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WASTEWATER TREATMENT ON SOILS OF LOW PERMEABILITY

Interim Report

by

Ronald E. Hoeppel, Patrick G. Hunt,
Thomas B. Delaney, Jr.

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FOREWORD

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The work was conducted during the period July 1972-June 1973 at the U. S. Army Engineer Waterways Experiment Station (WES), Vicksburg, Mississippi, by Mr. R. E. Hoeppel, Dr. P. G. Hunt, and PFC T. B. Delaney, Jr. The study was under the general supervision of Dr. John Harrison, Special Assistant for Environmental Studies.

Mr. Ronald DeLaune of Louisiana State University assisted in the preparation of the literature review and the redox measuring system.

Directors of WES during conduct of this study and preparation of the report were BG E. D. Peixotto, CE, and COL G. H. Hilt, CE. Technical Director was Mr. F. R. Brown.

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SUMMARY

Experiment I

A clay soil-Reed Canary grass system was used to study nitrification and denitrification conditions in overland flow treatment of wastewater and represented a small-scale simulation of a cannery wastewater disposal system. Wastewater containing 15- to 20-mg/ℓ nitrogen was applied to the 10-ft-long (3.05-m) model at a rate of approximately 2.5 in. (6.35 cm) per 5-day week over a 12-week period. This rate is equivalent to a field application of 0.014 mgd/acre (25,500 ℓ/hr/ha) over a 5-hr period.

Average removal efficiencies for nitrate N, ammonium N, and organic N over the course of experimentation were 90.4, 92.6, and 39.8 percent, respectively. Total nitrogen removal was calculated to be 81.7 percent. An amount equivalent to 75.9 percent of the applied nitrogen was incorporated into grass tissue. Since the soil contained over 0.2 percent nitrogen, much of the grass nitrogen probably originated in the soil rather than in the wastewater. Therefore the 7.0 percent of wastewater nitrogen unaccounted for was probably an underestimation of the gaseous losses of nitrogen from the systems (denitrification).

Nitrate removal efficiency appeared to decrease when the grass was cut to a height of a few centimeters and blue-green algae were allowed to form a crust over the surface of the soil. Ammonium and organic nitrogen treatment also decreased noticeably at this time, especially during periods of grass regrowth and resultant algal death. Upon regrowth of grass and the disappearance of the algal crust, the nitrate, ammonium, and TKN removal efficiencies increased. This resulted in the greatest removal of total N (92.8 percent) during the final week of experimentation.

The initial hypothesis that denitrification is increased by the combination of both aerobic and anaerobic zones within or under thick algal mats on overland flow treatment systems was rejected. It was instead hypothesized that denitrification in overland flow systems is principally a soil phenomenon, which probably functions similarly to denitrification in rice fields. In a flooded rice field, an "aerobic-anaerobic double layer" exists in the upper soil profile. This combination allows for oxygen-dependent nitrification and anaerobic-dependent denitrification to proceed simultaneously. The authors feel that this

Kentucky 31, tall Fescue (Festuca arundinacea), Perennial Ryegrass (Lolium perenne L.), and Bermuda grass (Cynodon dactylon L.). After 10 weeks of treatment, an average of 88 percent of the applied nitrogen had been removed from the wastewater after passage over the 10 ft (3.05-m) model. This value includes the relatively low treatment achieved during the first four weeks, which increased to greater than 93 percent in the following weeks. The grass removed 30 percent and the soil removed 34 percent of the applied nitrogen, while denitrification was hypothesized to be responsible for the remaining 24 percent. Results from this and a parallel study involving an unamended Susquehanna clay indicated that the addition of sludge did not significantly modify nitrogen transformations in the system.

Phosphorus removal was a surprisingly low 69 percent considering that the soil used had a pH of 5 and a high iron and aluminum content. However, the low phosphorus removal efficiency was observed in similar studies with Susquehanna clay and on the Campbell Soup Company's overland flow treatment system at Paris, Texas. The authors believe that phosphorus treatment may be the limiting factor in overland flow systems.

Potassium was very effectively removed from wastewater. Sodium was not removed effectively, but its concentration in the wastewater was relatively high, so that a significant increase in sodium occurred in the soil. In this wastewater, the concentration of sodium was uncommonly higher than the concentration of calcium and magnesium combined. If it was applied on a site for a prolonged period, there could most likely be both soil structure and high salt content problems.

Cadmium, copper, manganese, and lead were removed by greater than 90 percent from the wastewater. Nickel and zinc had removals of 83 and 75 percent, respectively. The low treatment efficiency of zinc probably resulted from its high concentration in the sludge, but the reason for poorer removal of nickel is not clear. The higher concentration of zinc in the sludge was reflected in the higher concentration of this element in the harvested grass. The rapid removal of cadmium by the soil system was also reflected by a significantly higher concentration of cadmium in the grass harvested from the first 2.5 ft (0.76 m) of the model. No significant increases in heavy metals occurred in the soil during this short study, but it should be recognized that over a long period of time significant amounts of these elements could be added from industrial wastewaters. These results indicated that short term incorporation of 6 percent sewage sludge into the upper inch of soil did not significantly modify wastewater treatment in the overland flow system.

concept is probably the key to much of the nitrogen lost on overland flow systems, since ammonium normally must go through the nitrification-denitrification cycle to be lost as nitrogen gas. It is possible that some algal crusts may physically or biochemically hinder the diffusion of nitrate and ammonium between aerobic surface and anaerobic subsurface zones, or may prevent the establishment of aerobic zones in the soil. This would decrease the nitrogen treatment efficiency.

