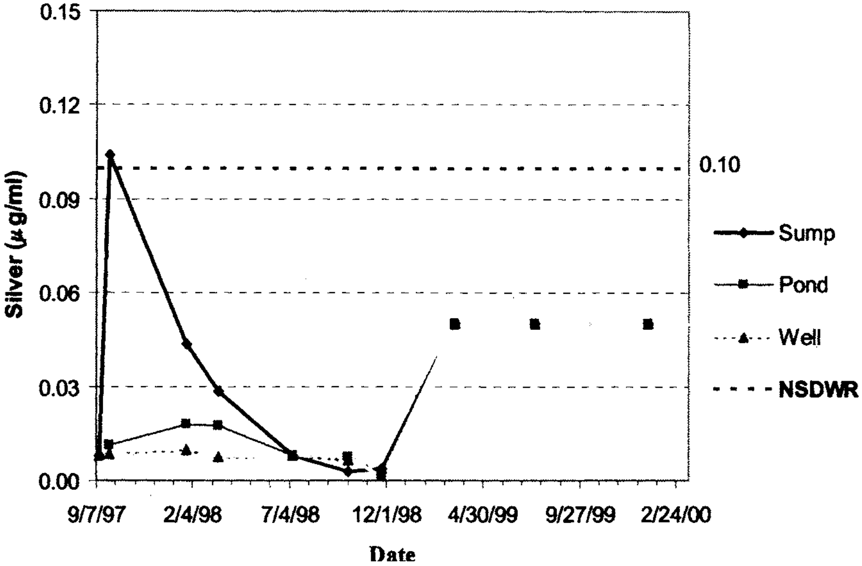


FIGURE 6-43. Concentration of Silver for Sump, Pond, and Well Samples.

toring (Figure 6-45). On filling the pond, the chloride concentration in the sump increased to about four times the NSDWR limit, but it has decreased since then to a level lower or slightly higher than the NSDWR regulation limit (Figure 6-46). Sulfate levels in the sump have generally been within the NSDWR limit (Figure 6-48). It should be noted that the

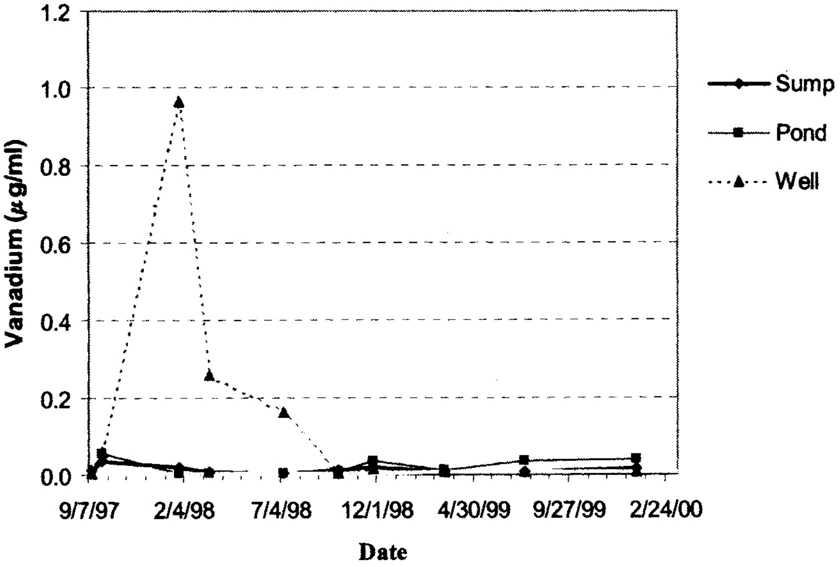


FIGURE 6-44. Concentration of Vanadium for Sump, Pond, and Well Samples.

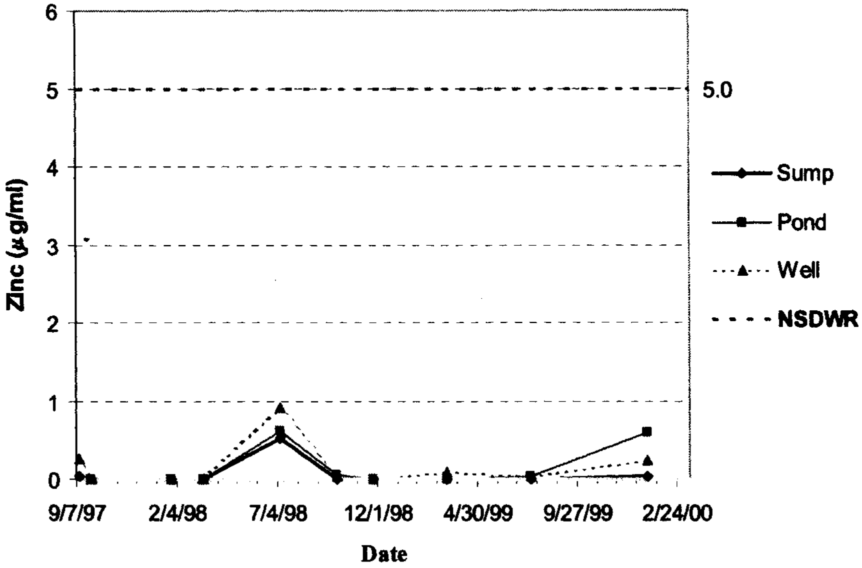


FIGURE 6-45. Concentration of Zinc for Sump, Pond, and Well Samples.

leachate from the full-scale FGD-material–lined facility meets most of the NSDWR limits. NSWDR limits are nonenforceable, non–health-related guidelines regulating potential contaminants that may cause cosmetic or aesthetic effects in drinking water. From a regulatory perspective, the leachate from the FGD material is not required to meet any of the NSDWR

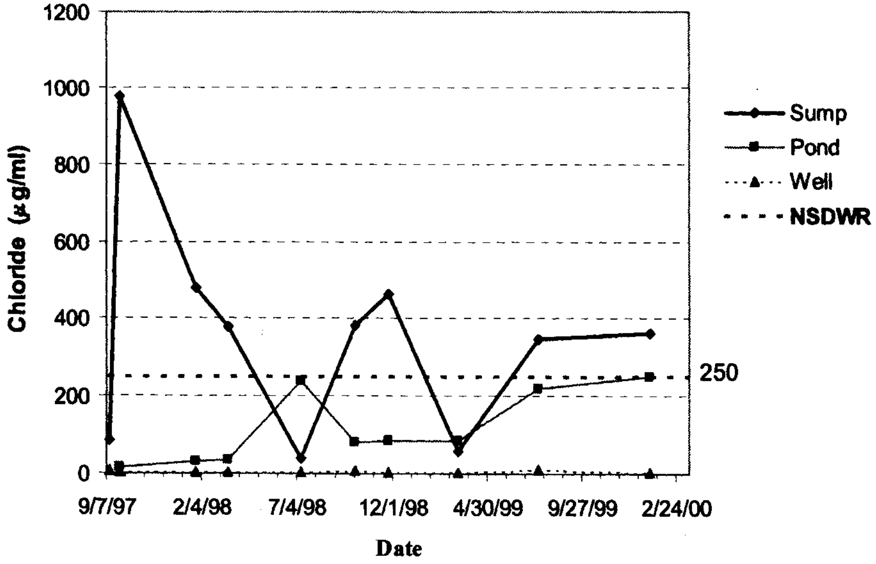


FIGURE 6-46. Concentration of Chlorides for Sump, Pond, and Well Samples.

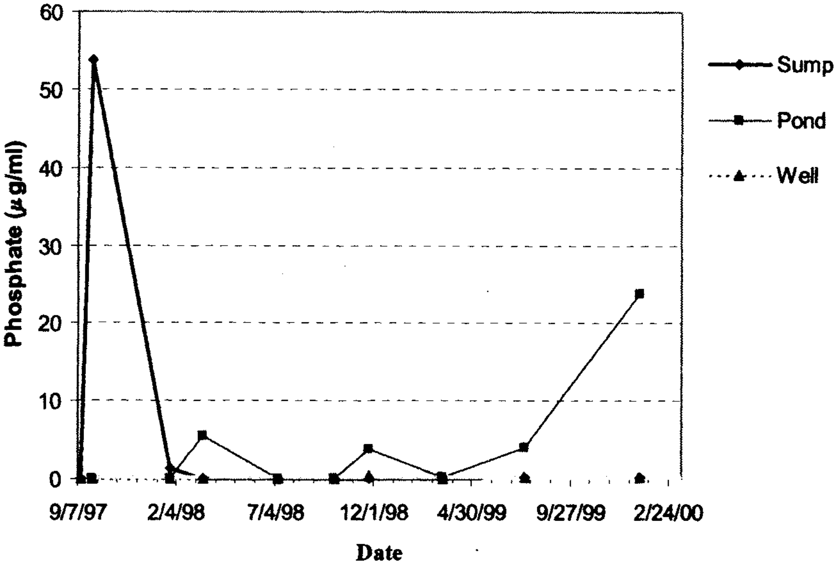


FIGURE 6-47. *Concentration of Phosphates (PO_4) for Sump, Pond, and Well Samples.*

limit values. Despite this, the leachate concentration levels measured from the FGD-material–lined facility are generally lower or comparable to the NSDWR limits for most potential contaminants of concern.

Boron, elevated levels of which can be phytotoxic to plant growth, generally had lower concentration levels in the sump than in the pond

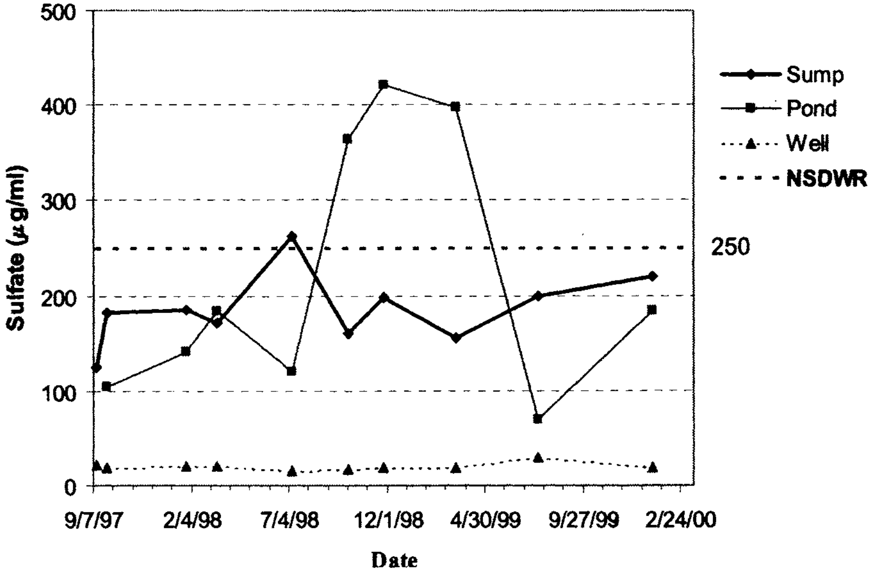


FIGURE 6-48. *Concentration of Sulfates (SO_4) for Sump, Pond, and Well Samples.*

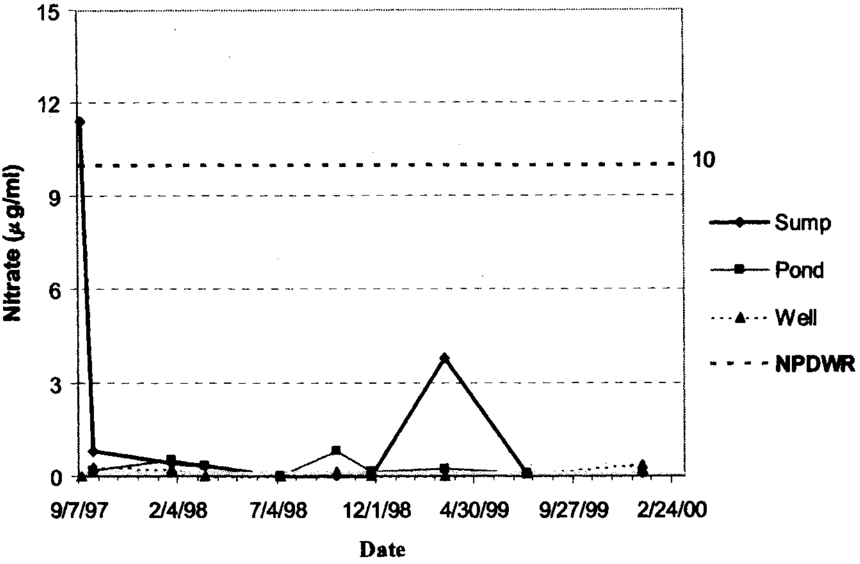


FIGURE 6-49. Concentration of Nitrates for Sump, Pond, and Well Samples.