Following the double layer concept and the assumption that both nitrification and denitrification would occur more rapidly in the soil than in the overflowing wastewater or in a biological surface crust, an interesting hypothesis can be made to explain the requirement for a rest period in an overland flow system. If the system is subjected to wastewater with a high BOD, the organic material could consume oxygen in the liquid film before it reached the soil surface. The result would be a rather low rate of nitrification and consequently a low rate of denitrification. A rest period would then be required to reestablish aerobic conditions at the soil surface. If the preceding hypotheses are valid, secondary treatment that pushed nitrogen to the nitrate form would make advanced treatment by overland flow more efficient for nitrogen. Under these conditions the presence of the aerobic layer for nitrification would not be required; only the anaerobic conditions for denitrification would be necessary. <u>However, it must be emphasized that these are only hypotheses.</u> It was proven in this study that at a depth of 15 mm in the clay soils reduced conditions existed throughout the wetting cycle and only after a weekend of drying were oxidized conditions approached. It was shown, however, that during weekend drying, the surface 3- to 5-mm layer changed from reduced to oxidized conditions, in which nitrate would remain stable. The presence of an aerobic layer during the wetting cycle was not proven, although its existence seems likely.

Algal inhibition of denitrification by photosynthetic production of oxygen was examined by running the model in darkness. No increase in nitrogen removal was observed. In order to determine if the algae were inhibiting nitrate reduction by biochemical mechanisms, anaerobic wastewater was incubated for several days with and without an algal cell extract. The extract did not decrease nitrate loss, but further analyses concerning short-term inhibition seem warranted.

The greatest removal of nitrate and ammonium nitrogen from the wastewater was displayed in the first downslope foot of the model. During the first four weeks, nearly half of the nitrate nitrogen was removed in this distance. Although the treatment efficiency decreased after this time, the initial effect of distance on removal rate remained apparent through the ninth week. Treatment efficiency was also found to vary inversely with flow rate.

<u>Experiment II</u>

This experiment was conducted to examine the effects of adding sewage sludge to the upper few centimeters of an overland flow system. The soil used was a previously untreated Susquehanna clay, and the grass cover was a mixture of Reed canary grass (<u>Phalaris</u> <u>arundinacea</u> L.)

WASTEWATER TREATMENT ON SOILS OF LOW PERMEABILITY

INTRODUCTION

It has been recognized for some time that secondary treatment of wastewater is not sufficient to prevent water pollution and the resulting eutrophication. However, only since the 1972 Clean Water Act have requirements for advanced treatment of wastewater been set. Advanced wastewater treatment may be accomplished using a variety of treatment processes. These may be physical, chemical, or biological in nature. More often than not, advanced waste treatment systems do not employ these processes singly but rather make use of compatible combinations of them. Additionally, the number of possible interactions within each process can be quite large. The fact that no single system has gained strong popularity among designers of municipal systems should be judged as sufficient evidence that none is obviously superior, and additional research, as well as operational evaluation, seems warranted.

This study has been limited to land treatment as a means of achieving advanced wastewater purification. Land treatment has the advantage of incorporating the recycling concept directly into its treatment mode, resulting in replacement rather than depletion of natural resources. Also, some form of control over ecologically damaging components is retained. However, these same components can be dispersed into the environment if proper care in selection and management of land treatment sites is not exercised. Normally, land treatment is thought of as the application of a moderate amount of wastewater to soils, which allows for vertical penetration at a rate sufficient to prevent ponding but slow enough to accomplish treatment. Such a system has been used successfully for 10 years by Pennsylvania State University and is probably the most studied in the United States. The Environmental Protection Agency (EPA) has recently funded the construction of a large municipal and industrial wastewater treatment facility at Muskegon, Michigan, which includes tertiary land treatment. Michigan State University is also in the process of constructing a related lagoon-land treatment system. It is hoped that the operation of these facilities will answer questions concerning the use of land treatment in large municipal areas. Studies being conducted at Phoenix, Arizona, and Tallahassee, Florida, are addressing the problems of land treatment in fairly porous soils.

Wastewater treatment on sloped and impervious soils, commonly referred to as the "overland flow" mode, has generally been considered to have little applicability to the treatment of municipal wastewaters. Only one system that utilizes low-permeability soils has been studied in any detail, that of the Campbell Soup Company's plant at Paris, Texas. This system has been operating since 1964 and has been shown by an EPA study (36) to have good nutrient removal capability. However, the system treats cannery rather than municipal wastewater. Much of the nutrient load in this system is in an organic form and would initially be bound with the organic fraction on or in soils or grasses. Therefore, the level of treatment achieved has often been dismissed as a mere physical filtration phenomenon, or rather an efficient retention of nutrients in the organic phase. Another enigma of the runoff-type system is the frequent occurrence of a significant amount of unaccounted for nitrogen. This has been attributed to the gaseous loss of nitrogen, termed denitrification. However, no quantitative and very little direct evidence of this fact has been produced even though denitrification in a similar system, the rice paddy, is well established (63). In both rice paddy and overland flow systems, the soil pores are filled with water, and oxygen must diffuse through a water layer of several millimeters to several centimeters thick. Since the diffusion coefficient for oxygen is approximately ten thousand times less in water than in air, the rate of movement of oxygen into the flooded soils is quite low. Consequently, anaerobic sites and concomitant denitrification quickly develop even in the upper few centimeters of flooded soil. Thus, considering the good performance of the Paris, Texas, overland flow system, the similarity of such a system to a rice paddy system, and the large amounts of underutilized low-permeability soils, it seems promising to pursue the concept of overland flow treatment for municipal wastewaters. The research presented in this report deals with wastewater treatment by the overland flow mode, with particular emphasis on nitrification and denitrification.

LITERATURE REVIEW

Land Treatment of Wastewater

Various methods of treating wastewater on land have been practiced for many years. Presently, there are several hundred municipalities in the United States utilizing land treatment for part or all of their wastewater treatment needs (54). The American Public Works Association has recently completed a survey of facilities using land application of wastewater (5); readers desiring information regarding details used in these systems should consult the referenced publication. For the purposes of this review, however, only a few of the more publicized and studied systems will be discussed.

The Penn State Land Treatment System is used to treat part of the secondarily treated wastewater for State College, Pennsylvania. This spray irrigation system has been in operation for 10 years (46). It handles a hydrological load of a few inches of applied wastewater at the treatment site per week, an amount typical of agricultural irrigation. With this type of land treatment, wastewater-soil particle contact is at a maximum. The opportunity for ionic exchange, fixation, adsorption, and precipitation is great, and microbial transformations and plant uptake are active components. Rates of nitrogen removal by corn of 0.0078 to 0.0168 kg/m^2 (65 to 127 percent of the nitrogen applied in wastewater) have been reported from the Penn State system (32). This system functions well in terms of both water-quality improvement and crop growth, but like most land treatment systems, it allows vertical penetration into the groundwater. Therefore, its performance and the hydrogeologic interactions must be constantly surveyed.

A system has been designed to avoid the possibility of groundwater contamination and hydrogeologic problems while utilizing the treating capacity of the upper few feet of soil profiles. In this system, which is presently under construction at Muskegon, Michigan, underdrains will be used to collect the treated water (44). The water will then be returned to the municipality for reuse or discharged into a waterway or lake. There is, however, the possibility of drains becoming clogged if reduced conditions are prevalent in the soil profile above the underdrains (24,58). It is hoped that this system will provide much needed information on the practicability of transporting large volumes of water from one watershed to another for treatment and returning the same to the original watershed for reuse or discharge.

Another type of land treatment, rapid infiltration, allows several feet of wastewater per day to be passed into porous and reasonably deep soil profiles. Nitrification, denitrification, and soil exchange are considered quite important in this system. The most studied system of this type is that at the Flushing Meadows Site near Phoenix, Arizona. The greatest weakness of this type of treatment is that, when large volumes of wastewater are passed through the rapid infiltration system, plant and microbial uptake of nutrients is rather inconsequential. When wastewater from the system is allowed to pass into the groundwater, simple dilution is probably a major factor in reducing excessive concentrations of water soluble constitituents. If, however, the treated wastewater can be kept separate from groundwater and reused as irrigation water, this system has considerable potential, primarily as a physical filter. In addition, the system functions adequately when the application rates are kept low enough so that plant as well as soil and microbial removal or conversion of nutrients is significant. Such a system is being used at Tallahassee, Florida, and major groundwater pollution has not been observed. Studies are continuing to establish what effect, if any, the wastewater application rate might have on the groundwater quality.

As stated previously, land treatment of wastewater on relatively impervious soils, which allow very little penetration of water, is commonly referred to as the overland flow treatment system. The Campbell Soup Company's plant at Paris, Texas, is the best known and most studied system of this type. This system gives high removal of biochemical oxygen demand (BOD) and nitrogen and moderate removal of phosphorus (36). However, when overland flow systems are considered for treatment of municipal wastewater, the question often arises as to treatment mechanisms and their applicability to such a wastewater. Plant uptake and denitrification have been proposed but not proven as the major mechanisms of treatment in the Paris, Texas, system (29,36). However, the possibility of surface denitrification does not seem so remote when the similarity of this system to the rice paddy system is considered. The importance and magnitude of denitrification in rice cultivation are well established, and the concept of denitrification under continually and intermittently flooded soils will be pursued in more detail later in this report.