(Figure 6-24). This may be due to absorption of some boron by the FGD material liner. Calcium level in the pond has continued to increase rapidly, while the sump indicates much lower concentration levels that have not increased significantly (Figure 6-26). Phosphorus concentration in the sump increased on addition of swine manure to the facility (Figure 6-38),

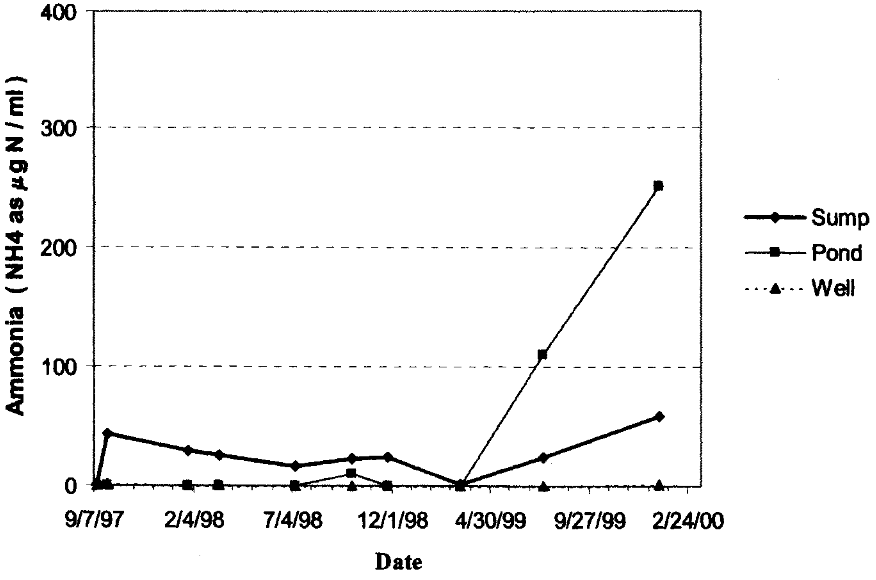


FIGURE 6-50. Concentration of Ammonia for Sump, Pond, and Well Samples.

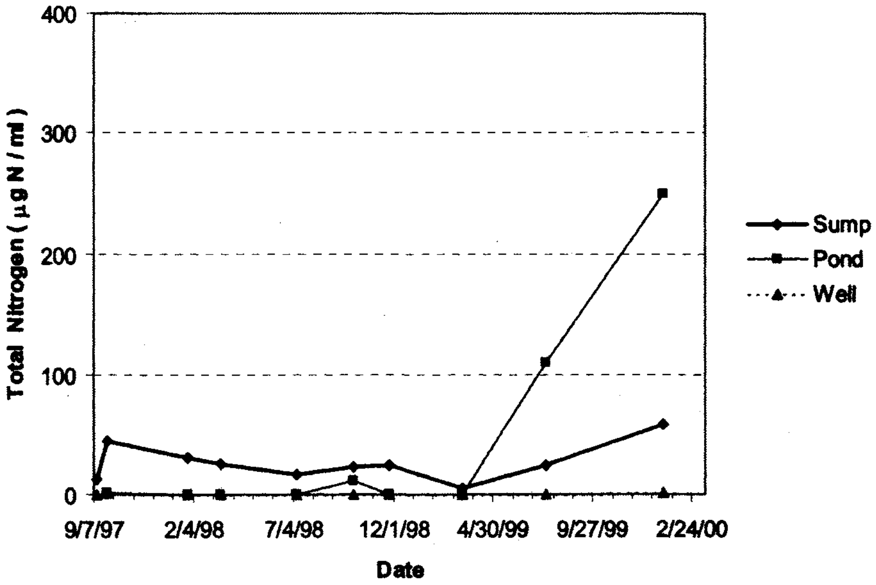


FIGURE 6-51. Concentration of Total Nitrogen for Sump, Pond, and Well Samples.

but the sump phosphorus levels have remained close to the detection limit. Phosphate levels in the pond increased on addition of swine manure, while the concentration levels in the sump remained relatively unchanged (Figure 6-47) at the detection limit. Ammonia levels increased in the pond with addition of swine manure. This resulted in an increase in the concentration of ammonia in the leachate (Figure 6-50). Ammonia, which can be converted to soluble nitrate in the presence of oxygen (nonsaturated conditions), may be a concern during the cleaning and removal of manure storage facilities (irrespective of whether the facility is constructed with clay or FGD material). The concentrations of highly soluble constituents, such as nitrates and phosphates, were observed to be relatively low in the sump.

Longer term monitoring of the facility with regular addition of swine manure is in progress, and it is expected that the facility will be monitored for an additional period of about 3.5 yr. This will allow the research team to collect permeability and water quality data for the FGD- material–lined facility for a total period of approximately 5 yr.

6.4 ADVANTAGES AND LIMITATIONS OF FGD LINERS

The use of compacted FGD liners in place of conventional clay or other synthetic liners for constructing pond and manure storage facilities has several advantages. The lime-enriched FGD material has low permeability coefficients (10^{-7} cm/s range) and has unconfined compressive strengths greater than clay. The permeability continues to decrease with time as the pozzolanic reaction progresses with time. The moisture density curve for most FGD materials is relatively flat, and hence the moisture content at compaction can be allowed to vary over a much wider range than clay to obtain maximum dry density. FGD materials may potentially serve as filters for manure passing through the liner. Compacted FGD material liners can be much cheaper than those constructed with other materials. If the FGD by-product could be used instead of clay or geomembrane, the savings could be as much as \$2 to \$3/ft^2. As an example, the FGD-material–lined demonstration facility was constructed in 1997 at a cost of approximately \$50,000. A similar concrete-lined facility, constructed by The Ohio State University 2 yr earlier, cost about \$200,000.

The use of FGD materials in the construction of liners does pose some potential limitations. In order to obtain low permeability, the material must be enriched with additional lime and fly ash (increased cost) and allowed to cure for about 60 days before freezing temperatures are expected. This will limit the construction season for FGD material liners to be installed in regions subjected to freezing temperatures (e.g., the engineered use of FGD material in Ohio is generally recommended between May and August). Additionally, the transport of the material from the producing facility to the site is expected to be economical only up to a distance of about 50 to 75 mi. Longer term studies on the technology are currently in progress.

6.5 CONCLUSIONS

Lime-enriched FGD material can be compacted in the laboratory to achieve permeability values lower than those generally recommended for lining manure containment facilities. A full-scale FGD-material–lined facility was constructed at The Ohio State University to study the permeability and leachate characteristics of a field-compacted FGD material liner. Water was held in the facility for the first year, after which swine manure was added to the facility.

Monitoring of the facility shows that the field-compacted FGD material liner has a low permeability (10^{-7} cm/s range), and the leachate that does flow through the low-permeability liner is comparable to drinking water standards. Leachate from the FGD material meets Ohio's non-toxic criteria for coal combustion by-products, and for most potential contaminants the national primary and secondary drinking water standards also are met. The low-permeability, non-toxic FGD material investigated in this study poses very minimal risks, if any, for groundwater contamination. The facility will continue to be monitored for a period of another 3.5 yr.

A project technical report on this subject (Wolfe et al. 2000) can be accessed at http://ccpohio.eng.ohio-state.edu/ccpohio/Liners%20report/Lower_permeability_liners.PDF. More information on this research initiative may be obtained from the Internet web address http://ccpohio.eng.ohio-state.edu/liner/. Additional information on the use of coal combustion by-products can be accessed at http://ccpohio.eng.ohio-state.edu.

CHAPTER 7

AIR QUALITY ISSUES ASSOCIATED WITH LIVESTOCK PRODUCTION

Ronaldo Maghirang and James P. Murphy

Livestock production is a major agricultural pursuit in the United States that involves more than 450,000 farms (GAO 1995). As a result of market forces and technological changes, the past few decades have brought a shift to fewer but larger animal feeding operations (AFOs). Also, greater confinement of animals has been adopted to reduce labor requirements. Larger operations with intensive confinement have resulted in the concentration of large quantities of waste and wastewater on limited areas. This, in turn, has created significant environmental concerns, including air pollutant emissions and surface and groundwater contamination. Animal feeding operations can emit ammonia (NH_3), nitrous oxide (N_2O), hydrogen sulfide (H_2S), carbon dioxide (CO_2), methane (CH_4), total reduced sulfur (TRS) compounds, volatile organic compounds (VOCs), hazardous air pollutants, and particulate matter (USEPA 2001).

Animal waste includes the fecal and urinary wastes of livestock and poultry; process water, such as from a milking parlor; and feed, bedding, litter, and soil with which they become intermixed (USEPA 1993). A typical animal waste management system includes six basic functions: production, collection, storage, treatment, transfer, and utilization (USDA 1992). A wide spectrum of animal waste management systems has been developed because of different climatic conditions, animal species, manure characteristics, and engineering technologies. Table 7-1 lists the components of animal waste management systems according to function.

Animal waste can be collected and stored as liquids, slurries, semisolids, or solids. Raw manure has a solids content of 8% to 25% depending on animal type; liquid, slurry, semisolid, and solid wastes have solids contents of <3%, 3%–10%, 10%–20%, and >20%, respectively (Roos and Moser 1997). Poultry, beef cattle, and sheep production systems use mainly solid systems. Dairy production systems use all waste forms. Most confined

TABLE 7-1. Components of Livestock Waste Management Systems

Function	Components
Production	Roof gutters and downspouts, diversions
Collection	Alley scrapers, flush alleys, manure packs, gutters
Storage	Ponds, tanks, dry stacks
Treatment	Lagoons, composters, solids separators, settling basins
Transfer	Pipelines, hauling equipment, gutters, pumps, use of ramps
Utilization	Irrigation systems, spreaders, commercial sale, refeeding, bedding, energy generator

Source: CAST 1996.

or partially confined swine operations handle manure as liquids or slurries. Safley et al. (1992) noted that more than 75% of the swine production systems in the United States use liquid manure handling systems because of their greater economic feasibility (Sievers and Ianotti 1982) and because of the extremely high biological oxygen demand (BOD) of the swine waste (Sievers and Ianotti 1982; Kobayashi and Kobayashi 1995).

Stored animal waste can impact air quality through the generation of various odorous gases. According to Miner and Suh (1997), surfaces of lagoons and manure storage tanks are among the most frequently identified sources of odors. These structures are relatively consistent in their odor generation and are regarded as having the most unpleasant odors.

This chapter reviews the current state of the art on air quality issues associated with animal waste containment. Outdoor waste storage and treatment systems are emphasized. Air quality control strategies also are summarized. Additionally, research and development needs are identified.

7.1 ANIMAL WASTE AND AIR QUALITY

Animal waste containment structures include tanks, lagoons, ponds, and sheds. Structures such as lagoons, retention ponds, and tanks are suitable for liquid and slurry; sheds are used for dry waste such as poultry litter. Figure 7-1 shows a storage lagoon for swine waste; Figure 7-2 shows an above-ground storage tank for dairy cow waste.

When the waste is collected and stored in tanks, ponds, or lagoons, it naturally undergoes decomposition from the metabolic action of microorganisms (Westerman and Zhang 1997). Available oxygen is consumed quickly by the high oxygen demand of the decomposing organic matter. As a result, the decomposition becomes anaerobic (i.e., without free oxygen),

FIGURE 7-1. Lagoon for Swine Waste.

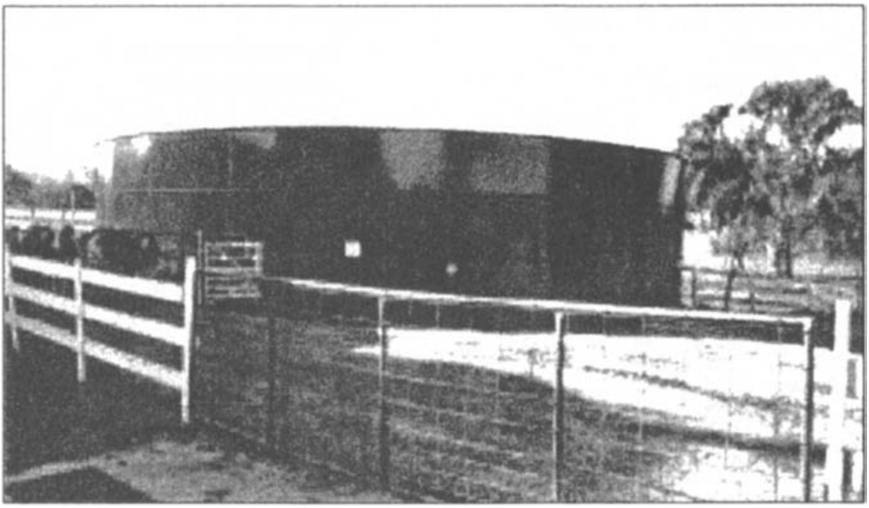


FIGURE 7-2. Above-Ground Storage Tank for Dairy Waste Source: GAO 1999).

and CH_4 and CO_2 are the primary gases produced. In an anaerobic lagoon, for example, about 60% to 70% of the gas produced is CH_4, and about 30% is CO_2 (USDA 1992). In addition to these two major gases, trace amounts of numerous other compounds have been identified in the waste or in the air exposed to degrading animal waste (O'Neill and Phillips 1992; Schiffman et al. 2001).

7.1.1 Properties and Effects of Manure Gases and Vapors

The gases and vapors of most interest and concern in animal waste management are CH_4, CO_2, NH_3, H_2S, and VOCs. Typical compounds resulting from anaerobic decomposition of animal manure are listed in Table 7-2 (ASAE 1998a). Table 7-3 summarizes the most significant characteristics of NH_3, CO_2, H_2S, and CH_4. Table 7-4 lists the properties of some of the major VOCs associated with swine waste odor. Important properties and effects of these gases and vapors are described below.

7.1.1.1 Carbon dioxide. Carbon dioxide is considered a greenhouse gas; it traps heat in the atmosphere, contributing to global warming. It can be an asphyxiant when it displaces normal air in a confined animal facility. Because it is heavier than air, it remains in a tank or other well-sealed

TABLE 7-2. Compounds Resulting from the Anaerobic Decomposition of Livestock and Poultry Manure

- Volatile Fatty Acids (VFAs):
  - Acetic
  - Propionic
  - Butyric
  - Isobutyric
  - Isovaleric
- Alcohols
- Aldehydes
- Esters
- Phenols and cresols
- Mercaptans:
  - Methylmercaptan
  - Ethylmercaptan
  - Propylmercaptan
- Sulfides:
  - Hydrogen sulfide
  - Dimethyl sulfide
  - Diethyl sulfide
- Ammonia and amines:
  - Ammonia
  - Methylamine
  - Ethylamine
  - Trimethylamine
  - Diethylamine
- Nitrogen heterocycles:
  - Indole
  - Skatol
- Odorless gases:
  - Carbon dioxide
  - Methane

Source: ASAE 1998a.

TABLE 7-3. Properties and Physiological Effects of the Most Important Gases Produced from Animal Wastes in an Anaerobic Environment

Gas	Density	Odor	Class	Effects
Ammonia	Lighter than air	Sharp, pungent	Irritant	Irritation of the eyes and throat at low concentrations; could be fatal at high concentrations with 30- to 40-min exposure.
Carbon dioxide	Heavier than air	Odorless	Asphyxiant	<20,000 ppm is safe level; increased breathing, drowsiness, and headaches as concentration increases; could be fatal at 300,000 ppm for 30 min.
Hydrogen sulfide	Heavier than air	Similar to rotten eggs	Poison	Headaches, dizziness at 200 ppm for 60 min. Nausea, excitement, insomnia at 500 ppm for 30 min; unconsciousness, death at 1,000 ppm.
Methane	Lighter than air	Odorless	Asphyxiant, flammable	Headaches at 500,000 ppm; can be asphyxiating.

Source: USDA 1992.

structure, gradually replacing the lighter gases (USDA 1992). Controlling emissions of CO_2 has received little research attention, primarily because it is an otherwise benign compound that also would be produced by many other waste treatment alternatives, including aerobic treatment.

7.1.1.2 Methane. Methane is produced from the decomposition of animal waste in an anaerobic environment. It is a highly flammable gas; as such, extreme care is required when attempting to generate and capture this gas for on-farm use (USDA 1992). With respect to animal waste, control efforts have focused on CH_4 because it is extremely effective at trapping heat in the atmosphere. It also is a precursor to the formation of tropospheric

TABLE 7-4. Physicochemical Properties of Selected Volatile Compounds

Chemical Name[a] (CAS Number)	Chemical Formula[a]	Molecular Weight[a] (g/gmol)	Density (g/cm³)	Vapor Pressure at 25 °C [b] (mm Hg)	Boiling Point[a] (°C)	Odor Detection Threshold[c] (mg/m³)	Odor Char.
Acetic acid (64-19-7)	$C_2H_4O_2$	60	1.05	15.58	118	0.025–10	Pungent
Propionic acid (79-09-4)	$C_3H_6O_2$	74	0.97	3.67	141	0.003–0.89	Pungent, rancid
Isobutyric acid (79-31-2)	$C_4H_8O_2$	88	0.95	1.80	155	0.005–0.33	Pungent
Butyric acid (107-92-6)	$C_4H_8O_2$	88	0.97	0.94	164	0.0004–42	Sweaty, rancid
Isovaleric acid (503-74-2)	$C_5H_{10}O_2$	102	0.94	0.43	177	0.0002–0.0069	Disagreeable, rancid-cheese
Valeric acid (109-52-4)	$C_5H_{10}O_2$	102	0.94	0.24	186	0.0008–0.12	Unpleasant
Hexanoic acid (142-62-1)	$C_6H_{12}O_2$	116	0.93	0.04	206	0.02–0.52	Char. "goat-like"
Heptanoic acid (111-14-8)	$C_7H_{14}O_2$	130	0.92	0.01	223	0.022–0.033	Disagreeable, rancid
Phenol (108-95-2)	C_6H_6O	94	1.07	0.52	182	0.022–4	—
p-cresol (106-44-5)	C_7H_8O	108	1.03	0.10	203	0.00005–0.024	—
Indole (120-72-9)	C_8H_7N	117	—	0.01	254	0.0006–0.0071	Intense fecal, nauseating
Skatole	C_9H_9N	131	—	—	265	0.00035–0.00078	Fecal, nauseating
Ammonia (7664-41-7)	NH_3	17	—	7474	−33	0.03–37.8	Sharp, pungent

[a] CAS, chemical formulas, molecular weights, and boiling points are from Mallard and Linstrom 1998.

[b] Vapor pressures are derived using the Antoine equation; the coefficients are from Yaws et al. 1999.

[c] Odor threshold values are from van Gemmert and Nettenbreijer 1977, as summarized by O'Neill and Phillips 1992.

ozone, a component of photochemical smog. Estimates in 1995 indicated that CH_4 emissions from animal waste management systems were about 10% of total United States CH_4 emissions, and about 31% of CH_4 emissions were from the agriculture sector (USEPA 1997a). Additionally, liquid-based manure management systems accounted for more than 80% of total CH_4 emissions from animal waste (USEPA 1997a).

7.1.1.3 Ammonia. Ammonia is produced from microbial and enzymatic decomposition of nitrogenous compounds in animal waste, such as protein and amino acids. It is primarily an irritant and has been known to create respiratory health problems in animals and workers in livestock confinement buildings. Irritation of the eyes and respiratory tract are common problems from prolonged exposure to this gas.

In the United States, the atmospheric deposition of NH_3 and other nitrogen compounds has received renewed attention as a major route of entry into watersheds, especially the lower river basins and coastal estuaries of the eastern United States (Doorn et al. 2002). Ammonia can have direct toxic effects on vegetation close to a strong source, such as slurry storage (Cumby et al. 1995). At lower concentrations, it can distort the nutrient balance of natural ecosystems and selectively enhance the growth of some nitrogen-responsive species (Cumby et al. 1995). It also may react with acidic gases, such as sulfur dioxide and nitrogen oxides, to form ammonium sulfate and ammonium nitrate, which contribute to fine particulate concentrations (USEPA 2001). These fine particulates can be transported and deposited over large areas, contributing to nutrient overenrichment in aquatic systems and acidification of the environment (Ap Simon and Kruse-Pass 1991; USEPA 2001). In forest ecosystems, deposition of ammonium salts leads to leaching of potassium, magnesium, and calcium ions, which affects the availability of these nutrients (Cumby et al. 1995).

7.1.1.4 Hydrogen sulfide. Hydrogen sulfide is a highly poisonous gas. Humans and farm animals have been killed by this gas after falling into or entering a manure tank or being in a building in which a manure tank was being agitated (USDA 1992). It has the distinct odor of rotten eggs. It deadens the olfactory nerves (the sense of smell); therefore, if the smell of rotten eggs seems to have disappeared, this does not guarantee that the area is free of the gas (USDA 1992). Although only small amounts of H_2S are produced in a manure tank compared to the other major gases, it is heavier than air and becomes more concentrated in the tank over time.

In general, H_2S occurs only in trace amounts at or near livestock facilities. Koelsch et al. (2002) conducted a field survey of TRS in the vicinity of beef cattle feedlots in Nebraska using an instrument that responded primarily to H_2S and also to other compounds, including alkyl sulfides, disulfides, mercaptans, and cyclic sulfur compounds. They observed that the TRS levels in the vicinity of beef cattle feedlots are not likely to exceed current regulatory thresholds used by states in the U.S. Midwest.

7.1.1.5 Odorants. In addition to the aforementioned gases, numerous other odorous compounds, herein referred to as odorants, are generated by the biological decomposition of animal waste (Tables 7-2 and 7-4). Manure is a complex mixture of undigested dietary residues, endogenous secretions, and bacterial cells and their metabolic end-products (Mackie et al. 1998). It is subject to anaerobic decomposition under a variety of temperature and moisture conditions, resulting in the generation of odorants. O'Neill and Phillips (1992) presented a list of 168 volatile compounds that have been identified in livestock wastes by previous researchers. Schiffman et al. (2001) identified a total of 331 different compounds from swine facilities in North Carolina; the compounds included many acids, alcohols, aldehydes, amides, amines, aromatics, esters, ethers, fixed gases, halogenated hydrocarbons, hydrocarbons, ketones, nitriles, other nitrogen-containing compounds, phenols, sulfur-containing compounds, steroids, and other compounds. According to Yu et al. (1991), indole, p-cresol, phenol, skatole, volatile fatty acids (VFAs) (e.g., acetic, propionic, isobutyric, butyric, isovaleric, valeric, caproic, and heptanoic), and NH_3 appeared to be the most important constituents of odor from swine waste. Zahn et al. (1997) indicated that, based on their research and available odor threshold data, the C2 through C9 VFAs represented the greatest threat to air quality because of high experimental transport efficiencies, high airborne concentrations, and low odor thresholds.

Mackie et al. (1998) reviewed the biological origins of the four principal classes of odorants, namely, branched- and straight-chain VFAs, NH_3 and volatile amines, indoles and phenols, and the volatile sulfur-containing compounds. They noted the following:

- The organic matter in feed, particularly structural carbohydrates, is fermented to VFAs, mainly acetic, propionic, and butyric acids and smaller amounts of valeric, hexanoic, and heptanoic acids.
- Indoles and phenols result from amino acid metabolism. Phenol, p-cresol, 4-ethyl phenol, and hydroxylated phenol-substituted fatty acids are the main products of tyrosine fermentation. Phenyl acetate and phenyl propionate are produced from phenylalanine. Indole and skatole are the principal products of tryptophan metabolism.
- Sulfur compounds are produced from sulfate reduction and metabolism of sulfur-containing amino acids (e.g., cysteine and methionine). Sulfates can be supplied by dietary means or by depolymerization and desulfation of endogenously produced, sulfated glycoproteins such as mucins.