Although land treatment systems differ greatly in some aspects, they have many underlying treatment mechanisms that are similar, including biochemical immobilization and denitrification, chemical exchange, and physical filtering by the soil as well as plant uptake. The proper management of a land treatment system in accordance with the properties of the site is of paramount importance. The remainder of this literature review will address land treatment principles with emphasis toward those sites that should be operated as overland flow systems.

Reduced Soil Conditions

When the gas phase of soil pores is readily supplied with diffusible oxygen, aerobic, oxidized conditions usually exist. The oxidative-reductive condition is normally expressed in terms of the oxidation-reduction potential (Eh) at a pH of 7 and a temperature of 25 C. According to Patrick and Mahapatra (49), oxidized soils have Eh values of +400 mv or greater, while soils with Eh values of +100 to +400 mv are moderately reduced, resulting in oxygen and nitrate instability. Consequently, oxygen and nitrate tend to buffer a drop of Eh when soils are flooded, but their effects are of a few hours and days, respectively. The rate of decrease in Eh is highly variable, and is related to the organic content, temperature, soil texture, and time between floodings of the soil (63).

Air must diffuse through water films to the sites of aerobic respiration when the soil is flooded or saturated. Since the diffusion coefficient for oxygen through water is approximately ten thousand times less than that in air, the potential for aerobic metabolism at more than a few millimeters into the soil is quite small (25). The actual depth to which oxygen can penetrate is dependent upon such things as the depth of the flooded water, the rate of metabolism in the system, and the length or tortuosity of the diffusion path. Although an oxidized layer may only be a few millimeters thick, it has a profound effect on the function of flooded soils, for in this layer aerobic activities, particularly nitrification, can occur (50). Under flooded conditions without this layer, nitrate would not be formed, and denitrification (which is dependent upon the formation of nitrate) would not occur. Defining denitrification as an asset or a detriment depends on whether wastewater is being treated or a wetland farming operation is being conducted.

Although oxygen would be rapidly depleted within a few millimeters of soil depth, its presence would tend to stabilize reduced soil conditions. The same is true of nitrate at these depths, but abnormally high concentrations would be required for nitrate to be available longer than a few days (62). More effective buffering of the decline toward reductive conditions is obtained from the Mn^{+3}, Fe^{+3}, and oxidized organic fraction of the soil (51). It is only under extremely reduced conditions that sulfate begins to be reduced, whereby oxygen is removed and hydrogen is added to the compound. As a result of the intermittent nature of flooding in the overland flow treatment of wastewater, extremely reduced conditions should occur only at depths of several centimeters in the soil profile.

Mineralization and Immobilization

Mineralization as discussed in this review is defined by Alexander (1) as "the conversion of organic complexes of an element to the inorganic state." The process may involve the processes of several microorganisms, or it may be a simple cleavage of an element or compound from an organic molecule. This review will consider the reverse of mineralization to be immobilization, although in the broadest sense incorporation into soil humus or plant material can be considered immobilization.

If a highly carbonaceous material is added to an aerated soil, immobilization of nitrogen and other elements is likely to occur. This is because under aerobic conditions, 30 to 40 percent of an applied carbon source is likely to be converted to cellular material (1,2). Similarly, other nutrients (such as nitrogen, phosphorus, and sulfur) will tend to be taken up as these are needed to form protein and other necessary organic compounds in an expanding microbial population. This biological immobilization is due to the high energy yielding reaction in electron transport with oxygen as the terminal electron acceptor. Concomitantly, the reduced carbonaceous materials tend to be extensively oxidized to CO_2. The result is a decrease in soil organic carbon while nitrogen continues to remain immobilized in soil organic material. Under reduced conditions, the energy yield is not as great, and the conversion (5 to 10 percent) to cellular material is much lower. Consequently, the demands on nitrogen and other nutrients are not as high, and more nitrogen is likely to be mineralized and remain in the inorganic form. In addition, the extent of organic matter oxidation is not as great under reduced soil conditions. After nitrate has been consumed and the terminal electron acceptors are organic compounds, degradation tends to stop at two- to five-carbon compounds, and in the absence of oxygen aromatic molecules are not readily cleaved. Thus, it is possible under reduced conditions to obtain an increase in both organic matter and ammonium.

In a soil profile that will allow significant leaching of nutrients, immobilization and slow mineralization are extremely important in maintaining nutrients in the root zone where plant uptake is an effective means of removal. However, in an overland flow system, the nutrients normally do not leach past the root zone, and immobilization and mineralization are not as important in the prevention of groundwater contamination. Their importance is in moving nitrogen in and out of the soil organic matter and microbial cells and in making ammonium available for nitrification during aerobic periods. Something worth considering is that a nitrogen increase of only 0.01 percent in the upper 15 cm of the soil profile is equivalent to an increase of 224 kg/ha.