Neighbors of AFOs reasonably expect to be able to enjoy their homes and the countryside without nuisance from offensive odors. As livestock production has evolved toward larger and more intensive units in recent years, the frequency and severity of odor complaints have risen in the United States and many other countries (Miner 1995). These complaints occur when acceptable odor intensities and frequencies are exceeded. Data from the Institution of Environmental Health Officers indicated that about 4,000 complaints about odors from farms are reported each year in England and Wales (Pain 1994). Pig farms tended to generate the most complaints, followed by poultry and cattle (Figure 7-3). Also, major sources of complaints include land application of manure, followed by buildings and then waste storage systems (Figure 7-4). A recent survey of Minnesota swine producers (Schmidt et al. 1996) showed that about 29% of the swine producers surveyed have experienced odor problems from

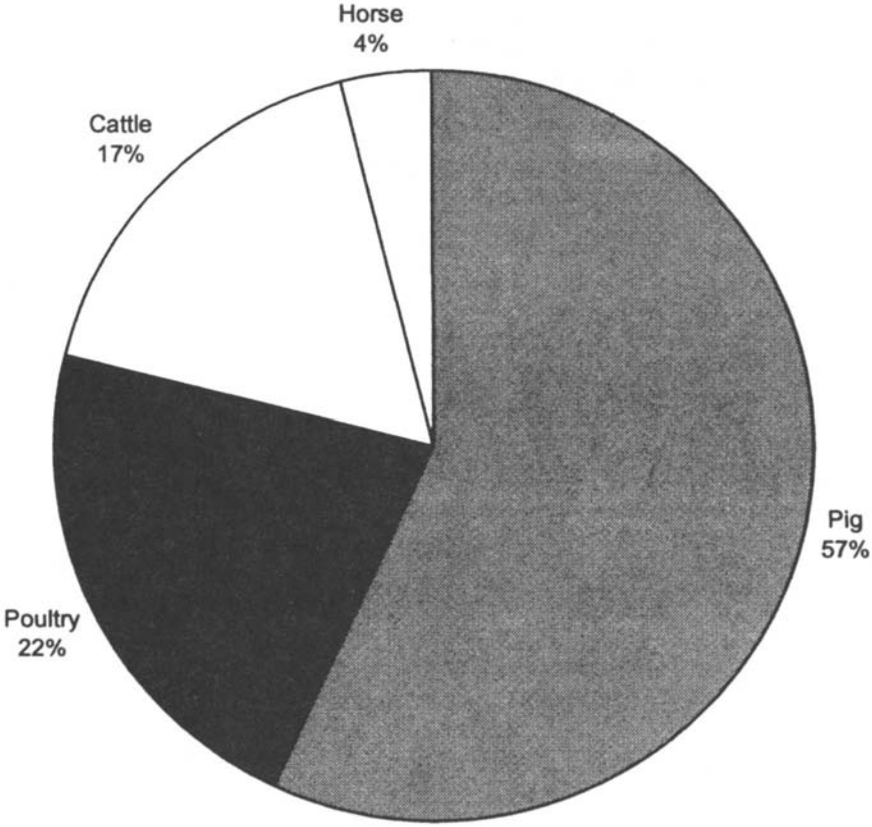


FIGURE 7-3. Odor Complaints from Different Types of Farms in England and Wales (Source: Pain 1994).

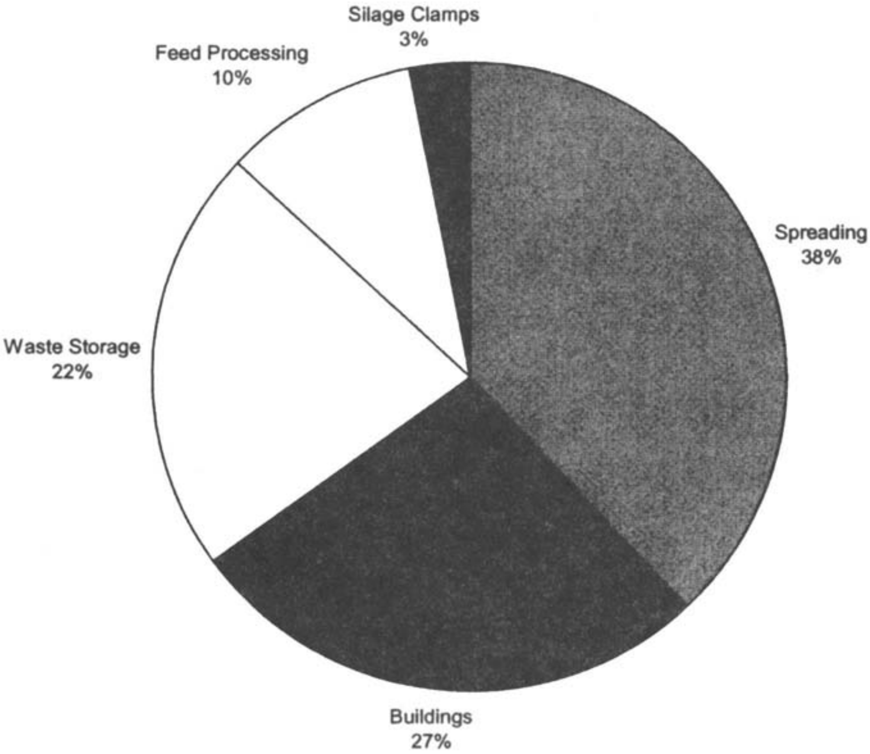


FIGURE 7-4. Sources of Odor Complaints from Livestock Farms in England and Wales (Source: Pain 1994).

their own operations and that about 11% have been subjected to odor complaints.

Environmental impacts of odorants from AFOs are largely unknown. No clear evidence exists that odor is directly injurious to human health; many of the odorants occur in trace amounts and are normally less than the amount that can be considered toxic. For example, Zahn et al. (1997) observed that the concentration of VOCs in swine production facilities is often one order of magnitude below the recommendations for time-weighted average and short-term exposure limits. However, NRC (1979) indicated that odor could elicit a wide range of physiological responses that range from irritation of the eyes, nose, and throat to nausea, headache, and vomiting to disturbance, annoyance, and depression. Based on a review of results of recent research in the United States, Thu (2002) concluded that neighbors of large-scale swine production operations can experience health problems at significantly higher rates than controlled comparison populations. Schiffman et al. (1995) reported that people living near a commercial swine operation who noticed the odors had significantly more tension, more depression, less vigor, more fatigue, and more confusion than control subjects on the same days. In a related study (Thu et al. 1997), neighbors within 3.2 km of a 4,000-sow swine facility in Iowa reported increased incidences of a number of interrelated symptoms, including headaches, respiratory problems, eye irritation, nausea, weakness, and chest tightness. Palmquist et al. (1997) noted that the presence of livestock operations in an area could result in decreased value of real estate properties. They found that proximity to swine operations in southeastern North Carolina caused a significant reduction in house prices of up to 9%, depending on the number of pigs and their distance from the house.

7.1.1.6 Insects. Animal waste can serve as a breeding ground for flies and mosquitoes. While flies and mosquitoes are not themselves air pollutants, they are a nuisance and can serve as major vectors of diseases in AFOs. Keeping the waste dry or flushing the waste regularly from confinement areas to a lagoon can reduce fly problems, reduce the chance for disease transmission within herds and flocks, and reduce the potential for nuisance complaints from neighbors. *Culex pipiens* or the southern house mosquito is the most common species in lagoons in the southern United States (Steelman et al. 1967). Axtell et al. (1975) noted that use of several insecticides and insect growth regulators has been effective in controlling mosquitoes in swine waste lagoons.

7.1.2 Air Pollutant Emissions from Animal Waste Containment Systems

Manure storage systems are perceived as the primary gas and odor sources in livestock operations even though, as indicated in Figure 7-4, odors from buildings and land application of manure have resulted in more complaints, at least in England and Wales. This perception is reinforced by high odor emissions from storage units with more manure solids and by turnover episodes in earthen basins that result from temperature changes in the spring and fall (Veenhuizen 1996).

7.1.2.1 Published/measured emission rates. Limited research has measured emission rates of compounds from animal waste containment systems. Most studies are based on small tanks or small samples, resulting in difficulties in evaluating emission rates. Field and laboratory research has largely focused on measuring concentration of odor or odorants in air within and in close proximity to confinement buildings and open lot feeding systems (Sweeten et al. 2000). Additionally, standard methods and procedures are generally lacking, making it difficult to compare results from different studies. Although generally accepted sample collection techniques typically have been used, test conditions that provide a standard basis for comparisons have not been established (USEPA 2001). The USEPA (2001) noted that available estimates of emission factors, rates, and concentrations from AFOs are sufficiently uncertain and provide a poor basis for regulating or managing air emissions from these operations. Evidently, more research needs to be done to establish emission rates, flux rates, and emission factors of various gases and odors from AFOs, including the waste storage and treatment systems.

Several methods have been used to estimate emissions from animal waste containment systems. Methods for estimating NH_3 emissions from waste containment systems include mass balance, micrometeorological, dynamic flow-through chamber, and trace gas techniques (Arogo et al. 2001; Liang et al. 2002). Recent research on determining emissions from animal waste containment systems is described in the following paragraphs.

Experiments with tanks of 4.5 m^3 capacity were conducted in the Netherlands by De Bode (1991). Odor concentrations were 200 and 120 odor units/m^3 of air for pig slurries in summer and winter, respectively, and 110 and 60 odor units/m^3 for cattle slurries. These values are low compared with those normally associated with buildings or land application of manure.

Carney and Dodd (1989) measured odors from different types of storage systems. They found that the strongest odors came from tanks in which slurry was agitated; threshold values were 0.45% relative concentration (222 odor units/m^3 of air) for agitated poultry slurry, 0.5% (200) for pig slurry, and 0.6% (167) for cow slurry. Also, slurries that were not agitated had a mean relative concentration of 1.56% (64), with slightly higher values for pig and poultry slurries than for cow slurry.

Hobbs et al. (1995) developed an odor emission chamber that can contain 200 L of waste. The chamber consisted of an enclosed system of stainless steel ducts (0.5 m × 0.5 m internal section) whose ends were connected into a Tedlar bag. The air in the system was pressurized by the bag. Odor samples were taken through the ports; concentrations of odorants, CH_4, H_2S, and CO_2 were measured. Preliminary emission rates from the slurry surface were determined over a 3-h period from 200 L of pig slurry with 1.0 m^2 surface area in an air stream moving at 1 m^3/s. Emission rates were 10,000 to 30,000 odor units/s for the more odorous samples; 0.3 to 30 mg/s for H_2S; 1.5 mg/s for acetic acid, phenol, and 4-methyl phenol; and about 20 μg/s for the other compounds. Emission rates of CO_2 and CH_4 were in the respective ranges of 1 to 5 mg/s and 0.02 to 0.05 mg/s.

Hobbs et al. (1999) used the same chamber to determine the emission rates of gases and compounds from pig slurry and their cumulative odors as perceived by a sensory panel over a 112-day storage period. Emission rates of H_2S, VFAs, phenols, and CO_2 decreased over the storage period, whereas those of CH_4 and NH_3 increased. No correlation was found between the olfactory response and any individual odor component. Table 7-5 summarizes the average daily emission rates from the pig slurry in this study. Note that these emission rates were obtained from stirred slurry and could explain the relatively high rate for H_2S.

Heber (1998) measured odor emission from an aerated lagoon in a 6,000-head swine grow-finish facility using a dynamic chamber with simulated wind speeds of 1.1 m/s. Emission rates ranged from 89 to 123 odor units/m^2-min and averaged 100 odor units/m^2-min. Compared to a similar lagoon that was not aerated, the aerated lagoon emitted 82% less odor while having twice the volumetric loading rate.

Aneja et al. (2001) determined NH_3-N flux from six anaerobic swine waste storage and treatment lagoons (primary, secondary, and tertiary) in North Carolina using a dynamic chamber system. Measurements occurred during the fall of 1998 through the early spring of 1999, and each

TABLE 7-5. Average Daily Emission Rates for Gases and Odors from Pig Slurry Stores

Gas/Odorant	Emission Rate (g/m^2-day)
Carbon dioxide	626
Methane	21.4
Hydrogen sulfide	66.6
Ammonia	4.35
Acetic acid	1.49
Propanoic acid	0.067
2-methyl propanoic acid	0.040
Butanoic acid	0.057
3-methyl butanoic acid	0.069
2-methyl butanoic acid	0.067
Pentanoic acid	0.015
Phenol	0.018
4-methyl phenol	0.442
4-ethyl phenol	0.027
Indole	<0.001
3-methyl indole	<0.002

Source: Hobbs et al. 1999.

lagoon was examined for approximately 1 week. Analysis of flux variation was made with respect to lagoon surface water temperature (~15 cm below the surface), lagoon water pH, total aqueous phase $NH_x(= NH_3 + NH_4^+)$ concentration, and Total Kjeldahl Nitrogen (TKN). Average lagoon temperatures (across all six lagoons) ranged from approximately 10.3°C to 23.3°C. The pH values ranged from 6.8 to 8.1. Aqueous NH_x concentration ranged from 37 to 909 mg N/L, and TKN varied from 87 to 950 mg N/L. Fluxes were the largest at the primary lagoon in Kenansville, N.C. (March 1999), with an average value of 120 μg N/m^2-min, and smallest at the tertiary lagoon in Rocky Mount, N.C. (November 1998) at 41 μg N/m^2-min. Emission rates were found to be correlated with both surface lagoon water temperature and aqueous NH_x concentration.

Liang et al. (2002) summarized recent measured emission rates of NH_3 from primary swine anaerobic lagoons (Table 7-6). Measured emission rates varied widely, ranging from 0.46 to 156 kg NH_3-N/ha-day. For each combination of measurement period and measurement method, measured rates also varied widely, with ratios of maximum-to-minimum rates of 1.4 to 8.9. Seasonal differences in emission rates also can be seen, with the ratio of summer to winter rates being as large as 10 or more.