It might be hypothesized that most of the nitrogen in cannery wastewater is in suspended organic matter and would thus be quite easily removed by filtration, while the nitrogen in secondarily treated municipal wastewater would be contained in the dissolved fraction and would not be easily removed by physical means. In the latter system, biological immobilization would appear to be the dominant process in soils. With cannery waste, immobilization would also tend to predominate due to the increased load of organic materials, but slight changes in microbial populations, deficits of certain nutrients, or other environmental factors could result in a latent release and subsequent mineralization of the profuse supply of organic nitrogen. The mineralized nitrogen, being primarily ammonium, would then be biologically oxidized (nitrified) to nitrate during periods of soil drainage and drying. Such an hypothesis might explain the need for rest periods in overland flow systems and the initial flushing of nitrate from the system after a rest period (36). The authors feel that the requirement of a rest period and the functioning of an overland flow site are very complex, probably involving such factors as oscillation between reduced and oxidized conditions, site slope and uniformity, buildup of toxic compounds, microbial sealing or coating of the soil surface, and possibility of nitrogen fixation (69,70).

Nitrification

Nitrification, the oxidation of ammonium to nitrate, is a reaction of great agricultural and ecological importance. It is nitrate nitrogen that is readily taken up by most crop plants as a nitrogen source, but it is also nitrate nitrogen that readily moves through the soil profile into groundwater and reservoirs causing public health and eutrophication problems. Nitrification is normally thought to occur through bacterial processes, by the conversion of ammonium to nitrite by *Nitrosomonas* and the subsequent conversion of nitrite to nitrate by *Nitrobacter* (1). However, it should be borne in mind that in the soil other microorganisms are involved in nitrification, and their activity may at times be quite significant (1,2).

Nitrification is normally a very rapid process. Ammonium is often converted to nitrate in the few minutes required for water to flow through a soil column only a few inches deep. Nitrosomonas and Nitrobacter, which derive energy solely from ammonium oxidations, are not as efficient in utilizing this energy as are microbes that obtain energy from the oxidation of various organic molecules. As a result, nitrifying organisms tend to oxidize a relatively large amount of ammonium per unit cell mass.

The authors tend to agree with Alexander (3) who feels that the relationship of pH to nitrification has been sufficiently stressed. However, it must be emphasized that significant nitrification does not occur in soils with pH values lower than 5, while ammonium and nitrite oxidations are thought to be adversely affected by ammonia and high pH values, respectively (57). The sensitivity of Nitrobacter to ammonium ions may result from the buildup of nitrite, which may subsequently have adverse effects on biological components (2). One of the more interesting studies on pH-nitrification phenomena was conducted by Morrill and Dawson (42). They noted that nitrite accumulated at pH values higher than 7.3, with further oxidation to nitrate occurring at a slow rate. On the other hand, low pH values (4.5 to 5.9) resulted in no nitrite buildup but also little or no nitrate accumulation. Neutral pH levels prompted the greatest rate of nitrification.

It is well to remember when thinking of overland flow treatment of wastewater that flooded soils tend to approach a neutral pH level and are therefore in the proper pH range for active nitrification. This factor is quite important since there is intense competition for the small amounts of oxygen present under flooded conditions.

Since nitrification is an aerobic process that is rather rapid under the proper conditions, neither ammonium nor nitrite is found in significant concentrations in aerated, neutral, and warm soils. Under such conditions, mineralization is the limiting step of nitrification. Soil moisture levels that do not limit oxygen diffusion are better than dry soil conditions. Moisture tensions near field capacity (0.1 to 0.3

bar) are normally found to promote the highest rates of nitrification (29).

The effects of temperature and organic matter on nitrification appear to be predictable. If the temperature is lowered or raised significantly from the range of 25 to 35 C, the rate of nitrification declines. Organic matter seems to have little effect on the rate of nitrification other than its effect on the amount of oxygen present and the rate of immobilization and mineralization.

Nitrification has been shown to occur in continually and intermittently flooded soils by Patrick (47,49). He postulates that a thin 3- to 10-mm layer of oxidized soil exists above the reduced zone of a flooded soil and that nitrification occurs in this layer. The thickness of such a layer depends upon the factors that establish the diffusion rate of oxygen, such as soil texture, depth of floodwater, and organic content. The widespread occurrence and acceptance of this phenomenon are evident in rice-cultivated areas. In rice paddies, surface-applied ammonium is rapidly nitrified to nitrate and subject to loss by denitrification in the underlying soil. Therefore, surface application of ammonium is recognized as a poor cultural practice in rice farming. These same properties, however, can be used to an advantage in overland flow treatment of wastewater by enabling the system to achieve or exceed required nitrogen removal rates.

Denitrification

Volumes have been written on the intricate biochemistry of denitrification, and this report attempts only to summarize the practical points of these studies. Readers that are interested in this aspect are referred to U. S. Army Engineer Waterways Experiment Station (WES) Miscellaneous Paper Y-73-1 and its cited review articles. In this section, however, an attempt will be made to circumvent the confusion surrounding denitrification and to emphasize the predominant processes as well as the ecological and engineering importance of these processes.