*7.1.2.2 **Factors affecting pollutant emission rates.*** Air emissions are likely to be influenced by the size and geometry of the storage system, type and concentration of waste, period of storage, and environmental conditions. One of the most important factors affecting the amounts of gases and odorants produced is how the manure is managed, because certain types of storage and treatment systems promote an anaerobic environment (USEPA 1997a). Liquid waste management systems tend to produce significant quantities of CH_4, whereas solid waste management systems produce little or none. Higher temperature and moist climatic conditions also promote CH_4 production.

Because the process of manure degradation is mainly dependent on the presence and activity of microorganisms, the level and type of microbial activity largely determines the amounts and types of air emissions. Microbes are sensitive to moisture content, temperature, pH, oxygen concentration, and other environmental parameters. Therefore, any changes in these parameters will alter the decomposition process and, hence, the air emissions.

The emission process is influenced strongly by the chemical and physical conditions of the surfaces where contact with air occurs. Emission rates of many VOCs and fixed gases will depend greatly on conditions that alter the solubilities or vapor pressures (Zahn 1998). Environmental factors that influence solubility and vapor pressure of odorants in solution include source concentration, surface area, net radiation, air temperature, wind velocity, humidity, solution pH, and surface composition and

TABLE 7-6. Emission Rates of Ammonia from Primary Anaerobic Swine Lagoons

Farm Type	Measurement Period	Measurement Method	Area (ha)	TAN[a] (mg/L)	Emission Rate (kg NH_3-N/ha-day)	Reference
Farrow-finish	Aug.–Oct.	Micrometeorological	0.39	917–934	73–130	Zahn et al. 2001
Farrow-finish	Summer	Micrometeorological	3.5	230–238	3.2–40	Harper et al. 2000
Farrow-finish	Winter	Micrometeorological	3.5	239–269	1.3–1.9	Harper et al. 2000
Farrow-finish	Spring	Micrometeorological	3.5	278–298	3.1–9.8	Harper et al. 2000
Farrow-finish	Summer	Micrometeorological	2.7	574	15.4–22	Harper and Sharpe 1998
Farrow-finish	Winter	Micrometeorological	2.7	538	4.7–12.1	Harper and Sharpe 1998
Farrow-finish	Spring	Micrometeorological	2.7	741	5.2–15.4	Harper and Sharpe 1998
Farrow-wean	Summer	Micrometeorological	2.4	193	2.9–8.4	Harper and Sharpe 1998
Farrow-wean	Winter	Micrometeorological	2.4	183	6.0–9.1	Harper and Sharpe 1998
Farrow-wean	Spring	Micrometeorological	2.4	227	3.0–6.6	Harper and Sharpe 1998
Farrow-finish	Summer	Dynamic chamber	2.5	—	34–123	Aneja et al. 2000
Farrow-finish	Fall	Dynamic chamber	2.5	—	5.3–28	Aneja et al. 2000
Farrow-finish	Winter	Dynamic chamber	2.5	—	1.3–10	Aneja et al. 2000
Farrow-finish	Spring	Dynamic chamber	2.5	—	12.3–52	Aneja et al. 2000
Farrow-finish	May	TG OP-FTIR[b]	2.5	—	37–122	Todd et al. 2001
Farrow-finish	Nov.	TG OP-FTIR	2.5	—	7.8–67.6	Todd et al. 2001
Farrow-finish	Sep.	Dyn. flow-through chamber	0.78	101–110	0.57–3.5	Aneja et al. 2001
Farrow-finish	Nov.	Dyn. flow-through chamber	0.36	350	0.46–1.73	Aneja et al. 2001
Finish	Feb.–Mar.	Dyn. flow-through chamber	0.49	543–560	0.72–5.39	Aneja et al. 2001
Finish	Mar.	Dyn. flow-through chamber	0.39	709–909	0.82–2.95	Aneja et al. 2001
Breed-wean	Apr.–Jul.	Dyn. flow-through chamber	3.07	978–1143	104	Heber et al. 2001
Breed-wean	May–Jul.	Dyn. flow-through chamber	1.24	326–387	39	Heber et al. 2001

Source: Liang et al. 2002.

TAN = Total ammoniacal nitrogen; TG OP-FTIR = Tracer gas open-path Fourier transform infrared spectroscopy.

homogeneity (Zahn 1998). The relative effects of these factors are described in the following paragraphs.

Source concentration. The concentration of compounds present in the liquid phase is one of the most important factors affecting emission potential of the system (Zahn 1998). The emission of a compound from the surface of a liquid is proportional to the difference in concentrations of the compound in the liquid and gas phases. In the study conducted by Aneja et al. (2001), described in Section 7.1.2.1, the NH_3-N flux was modeled as

$$\ln(\mathrm{NH_3\text{-}N\ flux}) = 1.0788 + 0.0406T_L + 0.0015([\mathrm{NH_x}])\ (R^2 = 0.74)$$

where

NH_3-N flux = flux rate of NH_3 from the lagoon surface ($\mu g/m^2$-min)
T_L = lagoon surface water temperature (°C)
$[NH_x]$ = total NH_3-N concentration (mg/L)

Surface area. The emission rate depends on the surface area of the liquid. If the concentration of material in solution remains the same and the surface area increases, emission rate is expected to increase proportionally with the change in surface area. For example, for acetic acid, a common odorant in swine production, a change in the surface area from 18 to 62 cm^2 (3.5-fold increase) resulted in a 3.7-fold increase in emission (Zahn 1998).

Net radiation and temperature. Controlled environment studies evaluating the effects of humidity, temperature, wind velocity, and net radiation have suggested that net radiation is individually the most significant factor in volatilization of VOCs from stored swine wastes (Zahn 1998). Net radiation tends to increase the temperature of the liquid and the vapor pressure of chemical components present in the waste material. For example, a change of 25°C can elevate the vapor pressures of acetic, propionic, and butyric acids by nearly one order of magnitude (i.e., from 0.6 kPa to 6 kPa) (Gallant and Yaws 1993).

Arogo et al. (2001) indicated that an increase in temperature increases the mineralization of organic nitrogen in slurry, thereby increasing the production and emission of NH_3. According to Nicolai (1996), the manure decomposition process almost stops at temperatures below 4.4°C. Therefore, lagoons and storage units in northern areas of the United States become inactive during the winter. As the unit warms up in the spring, the contents turn over, that is, the bottom waste rises and the top waste drops. Because of minimal bacterial action during the winter, the odor-producing material from the bottom, which has not decomposed during the winter, rises to the surface. The higher springtime temperatures increase microbial action and result in higher odor levels during this turnover period.

Air velocity. High wind speed across a liquid manure surface enhances the mass transfer rate from the liquid to the gas phase. Thus, high wind speed over the liquid surface of an animal waste containment unit would greatly increase the emission rate. Generally, this is true for soluble gases, such as NH_3, and for organic compounds whose volatilizations are controlled mainly by the gas film (Arogo et al. 1999). Air velocity tends to reduce the boundary layer thickness over a liquid surface which, in turn, lowers the resistance to the volatilization process. Consequently, the mass transfer coefficient and mass transfer rate will be higher. Wind also can promote mixing of liquids and increase the concentration of the odorant at the surface.

Much of the work relating to the effect of air velocity on emission has been done on NH_3 and under controlled laboratory conditions. Cumby et al. (1995) observed that changing air velocity from 1.4 to 5.6 m/s across open surfaces of mixed aqueous NH_3 solutions increased the mass transfer coefficient by approximately 60%. They also noted that the mean NH_3 emission rates from open surfaces of pig slurry were 3.6 and 4.2 g/m^2-day with wind speeds of 1.4 and 4.3 m/s, respectively.

Arogo et al. (1999) evaluated the mass transfer coefficient of NH_3 from liquid pig manure in a convective emission chamber where the air velocity, temperature, turbulence, and relative humidity were controlled. They found that the mass transfer coefficient tended to increase with increasing air velocity and liquid temperature.

Humidity. According to Zhan (1998), humidity has a slight inhibitory effect on the volatilization of compounds from a waste storage system. For slightly soluble gases, such as dimethyl sulfide, CH_4, and other VOCs, the aqueous boundary layer (at the solution/air interface) tends to retard transport of these compounds into the atmosphere (MacIntyre et al. 1995). The rate limiting the movement of these gases into the atmosphere is then the exchange across the air/liquid interface. High relative humidity tends to stabilize the air/liquid boundary, whereas low humidity tends to destabilize the interface (Zahn 1998). Destabilized boundary layers tend to have higher volatilization than stabilized boundary layers.

pH. Solution pH alters ionization of organic and inorganic acids and bases and chemical/biological transformations occurring within the solution (Zahn 1998). Like temperature, the ionization state of acids and bases in solution greatly alters partial pressure and, therefore, emission rate from the solution. For example, for the weak base NH_3, at a pH of about 9.24, half of the species in solution are in the ammonium form and resistant to volatilization, and the other half are in the NH_3 form and capable of movement into the air (Zahn 1998). Increasing the acidity of the waste storage system reduces volatilization losses of NH_3 because it is converted to the ammonium form. Acids, such as VFAs, H_2S, and other organic sulfides, exhibit the opposite behavior.

Surface composition and homogeneity. The presence of crusts and scum layers on the surfaces of waste storage systems can reduce the transfer of volatile compounds into the ambient air. Floating scums have been recognized for several years as being effective for odor control (Miner and Suh 1997). These scums operate by promoting aerobic processing of the odorous compounds and protecting the liquid surface from wind action, which continually renews the gas layer and promotes rapid volatilization.

Although the relative effects of the above factors are well understood, insufficient scientific data is available to quantify the effects in detail. More research is needed to identify the kinetic release mechanisms for gases and odorants from waste containment systems. Liang et al. (2002) developed a mathematical model to estimate NH_3 emission from anaerobic swine lagoons based on the classical two-film theory. Inputs to the model were wind speed and lagoon liquid properties such as total ammoniacal nitrogen (TAN) concentration, pH, and temperature. Predicted emission rates of NH_3 increased when any of these parameters was increased, but the relationship was linear only with TAN concentration. The dissociation constant (K_d) for NH_3 in lagoon liquid also was an important factor, with higher predicted flux rates for higher K_d. The authors validated the model by comparing predicted results to measured flux rates from two lagoons in North Carolina. Predicted NH_3 flux rates for the two lagoons ranged from 1 to 38 kg NH_3-N/ha-day, which was wider than the measured flux rates (2.5 to 22 kg N/ha-day). The authors indicated that further investigation is needed on the K_d values for anaerobic lagoons; possible gradients in temperature, pH, and TAN near the liquid surface; and possible effects of turbulence and wave action on emission.

7.2 STRATEGIES FOR AIR QUALITY CONTROL

The major air pollution aspect of animal waste containment systems is the production of gases and odorants including H_2S, NH_3, and trace amounts of VOCs. Many aspects of the relevant processes involved in odor generation, emission, transport, and detection are not yet completely understood (ASAE 1998a). No federal standards or laws exist in the United States for specific odor control (Shultz and Collar 1993) but counties or municipalities may have local ordinances.

Complete elimination of odors around AFOs generally is neither technically nor economically feasible (Sweeten and Humenik 1984). However, limiting odor generation and minimizing odor complaints are possible and practical in many cases. ASAE (1998a) has developed an engineering practice for control of manure odors to assist engineers, pollution control officials, land use planners, and livestock producers in the location, planning, construction, and operation of livestock operations so that odor complaints can be minimized. Numerous publications have summarized the state of the art of odor control, especially for swine operations (e.g., Ritter 1989; Jacobs 1994; Miner 1995; Wilson 1996; Odor Control Report Task Force 1998; Sweeten et al. 2000; Jacobson et al. 2001).

Effective air quality control requires reducing air emissions from various sources, including buildings, waste storage facilities, and land application of manure. Many technologies for control of odor and odorants have been developed over the last few decades; however, most of the technologies have not been evaluated properly or systematically (Sweeten et al. 2000). Additionally, the USEPA (2001) indicated that much of the information on control methods was developed from pilot-scale or research studies and that there is a general paucity of credible scientific information on the effects of abatement measures on concentrations, rates, and fates of air emissions from AFOs.

Air emissions can be controlled using various waste treatment techniques by inhibiting the formation of emitted substances, suppressing emissions of substances once formed, or capturing or controlling a substance that is emitted (USEPA 2001). In addition, air quality problems can be minimized through proper site selection, design and construction of the facility, and operation and management of the system. These methods are described in the following paragraphs.