Denitrification is the process whereby nitrate and nitrite are converted to more reduced gaseous forms of nitrogen (15). The most common form is elemental nitrogen, but nitrogen oxides can be formed and more readily persist under acidic conditions. Denitrification can be either biological or chemical; however, biological denitrification is the more common and significant process. Biological denitrification is a rather straightforward anaerobic process in which nitrate serves the function that oxygen does under aerobic conditions, the terminal acceptor of electrons in cell respiration. The terminal acceptor of electrons can be thought of as the final step in energy formation by an organism. The acceptor makes the formation of energy continuous by rejuvenating the last component in an energy formation chain. This is done by means of oxidizing it at the sacrifice of nitrate reduction, whereby nitrate accepts the transferred electrons. In turn, each preceding component can also be reoxidized (electrons removed) through a chainlike transfer system. The oxidized compounds in this chain are again reduced by ingestion of other reduced materials in the environment. The same process takes place in aerobic oxidation with oxygen serving as the rejuvenator of the energy formation chain, the <u>terminal electron acceptor.</u> The end products of aerobic respiration are CO_2, H_2O, and N_2.

Most denitrifying microorganisms use oxygen under aerobic conditions and nitrate under anaerobic conditions and are thus referred to as facultative anaerobes. There is some evidence of the processes occurring simultaneously in the same organism at very low oxygen concentrations (35). However, denitrification should be considered a strictly anaerobic process, with the realization that in a soil system strictly aerobic and strictly anaerobic sites can exist within millimeters of each other.

Another confusing point about denitrification is whether or not denitrification is controlled by oxygen concentration or oxidation-reduction (redox) potential (15). Here again, the point is of little consequence since the measure of one is not independent of the other. The authors feel that redox measurements are the better indicator of denitrifying conditions because they are more reliable under reduced conditions. The exact redox potential at which nitrate is unstable is not agreed upon because of variability in the measurement. However, below a potential of +220 mv, adjusted for a pH of 7 and corrected to the hydrogen ion as a reference, nitrate can be considered unstable. This is a significant point when the quickness with which a potential of +220 mv can be reached by flooding or rapid consumption of oxygen is considered. Thus, denitrification occurs before what might be thought of as intense reducing conditions.

Denitrification is favored by both extreme physiological temperatures and neutral pH levels (15). However, normal field temperatures of 20 to 30 C do not suppress denitrification greatly, and flooded soils tend to equilibrate at a near neutral pH level (53). As a consequence, significant denitrification under flooded conditions is normally not limited by the pH or temperature.

Phosphorus Fixation and Availability

Well-drained soils have been shown to differ in their capacity to fix phosphorus depending on the quantity of iron and aluminum oxide present (20,67,10), the kinds and amounts of clay minerals (59,13,39), and the soil pH values (20).

Iron and aluminum compounds are largely responsible for the fixation of phosphorus, especially under acidic well-drained soil conditions (20).

Bear and Toth (10) noted that the amounts of soluble iron and aluminum are very low when compared with the amount of phosphorus a soil is capable of fixing. Coleman (20) concluded that iron and aluminum do not have to be in solution in order to fix phosphates. It was assumed that iron and aluminum are present in soils as film coatings of oxides and hydroxides.

The behavior of phosphorus in flooded soils is remarkably different from its behavior in well-drained aerated soils (49). Reasons for the wide difference in phosphate response observed under aerobic and anaerobic conditions have not been clearly defined, but it is generally understood that the chemical nature of phosphate compounds and their reduction, hydrolysis, fixation, and refixation in the soil are responsible for these differences. This difference in behavior is of great practical significance in the phosphorus nutrition of rice (49).

The phosphorus in soils is inorganic or organic. The inorganic forms of phosphorus have been classified by Chang and Jackson (17) into four subgroups: Calcium phosphate, aluminum phosphate, iron phosphate, and reductant-soluble iron phosphate. Both types of iron phosphate are of special importance in flooded soils used for rice growing.

According to Chang and Jackson (18), calcium and aluminum phosphates are more likely to be formed than iron phosphate immediately after phosphate fertilization of rice fields, with movement into the iron phosphate fraction taking place as time passes. This is to be expected in view of the lower solubility product of iron phosphate (17). Movement into the reductant-soluble iron phosphate fixation is believed to be a slow process related to soil maturity (18). The formation of reductant-soluble iron phosphate is apparently unhampered by soil cultural practices (18). This soil phosphate fraction may be of significant importance as a phosphorus source for plants grown on waterlogged soils even though it is not considered so for upland crops.

Considerable research has been conducted in flooded soils to evaluate the crop response to the different inorganic phosphate compounds mentioned above (41,22,56,48,28,6). It is fairly well established that flooding a soil results in an increased solubility of phosphorus (56). Iron phosphates have generally been found to be more soluble than aluminum phosphate or calcium phosphate under flooded conditions.