7.2.1 Waste Treatment Technologies

Undesirable odors associated with waste treatment are generally a result of anaerobic bacterial activity. Anaerobic liquid systems, in which the waste is diluted with water for ease in handling, provide an environment more conducive for the production of odorants than does a dry or an adequately aerated system. Odors from diluted anaerobic liquid manure tend to be more offensive than odors from the undiluted fresh manure. As such, inhibiting anaerobic decomposition of manure (e.g., through drying and aeration) can reduce odor emission (ASAE 1998a). Chemicals also can be added to the manure to prevent release of or overcome odorants. Other waste management technologies include solid–liquid separation to reduce the source strength and capture of the gases and odorants.

7.2.1.1 Aeration. Aeration to maintain dissolved oxygen can replace anaerobic decomposition with the nearly odorless aerobic process. The two basic aerobic treatment methods are (1) completely mixed aeration in a treatment vessel, and (2) extended surface aeration in a manure storage or treatment lagoon (ASAE 1998a). If aerobic treatment is sufficient to stabilize the waste, it can be stored for a long period without causing odor problems (Westerman and Zhang 1997). Complete stabilization of the waste can be achieved with a minimum oxygenation capacity of twice the daily BOD loading, a hydraulic retention time of 10 days or more, and a dissolved oxygen level of 1 to 2 mg/L in the aeration basin (NZAEI 1984). Westerman and Zhang (1997) estimated that, for waste stabilization with continuous aeration, the total power cost for running the system would be about $0.039/day-pig space (each space will provide approximately 2.6 pigs/yr). This estimate was based on the following assumptions: a 68-kg pig produces 0.21 kg BOD_5/day; twice the BOD loading is required for stabilization; aeration efficiency of mechanical aerators is 1 kg O_2/kWh; and cost of electricity is $0.09/kWh.

Because complete stabilization of livestock manure by aerobic treatment normally is not economically feasible, lower levels of aeration have been recommended and used for partial odor control (Westerman and Zhang 1997). Oxygenation capacity to supply 33% to 50% of the BOD load has been recommended (NZAEI 1984). Using a lower rate of aeration, compared to that required for complete stabilization, reduces the release of VFAs and other odorants and also allows some oxidation to less odorous compounds. Use of low-rate aeration will favor the growth of facultative microbes (those that can live with or without air) and can provide some reduction in strength as well as control odor. Based on the above assumptions for complete stabilization, Westerman and Zhang (1997) estimated that, if 33% of the BOD load is used for odor control, the power cost for running the system would be about $0.013/day-pig space ($4.67/yr-pig space).

Mechanical aeration for partial odor control can be confined to the upper layer of the lagoon, thereby reducing surface area required for oxygen diffusion and energy required for aeration (Westerman and Zhang 1997). Floating aerators, perforated pipes, rotating aerators, and rotary blowers appear to be satisfactory. Figure 7-5 summarizes the biological and chemical principles involved in surface aeration of anaerobic lagoons. According to Zhang et al. (1997), such a system takes advantage of both anaerobic and aerobic degradation processes. Anaerobic degradation decomposes and stabilizes the influent manure at the bottom of the lagoon; aerobic degradation, on the other hand, controls odor emission at the surface of the lagoon.

Recent research has investigated surface aeration of anaerobic lagoons (Schulz and Barnes 1990; Zhang et al. 1997; Heber 1998). Schulz and Barnes (1990) reported that the redox potential in the surface layer needed to be maintained greater than −76-mV Eh by continuous aeration in order to maintain a nonodorous environment. In two case studies of full-scale

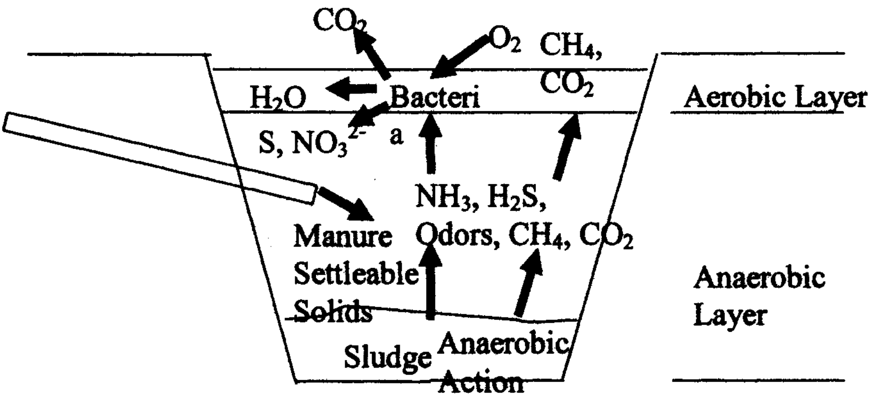


FIGURE 7-5. Surface Aeration of Anaerobic Lagoons for Odor Control (Source: Zhang et al. 1997).

operations, mechanical surface aerators were successful in preventing the generation of odors.

Zhang et al. (1997) studied, under laboratory conditions, the surface aeration of swine manure stored in anaerobic lagoons for odor control. The effects of different aeration rates and depths on reduction in emission of odorous gases were determined for both continuous and intermittent aeration processes. Surface aeration was found to be effective for controlling odors of anaerobic lagoons. Continuous low-rate aeration to maintain the dissolved oxygen in the surface liquid layer at 0.5 to 2.5 mg/L was effective for odor control but resulted in high NH_3 emission rates. Intermittent aeration was found to be a feasible approach to minimize the energy requirement for aeration, while still effectively controlling the odors of the manure.

Heber (1998) investigated the odor-reduction potential of surface aeration using a static-tube aeration system in a full-scale lagoon. He noted that the aerated lagoon emitted 82% less odor at twice the volumetric loading rate compared to an unaerated lagoon.

7.2.1.2 ***Anaerobic treatment.*** Application of anaerobic lagoons in the treatment of animal waste is widespread in the United States because of their low initial cost, ease of operation, and convenience of loading by gravity flow from livestock buildings (Day and Funk 1998). However, they tend to produce objectionable odors, especially during spring warm-up. The operation and management of anaerobic lagoons to minimize odor release are described in Section 7.2.2.3.

Anaerobic digestion is a process that controls the degradation process and results in the generation of biogas, which then can be used to produce electricity (Jacobson et al. 1998). Anaerobic digesters use the same microbial processes for stabilizing manure as anaerobic lagoons; however, they are closed reactors that are heated and possibly mixed to optimize the production of CH_4 from the anaerobic decomposition process (USEPA 2001). A small number of full-scale anaerobic digesters are in operation at commercial dairy and swine farms. The biogas produced—which contains about 60% to 70% CH_4, about 30% to 40% CO_2, and trace amounts of H_2S, VOCs, and moisture—is used either as a boiler fuel for heating or fuel for engine-generator sets to produce electricity (USEPA 2001). In general, use of anaerobic digesters is technically feasible (Clanton 1997); however, economics, equipment maintenance costs, erratic biogas production, and requirements for increased managerial skill have limited the adoption of this technology for manure utilization (Odor Control Report Task Force 1998). The U.S. Environmental Protection Agency (USEPA) has established the AgStar Program and developed a handbook to promote the recovery and utilization of biogas resulting from anaerobic digestion of livestock wastes (Roos and Moser 1997). The handbook provides guidance in conducting a technical and economic feasibility assessment of biogas technology at a specific facility.

7.2.1.3 ***Composting.*** Composting is a solid waste management strategy in which the organic component of solid waste is decomposed biologically under controlled conditions to a state in which it can be handled, stored, and applied to cropland without adversely affecting the environment (Golueke 1977). Compost piles are either aerated continuously using air forced upward through the pile or turned periodically to ensure aerobic conditions (USEPA 2001). For effective composting, the moisture content of the manure needs to be 40% to 60% (ASAE 1998a). In general, composting can reduce offensive odors, reduce difficulties in handling due to stickiness, inactivate pathogens, and stabilize the organic constituents, thus producing a uniform organic material suitable for soil application (Harada 1990). When properly operated, organic compounds are degraded with the oxidation of organic carbon to CO_2; in addition, reduced sulfur compounds are oxidized to sulfates (USEPA 2001).

7.2.1.4 ***Chemical treatment.*** Chemical treatments of manure have been developed to prevent or overcome odorants. Some volatile compounds present in the feed, manure, or manure slurry can be converted to a less volatile or less odorous form by pH control or by biologic conversion (NRC 1979). An example is the addition of lime as a base to control the release of H_2S (Day 1966). The dissociation of H_2S is a strong function of pH; if the pH is raised above 9.5, the escape of H_2S tends to be insignificant (NRC 1979). Sulfides are oxidized to sulfates under aerobic waste treatment, as is practiced in oxidation ditches and aerated lagoons (NRC

1979). Paraformaldehyde has been added to manure as a means of converting NH_3 to nonvolatile hexamethyl-enetetramine (Seltzer et al. 1969).

Addition of odor-control chemicals to manure storage tanks has attracted widespread interest in recent years. These chemicals act by preventing the release of odorous compounds, inhibiting their formation, or masking their odor. A wide variety of products have been marketed to treat and prevent odors in feedlots, liquid waste storage tanks, and lagoons (Ritter 1989). These odor-control agents can be classified into the following categories (ASAE 1998a):

- Masking agents: Mixtures of aromatic oils that have a strong but more acceptable odor of their own, designed to cover up the objectionable waste odor.
- Counteractants: Mixtures of aromatic oils that tend to neutralize the waste odor so that the intensity of the mixture is less than that of the individual constituents.
- Deodorants: Strong oxidizing agents that may inhibit microbial activities, alter the digestive process, or change the chemistry of the odorants.
- Adsorbents: Products with large surface areas that are used to adsorb the odorants before they are released to the environment.

Because of the expense and the limited information on the performance of odor control chemicals, their use generally has been limited to short-term applications in particularly offensive areas, such as manure storage pits immediately before hauling (ASAE 1998a). Schmidt et al. (1996) reported that 11% of Minnesota swine producers surveyed had used some odor-control additives with various levels of effectiveness in odor reduction.

Controlled laboratory and field tests are underway to evaluate or document the effectiveness of odor-control chemicals. Because the performance of these chemicals will depend greatly on management and environmental conditions, a uniform and standard procedure is needed to evaluate the effectiveness of these chemicals. Various protocols have been developed and proposed (Miner et al. 1995; Williams and Schiffman 1995; Zhu et al. 1997; NPPC 1998).

7.2.1.5 ***Drying.*** When the moisture content of the manure is lowered to 50% or less, it is sufficiently porous to permit air diffusion and to preclude anaerobic decomposition (ASAE 1998a). The advantages of drying include (1) reduction in odor production, (2) less material to handle, (3) ease of storage, and (4) conversion of useful products.

7.2.1.6 ***Solid–liquid separation.*** Separating the solids from the liquid phase can reduce odor emissions from AFOs, especially those with anaerobic lagoons (Lorimor and Edwards 1998g). Manure solids can be separated using settling tanks or mechanical separators. Settling tanks work well for swine manure; mechanical separators, on the other hand, work well for beef and dairy cattle manure. As the solids are removed, less manure is loaded into lagoons, resulting in reduced odor. Smaller lagoons (i.e., those with less surface area) or those that are more lightly loaded have less potential for odor production than larger or overloaded lagoons (Lorimor and Edwards 1998g). Separated solids are still quite wet, with the moisture content at least above 70% (Zhang and Westerman 1997); as such, they must be handled so that they do not become a significant odor source. Composting or drying immediately after separation will help keep the odors of manure solids at a minimum level (Zhang and Westerman 1997).

*7.2.1.7 **Capture or physical confinement of odorants.*** Release of odorants into the surroundings can be minimized by physical confinement of the odorants using manure storage covers (NRC 1979), including synthetic floating covers or biocovers (e.g., straw or chopped cornstalks). According to Jacobs (1994), storage covers reduce occasional manure agitation caused by wind and rain, movement of odorants from storage areas to neighboring residences, and addition of water from rain and snow, thereby also reducing the total volume of manure to be spread.

Size of the manure containment unit, climatic conditions, and costs are factors that should be considered in choosing whether to cover the unit and the cover that should be used. For example, plastic covers may not be economical for units with large surface areas, such as lagoons. Excessive precipitation tends to increase the management effort requirement for covers, and windy conditions will make it difficult to keep the biocovers in place. Additionally, biocovers will sink into the manure storage area over time, add organic matter to the slurry, increase the loading rate, and increase the cost of manure separation.

Odor reduction efficiencies of different covers are summarized by Mannebeck (1985) and are presented in Figure 7-6. Roofs and impermeable covers can provide odor reduction efficiencies from 95% to 100% compared to an uncovered waste storage unit.