Fujiwara (22) explained that the better response of rice to ferric phosphate is due to the occurrence of hydrolysis under flooded conditions and the resultant increase in available phosphorus. Shapiro (56) suggested that, since both aluminum and iron undergo hydrolysis, the superiority of the iron phosphate must be attributed to reduction as well as hydrolysis. Patrick (48) reported a marked increase in the extractable phosphorus with the lowering of the redox potential below +200 mv (pH of 5.7). At +200 mv the ferric ion also begins to be reduced to the ferrous form. This tends to confirm the belief that the increase in phosphate results from the conversion of ferric phosphate to the more soluble ferrous phosphate.

Valencia (64) found that immediately after flooding there was an increase in phosphorus availability, apparently due to hydrolysis of aluminum phosphates and reduction of ferric phosphates, but after a long incubation period less phosphorus was available. This was explained as refixation of the available phosphorus.

Significant changes in the solubility patterns of soil phosphorus occur when a soil is flooded, especially in the presence of a good supply of easily decomposable organic matter. The effects of organic matter on phosphate availability have been investigated by Bass and Sieling (9). Generally, organic matter increases both the solubility and availability of soil phosphorus to plants. A number of mechanisms have been postulated to explain the increased availability of soil phosphorus by organic matter. Under flooded conditions, it has been suggested that organic matter affects available phosphorus through reduction and chelation phenomena (56). It should be noted, however, that a reduction in the amount of available phosphorus can occur through the process of assimilation, that is, by bacteria decomposing the organic matter.

As pointed out earlier, the soil pH can have a remarkable effect on the availability of phosphorus. However, under flooded conditions, reduction of phosphate compounds overcomes the pH factor in determining the solubility of soil phosphorus. Even so, Aoki (6) found that the solubility of phosphorus was minimal at pH values between 5 and 7. At pH values below 5 and above 7, the quantities of soluble phosphorus were much larger under flooded soil conditions.

The solubility-pH curves for flooded and nonflooded soils resemble those of precipitated ferrous and ferric phosphates, respectively (41). This fact led Mitsui (41) to postulate that phosphate availabilty under flooded conditions was governed mainly by the solubility of iron phosphate. The high solubility of phosphate in the alkaline pH range was presumed to be due to the hydrolysis of this compound.

Heavy Metals

Both manganese oxides and ferric iron compounds that predominate in well-drained soils are reduced to the more soluble manganous and ferrous forms when a soil is flooded (51). The kinetics of the reduction of manganese and iron are similar in nature, the only difference being one of degree, manganese being more easily reduced than iron (51). Turner and Patrick (62) have reported that manganese becomes reduced at an Eh of +400 mv and is essentially completely reduced at an Eh of +200 mv. Iron, on the other hand, becomes reduced when the Eh falls below +200 mv (48). Patrick (48) attributed this increase in reduced iron (ferrous iron) to the reduction of insoluble ferric compounds that were unstable at this reducing potential.

The conditions that favor the reduction of manganese and iron in flooded soils have received much study (51,23,60,1,40,37,45). These studies have shown that the reduction of manganese and iron is favored by (a) the absence of substances at a higher level of oxidation, such as nitrate (57), (b) the presence of readily decomposable organic matter (51,23,40,37,45), and (c) a good supply of active iron (51). There have been numerous mechanisms hypothesized that might govern the reduction of manganese and iron in flooded soils. The general consensus of the numerous reports indicates that manganic and ferric compounds are reduced to the more soluble manganous and ferrous forms either (a) by serving as biological electron acceptors (45,60,1) or (b) by being reduced chemically by organic compounds during the anaerobic decomposition of organic matter (1,40,60,23,51). According to Mann and Quastel (40) and Alexander (1), several factors may be involved in these mechanisms. The solubilization of unavailable manganese and iron oxides can be accomplished by an increase in acidity accompanying fermentation that favors the mobilization of manganese and iron; the depletion of oxygen as a consequence of microbial metabolism tends to lower the Eh and lead to the reduction of manganic and ferric compounds; fermentation products (reducing organic substances) react directly with the oxidized forms of manganese and iron and shift the equilibrium from oxidized forms to the reduced manganous and ferrous forms; and finally, electron transport results, with the iron and manganese functioning as an electron acceptor in cell respiration in a manner analogous to the reduction of nitrate by denitrifying bacteria.

According to Leeper (37), microbial reduction of manganese can take place at any pH value, if the oxygen tension is low, when the anaerobic bacteria use the higher oxides as a source of oxygen. Reduction of the higher oxides in flooded soils takes place when the biological oxidation of organic matter proceeds so rapidly that air cannot supply oxygen in adequate amounts. When this occurs, reduction of the higher oxides of manganese takes place. Oxygen is thus supplied with a subsequent increase in the available manganese. Redman and Patrick (53) reported a sixfold average increase in extractable manganese after 30 days submergence.