Recent research has evaluated the effectiveness of various types of covers, mostly under small-scale or laboratory conditions. Using small tanks, Bundy et al. (1997) evaluated different covering materials that were applied on the manure surfaces. Odor concentrations above the covers and liquid manure surfaces were sampled and analyzed weekly using olfactometry. The authors observed that most of the covering materials reduced the emission of odorants from manure surfaces. They concluded that straw and chopped cornstalk covers may be a cost-effective alternative for reducing odor emission from a surface; a thickness of 15 cm was

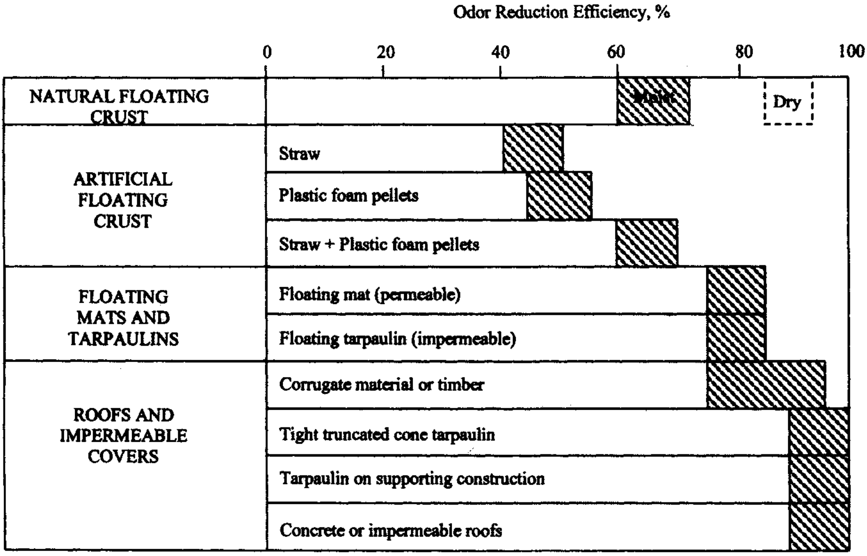


FIGURE 7-6. Odor Reduction Efficiency of Different Types of Covers (Source: Mannebeck 1985).

required to maintain cover integrity through the summer. However, these cover materials did not stay afloat throughout the winter.

Miner and Suh (1997) evaluated different polystyrene foam materials ranging in thickness from 1 to 3 cm for their ability to reduce the escape of NH_3 and odors from the liquid surface of stored swine manure. They observed that the NH_3 concentrations were from 45% to 90% less over the covers than over the control (uncovered manure). Additionally, odor panelists were able to identify a reduction in odor intensity as a result of using covers.

Hornig et al. (1999) investigated the odor emission reduction and floating behavior of low-cost covers (i.e., commercially available granules, chopped straw, and rape oil) under laboratory and field conditions. They also considered tent roofs consisting of polyethylene film set up with cables. Granules and straw covers reduced odor emission by about 83% to 91% and NH_3 emission by about 83% to 91%. Tent roofs reduced odor and NH_3 emissions by 82% and 99.5%, respectively. The researchers recommended that at least 4 kg of straw be used per square meter of slurry surface because windy conditions destroyed the chopped straw covers when not enough straw was used.

Clanton et al. (1999) studied seven cover treatments including no cover (control), wheat straw mat, vegetable oil mat, straw/oil mat, clay ball mat, polyvinyl chloride (PVC)/rubber membrane, and geotextile fabric. All covers reduced odor and H_2S concentrations at various points in the study, although reduction was inconsistent. The straw mat and PVC/rubber membrane significantly and consistently reduced both odors and H_2S during the first three collection periods. Mixing vegetable oil with straw appeared to increase longevity of the cover. Clay balls reduced emissions, but not as effectively as other covers. A geotextile fabric was considered to be a possible cover choice, because the fabric was self-floating and supported a biofilm that could self-seal the cover. A straw (possibly mixed with oil) mat and PVC/rubber membrane appeared to be the most effective covers for reducing both odors and H_2S emissions.

Based on the Clanton et al. (1999) study, Clanton et al. (2001) evaluated four thicknesses of geotextile fabric and four thicknesses of straw mat floating covers using laboratory-scale swine and dairy manure storage units. There were 38 160-L open-topped PVC pipes (columns) containing swine or dairy manure used. Collected air samples were tested for odor strength using a dynamic olfactometer, for H_2S concentration using a Jerome® meter, and for NH_3 using a boric acid trap. Straw thickness and time significantly affected odor, H_2S, and NH_3 reductions, while manure type and geotextile thickness did not affect these reductions. The recommended minimum thickness of straw may be 20 cm, since little additional reduction in odor, H_2S, or NH_3 was gained by increasing thickness to 30 cm. However, 30-cm (11.8-in.) depth may be needed to keep the straw material afloat or to keep the upper portion dry to allow the straw to absorb gases and/or act as a biofilter. Even though the geotextile did not significantly impact reduction of odor, H_2S, and NH_3, the fabric may play a role in creating a barrier between the manure and straw to increase the life and effectiveness of the straw cover.

Zahn et al. (2001) investigated the efficiency of a polymer biocover for abatement of H_2S and NH_3 emissions from an east-central Missouri swine lagoon with a total surface area of 0.78 ha. Flux rates were monitored continuously from two adjacent circular control and treatment plots using a nonintrusive micrometeorological method during three independent sampling periods. The authors observed that polymer biocovers could reduce the rate of NH_3 and H_2S emission from lagoon surface by up to 58% when compared to control means. Additionally, the biocover enhanced the rate of anaerobic digestion by 25% when compared to the control.

7.2.2 Production Strategies

Air quality problems can be minimized if specific attention is given to site selection, design and construction of the facility, and operation and management of the system. Sweeten et al. (2000) indicated that overly optimistic assumptions have frequently been utilized as cost-saving measures in site selection and facility design. Such short-term expediencies could result in odor conflicts and environmentally unsustainable systems in later years.

7.2.2.1 Facility site selection. Site selection is the most important control strategy for a new livestock operation (NRC 1979). Transportation, feed supply, and zoning regulations may make a site ideally suited for an AFO; however, existing or proposed development in the area may make the site inappropriate. Factors that should be considered in site selection include proximity to urban areas, climatic conditions, and land area for waste management systems.

An AFO should be located at a reasonable distance from residential areas, places of employment, institutions, and other areas frequented by people other than its operators (ASAE 1998a). Although distances have not been established beyond which complaints are invalid, a livestock operation should be located more than 1.6 km from housing developments and 0.4 to 0.8 km from neighboring residences; greater separation distances should be considered for larger operations (ASAE 1998a). Van Kleeck and Bulley (1985) conducted a survey of neighbors around seven 100- to 225-sow farrow-to-finish operations to assess the relationship between the perception of odor nuisance, separation distance, and the facility size. They found that the frequency of perceiving swine farms as a nuisance was inversely proportional to the square of the separation distance. Also, about 20% of the neighbors living as far away as 0.67 km from a swine farm perceived it to be a nuisance; farm size appeared to have no effect at distances between 0.18 and 0.37 km.

Wind direction is also important in evaluating a site with regard to odors. Because most locations have winds from several directions during the year, locating "downwind" of development is not sufficient to ensure acceptability. From climatic data, the percentage of time that the wind is expected to blow from the operation to a receptor can be estimated. In this case, a more rational decision concerning site suitability can be made. When distance is used as the only criterion, it must be expected that odors can be transported further than a distance of 1.6 km downwind under appropriate climatic conditions (NRC 1979). For example, downhill airflows during ground surface inversion (i.e., cool air collects at ground surface and moves downhill) can cause repeated odor problems.

Adequate land area should be provided for waste management systems. Sweeten (1997) indicated that using a nutrient balance approach appropriate for the facility size and type of manure- and wastewater-management system will ensure considerable separation distance of the facility from nearby residences. He noted that, in many cases, the land area would be large enough to ensure an adequate buffer distance for odor control. For example, using the ASAE (1993) published values, he found that the land requirements for swine lagoon effluent irrigation or liquid manure application should be 162 or 445 ha, respectively, for nitrogen balance considerations, and 1,053 or 1,215 ha, respectively, for a phosphorus balance per 10,000-head capacity in a farrow-to-finish operation. If these spaces are provided, separation distances of 0.70 to 1.95 km will be available for odor dispersion.

The USEPA (1997b) has attempted to integrate water and air quality issues for evaluating the site selection of future AFOs. Region 6 of the USEPA developed a screening tool, Cumulative Risk Index Analysis (CRIA), to consider the potential for significant, cumulative, and environmental effects of confined swine feeding operations (USEPA 1997b). The CRIA evaluates the potential environmental effects of a proposed swine feeding operation by a watershed subunit, which is created by merging watershed area data and state information on stream segments. It evaluates not only water quality criteria but also issues that pertain to both water and air quality, such as the density of nearby AFOs, land application technology, storage capacity of liquid manure, nutrient budget for land application purposes, and the proximity of the proposed swine-feeding operation to development. The scoring model also evaluates the potential for odor by multiplying the number of swine proposed for the feeding operation by the threshold of 750 animals to obtain a scoring ranging from 1 to 5. Because the CRIA considers a total of 31 criteria, the potential for odor/air quality problem alone may not play a significant role in the overall score for the siting of a potential swine-feeding operation. Currently, the USEPA uses CRIA as an environmental assessment tool to assist the agency in making decisions, rather than solely relying on the scoring model to evaluate the potential effects from an AFO.

*7.2.2.2 **Planning, design, and construction.*** Appropriate selection of manure storage and treatment systems can help minimize odor problems (ASAE 1998a). ANSI/ASAE EP 403.2 describes the minimum criteria for design and construction of anaerobic animal waste lagoons located in predominantly rural or agricultural areas (ASAE 1998b). Jacobs (1994) also presented the following guidelines to reduce the potential for nuisance odors from manure storage:

1. Provide additional storage volume for greater flexibility in the timing of manure application.
2. With solid and semisolid manure management systems, separate the liquid from the solid portions in storage to reduce the promotion of anaerobic conditions.
3. Avoid the addition of silage effluent and waste food products to the manure storage reservoir.

*7.2.2.3 **Operation and management.*** The operation and management of the livestock production facility also offers considerable opportunity for controlling odor. Overflowing manure storage tanks, malfunctioning manure scrapers, leaking waterers, and ruptured retention ponds and dikes are among the most common causes of odor complaints (NRC 1979). A clean, orderly appearance of the livestock production unit, including landscaping, can be effective in suggesting a nonoffensive condition (ASAE 1998a).

Anaerobic lagoons for swine waste treatment are of special concern in odor control (NRC 1979). Properly designed and managed lagoons will not be free of odors but will be much less likely to cause serious odor problems than over- or shock-loaded lagoons. If multiple-celled lagoons are used, the cell or cells receiving fresh manure should not be loaded in excess of the recommendations for a particular area. Anaerobic lagoon odors are most common in the late spring and early summer, when the water temperature increases and manure accumulated during the winter undergoes rapid decomposition. Odor control strategies for anaerobic lagoons include the following:

- Removing as much of the lagoon contents as possible and refilling the lagoon to the normal operating level with clean water.
- Lowering the loading rate (e.g., by building larger lagoons or reducing the number of animals served).
- Using a mechanical surface aerator for partial odor control.

Shelterbelts around waste storage and treatment systems can help disperse the odors. If they are upwind, they help deflect the air currents up and over the odor source, so that less odor is collected and carried offsite (Lorimor and Edwards 1998f). If they are downwind, they promote mixing and cause the air currents to rise, resulting in increased odor dilution (Lorimor and Edwards 1998f). Numerical simulation of the effects of tall barriers around manure lagoons predicted reductions in downwind odor lagoon emissions of 26% to 93% (Liu et al. 1996). Veenhuizen (1996) recommends windbreaks of two to three rows of trees to reduce aerosol drift.

*7.2.2.4 **Feeding practices and dietary manipulations.*** Improving feeding practices can help reduce the amount of waste produced and odor problems. For example, feed wastes can be reduced by using pelletized feeds and improving feed delivery systems and waterers. Numerous advances also have been made to improve feeding efficiency. Heber et al. (1999) reviewed previous research and identified several feasible methods to reduce odor from swine facilities:

- Provide good quality drinking water low in sulfates and nitrates.
- Implement split-sex feeding and/or phase feeding to enhance nutrient efficiency and reduce nutrient excretions.
- Formulate, grind, and/or pelletize feed properly to enhance digestibility of feeds, reduce feed losses, and reduce nitrogen excretion.

- Implement wet feeding (3:1 water-to-feed ratio) to reduce water and feed spillage.
- Add oils or fats to feed (1% or more) to reduce generation of dust, which can carry odors.

Recent research has evaluated the potential of removing or reducing some of the constituents in the diet. According to Sutton et al. (1997), current standard diets for pigs often meet the nutritional needs of the pigs for the most limiting amino acids, but exceed requirements for other amino acids, resulting in excessive nitrogen secretions. Removing or reducing constituents in the diet that can cause emissions is perhaps the most direct and the most economical strategy for reducing odorous emissions (Schulte 1997). Some of the promising pollution-prevention methods that are being developed are reduction and alteration of protein profiles, use of additives, and application of urease (Schulte 1997).

Sutton et al. (1997) reported that reducing the crude protein level of pig diets and supplementing with essential amino acids significantly reduced total nitrogen excretion and NH_3 concentration and altered the concentrations and ratios of selected volatile compounds in fresh manure. In addition, they found that low-level additions of an oligosaccharide and cellulose further reduced total and NH_3 nitrogen in fresh manure, indicating a stimulation of microbial fermentation in the colon.

In Kay and Lee's study (1997), growing and finishing pigs were offered either commercially available diets containing 225 and 205 g/kg crude protein or low crude-protein diets containing 165 and 140 g/kg crude protein, respectively. They found that the low crude-protein diets reduced the volume of slurry by 28% and also reduced NH_3 emission by more than 46% compared to commercial diets.

Hobbs and Pain (1996) evaluated two experimental reduced crude-protein diets for finishing pigs and compared them to a standard commercial diet. The experimental diets were formulated on a least-cost basis using either a full commercial diet or mainly home-grown (United Kingdom) cereals and pulses. Synthetic amino acids were included in both formulations to obtain essential amino acids in an ideal protein ratio, with the crude protein being as low as possible. Nitrogen excretion by both growing and finishing pigs that consumed the low crude-protein diets was significantly lower than that of the pigs that consumed the commercial diets. Major odorants responsible for odor from slurries were identified and their concentrations determined using gas chromatography-mass spectrometry. For growing pigs, feeding each of the low crude-protein diets significantly lowered the concentrations of 5 of 10 odorants compared to feeding the commercial diet. For finishing pigs, feeding the least-cost and home-grown diets significantly lowered the concentrations of 9 and 4 of 10 odorants, respectively, compared to feeding the commercial diet.

7.2.3 Economic Considerations

For most of the available control technologies described, information is limited about the cost that is derived from long-term operating experience under field conditions (USEPA 2001). Estimated costs of the various odor-control technologies ranged from about $0.25 per pig marketed for straw covers to over $4.00 per pig marketed for aerobic treatment (Table 7-7).

Jolly and Kliebenstein (1995) presented the major economic issues in livestock odor reduction. They noted that environmental problems associated with livestock production arise because markets fail to reflect all of the costs or benefits of production. Livestock odor or air quality is an "externality" of the production process; it is a cost not directly encountered in the production process, such as feed fed or interest. Jolly and Kliebenstein (1995) indicated that air quality improvement is associated with both costs and benefits and can be evaluated within a cost-benefit approach. Figure 7-7 presents the general expected relationship. Assuming that the optimum level of air quality is Q, the corresponding optimum level of expenditures is P. Air quality levels above Q, such as Q_1, have control benefits of P_1 and control costs of P_1^1. In this case, costs exceed benefits, so that expenditures should be reduced. For air quality levels below the optimum, such as Q_2, the control costs, P_2, are less than the benefits, P_2^1. In this case, expenditures for air quality improvement could be increased. These relationships provide incentive to have an air quality level of Q with an expenditure of P. More is known about costs of air quality control than benefits; research should therefore be conducted to determine the benefits of air quality improvement.

7.3 RESEARCH AND DEVELOPMENT NEEDS

Concern with possible health and environmental effects of various air emissions from AFOs has grown with the increasing size, geographic concentration, and suburbanization of the operations in what was formerly rural, sparsely populated agricultural areas (NRC 2002). As such, an urgent need exists to develop comprehensive, cost-effective air quality management strategies for AFOs. Research is needed in such areas as characterization and measurement, abatement measures, and health and environmental effects. While many control methods have been developed, more study is needed to assess the technical and economic feasibility of applying such technologies in AFOs (USEPA 2001).

According to Miner (1995), air quality research should include effects of animal diet; manure treatment; handling (management) systems; chemical, physical, and microbiological methods; weather components; facility design; and related factors on the quality and quantity of odor-causing compounds. Characterization and measurement, especially for

TABLE 7-7. Summary of Air Quality Control Technologies for Manure Management and Treatment Systems

Technology	Description	Cost ($ per pig marketed)	Source
Chemical treatment	Chemical or biological products are added to the manure	$0.25–>$1.00	Lorimor and Edwards 1998a
Solid-liquid separation	Solids are separated from liquid slurry through settling basins or mechanical separators	$1.00–$3.00	Jacobson et al. 1998
Composting	Mechanical devices are used to provide oxygen to a compost pile, keeping it aerobic	$0.20–$0.40	Lorimor and Edwards 1998b
Anaerobic digestion	Manure is converted to methane and carbon dioxide by anaerobic bacteria under controlled conditions of temperature and pH	$1.10–$4.00	Lorimor and Edwards 1998c
Aeration	Mechanical aerators are used to provide aeration and oxidize the waste	$2.00–$4.00	Jacobson et al. 1998
Biocover (straw, chopped cornstalk)	Straws or cornstalks are blown to the surface of storage unit	$0.25–$0.40	Lorimor and Edwards 1998d
Synthetic cover	Synthetic covers are placed over manure storage units (floating or rigid structures)	$4.00	Lorimor and Edwards 1998e

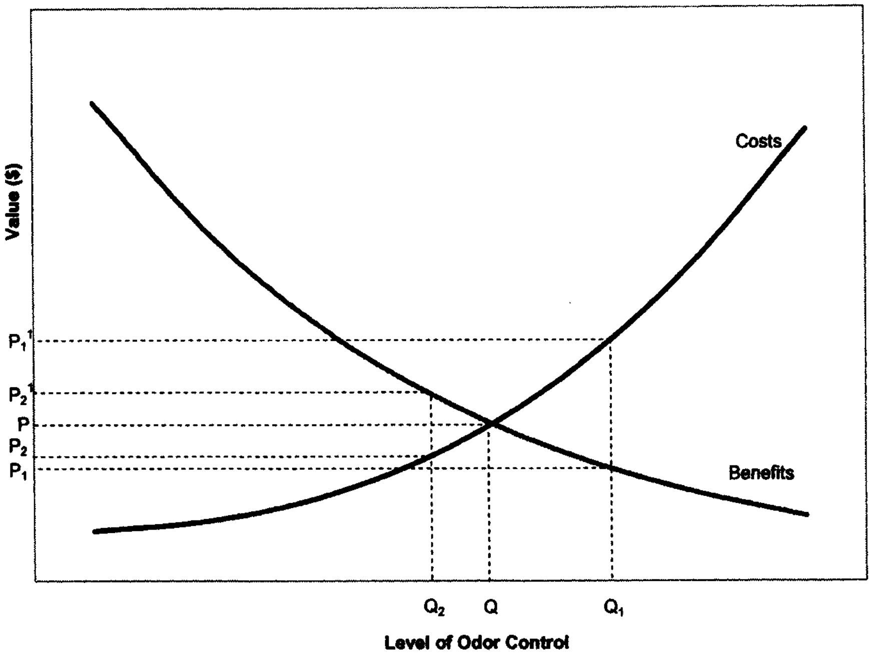


FIGURE 7-7. Comparison of Costs and Benefits from Livestock Odor Reduction (Source: Jolly and Kliebenstein 1995).

odors, is a critical research component. Mackie et al. (1998) noted that, in order to develop environmentally sound, sustainable animal production systems, researchers need to integrate research that focuses on modern analytical techniques and the latest sensory technology for measurement and evaluation of odor and pollution, together with a fundamental knowledge of animal and microbiological factors that are the basic units contributing to production of odor and pollutants. They indicated that, without a thorough understanding of what odor is, how to measure it, and where it originates, odor will be difficult to control.

Various groups and researchers have identified research programs related to air quality issues associated with AFOs. According to Miner (1995), research opportunities having potential to reduce odor complaints from swine operations include improved odor identification and measurement, better building design, improved manure management systems, and treatment options prior to land application, as listed below:

1. Odor identification and measurement:
    a. Improve electronic detection systems that offer potential to eventually replace labor-intensive, high-cost methods of olfactometry.
    b. Better define the interaction between odor production, separa-

TABLE 7-8. Recommended Research Program Related to Air Quality Regulations of Agricultural Odors

Objectives/Sub-Objectives	Recommended Support
Agricultural odors remain a complex issue with both a measurable component based on the presence of small amounts of specific gas molecules and a more subjective component based on individual sensitivity. Support for expanded research activities is needed to fill the gaps between technology development and the needs of agricultural producers and the public.	$2 M/yr
Expanded research support is required in the following areas to better identify and measure odors, determine the relationship between odorous compounds and the environment, identify human response to odors, and identify economical control methods and reduction strategies: • *Determine whether odors or specific odorants are useful measures of other contaminants that are more difficult to detect.* • *Determine relationships between biological particulate matter and odors as a function of distance from a site.*	$1 M/yr
Determine odor sources from agricultural production and the impact of design and management practices on odor release and transport. • *Develop understanding of chemistry of anaerobic impoundments and develop new methods that can reduce odors and enhance treatment. (Example: Development of inexpensive cover that is an aerobic biological reactor which oxidizes hydrogen sulfide and ammonia.)*	$1 M/yr

Improved dispersion modeling methodology including odor release, transport, and receptors.	$1 M/yr
• *Determine whether air quality beyond property lines may be improved by using structural barriers, trees, and other vegetation to adsorb odors and chemicals and potentially enhance dispersion.*	
Standardized measurement methodology, technologies, and devices for odor detection including frequency, intensity, duration, and offensiveness.	$1 M/yr
Determine the chemical and physical properties of odor including odor production processes, interaction of environmental variables, odor release pathways, interactions among odorants, and kinetics.	$1 M/yr
• *Assess potential relationships between emission constituents and their concentration levels and health symptoms of neighbors.*	
Development and implementation of economically and technologically feasible odor control and reduction strategies.	$1 M/yr
• *Develop technologies to reduce odors and emissions for housed animals, treatment, and land application systems.*	

Source: Sweeten et al. 2000.

    tion distance, climatic data, and local land use.
   - c. Develop appropriate odor indicator compounds such as long-chain VFAs or specific microbes.
2. Building design:
   - a. Improve manure removal efficiencies from surfaces.
   - b. Reduce manure volume and surface area.
   - c. Develop innovative building exhaust air treatment processes.
3. Manure management systems:
   - a. Develop systems that conserve rather than volatilize nitrogen.
   - b. Develop energy recovery schemes, including biogas production.
   - c. Implement scientifically sound programs of evaluating new products (e.g., odor control additives, and permeable lagoon covers).
4. Application of manure to cropland:
   - a. Develop short-term temporary treatment alternatives for odor reduction (e.g., aeration, chemicals, and dilution) prior to land application.

The U.S. Department of Agriculture (USDA) Agricultural Air Quality Task Force (AAQTF), established under the 1996 Farm Bill, has recommended a research agenda related to air quality regulations of agricultural odors (Table 7-8). The AAQTF recommends a program of accelerated research, education, technical training, technology transfer, and financial assistance to cope with air quality problems. According to Sweeten et al. (2000), while some work is in progress in relation to the AAQTF research agenda, much remains to be done.

7.4 CONCLUSIONS

Livestock production is a major agricultural pursuit in the United States that involves more than 450,000 individual farms. Recent trends have indicated a shift to larger and less numerous livestock operations. As production facilities have enlarged, greater confinement of animals has been adopted to minimize labor requirements. Increased size of facilities and intensive confinement has resulted in concentrated manure accumulations in small areas. The decomposition of this manure is the predominant source of air quality/odor problems in animal feeding operations.

Current air quality control techniques include judicious site selection to avoid locations near residential or other sensitive areas, inhibition of anaerobic decomposition by mechanical surface aeration, and confinement of manure in covered storage units. Many odor-control chemicals for inclusion in feed or direct application to manure have been proposed, but none has received widespread acceptance. An urgent need exists to develop comprehensive air quality management strategies for livestock operations. Multiple combinations of nutritional management and treatment options, as well as storage and disposal of animal wastes, will be required to reduce air quality problems and provide for long-term sustainable growth. A program of accelerated research, education, technical training, technology transfer, and financial assistance is necessary to cope with air quality problems.

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