Manganese and iron reactions in reduced environments are very closely related (34,11). In solutions with low oxidation-reduction potentials, manganese and iron form carbonates, sulfides, and silicates that are fairly insoluble in neutral or basic solutions. Berner (11) stated that once hydrogen sulfide is formed in flooded sediments, it reacts with various iron-containing minerals to form insoluble iron sulfate. Krauskopf (34) found that manganese sulfide was readily oxidized under aerated conditions to give insoluble oxides of higher valence. Oxidation of manganous compounds requires higher potentials than does oxidation of ferrous compounds, and hence manganese sulfide is more soluble than iron sulfide. Connell and Patrick (21) reported that reduced manganese was found to be less efficient than reduced iron in precipitating hydrogen sulfide. The fact that ferrous sulfide is more insoluble than manganous sulfide indicates that iron is a more effective sulfide-precipitating agent than manganese.

Very little research has been done on the availability and solubility of zinc, copper, cadmium, nickel, and lead under anaerobic conditions. Jenne (31) suggests that the role of hydrous oxides of manganese and iron is important as a control on heavy-metal solubility through sorption and desorption reactions with the heavy metals. Jenne (31) indicated that the principal factors affecting the availability of hydrous-oxide-occluded heavy metals are oxidation-reduction potentials, pH level, concentration of the metal of interest, concentration of competing metals, and concentration of other ions capable of forming inorganic complexes and organic chelates. Jenne (31) stated that the most significant factors probably are the oxidation-reduction potentials and pH level.

Geologists have demonstrated that heavy metals readily form very insoluble sulfide salts when exposed to anaerobic conditions in which hydrogen sulfide is present (34).

Two types of soil systems for overland flow treatment of wastewater were investigated during this study. One soil was from an 8-year-old commercial cannery wastewater treatment site; the other was from an untreated natural site in a national forest that was low in indigenous soil organic matter. Consequently, this latter system was amended with sludge in order to increase its organic matter content. Thus, both experimental soils represented soil systems that had more organic matter and biological activity than an average heavy clay soil. Overland flow on previously untreated, low-organic clay soil was investigated in an independent but parallel study (16).

Experiment I

Model Construction and Location

The experimental model consisted of a wooden box that was 3.05 m (10 ft) long by 1.52 m (5 ft) wide by 19 cm (7.5 in.) deep. The box was constructed at a 0.91-m (3-ft) height on a permanent 2 percent slope. Bottom sampling ports were inserted at distances of 0.76, 1.52, 2.29, and 3.05 m from the upslope end, with four ports being spaced 25.4 and 50.8 cm from each edge at each unit length. At the downslope end, four ports were similarly placed across the box at 7.6 and 15.2 cm above the bottom. The upper set coincided with the soil surface within the box, and the lower set allowed the exit of soil water from intermediate depths.

The model was heavily coated with an epoxy paint, Cono-glaze,[a] to prevent interaction between the construction materials and the soil system and to allow sealing of the Plexiglas nipples and collection trays. All ports were fitted with 0.79-cm I.D. clear Plexiglas fittings. In addition, ports at the 7.6- and 15.2-cm heights of the 3.05-m downslope end of the box were fitted with 4 Plexiglas collection trays,

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[a] Manufactured by Con/Chem, Inc., 15524 S. Broadway, Gardena, California.

2.54 cm wide and 38.1 cm long, which were sloped to the bottom of each port. Tygon tubing led to various sized narrow-mouth polyethylene bottles from the collection ports. The bottles allowed for the collection of runoff and subflow liquid and for monitoring of flow rates and total volumes. The remaining bottom ports were filled and covered with fiberglass cloth that acted as a wick for trapping subsurface liquid and prevented soil from sealing the outlet fittings. The ports were then sealed from underneath with rubber stoppers into which a needle could be inserted through a septum for withdrawal of samples.

The model was located in a fiberglass-covered Growmore greenhouse, measuring 9.75 by 6.70 m. The greenhouse was electrically heated and was cooled through evaporative pads on the north end, contributing to a temperature range of 4 to 35 C during the experiment.

Soil Collection and Preparation

Soil and grass samples were obtained on 1 February 1973 at the spray irrigation overland flow site of a commercial cannery in northeastern Texas (see Figures 1-3). The sampled area had been used repeatedly since the latter part of 1964 as a disposal area for the cannery wastewater and was being irrigated at the time of collection. The dominant vegetation at the site was Reed canary grass (*Phalaris arundinacea* L.), which at the time was in winter dormancy. The soil was classified as a gray podzol with a regolith composed of calcareous gray clay and sandstone of the Austin Formation.

The soil and grass were kept intact after sod clumps of approximately 15 cm on a side were removed by shovel and wrapped in polyethylene sheets. The material was transported by truck to the experimental site at WES, and it was placed in the experimental model within a 96-hour period.

The collected sod was shaped to the desired 15-cm thickness by inserting each clump in a 15-cm-deep wooden box and removing excess soil. The sides of the sod clumps were similarly squared so that a tight fit was achieved after insertion of each into the model. Any remaining cavities between adjacent sod were filled by hand to prevent excessive subterraneous flow of applied liquid. When the soil was in

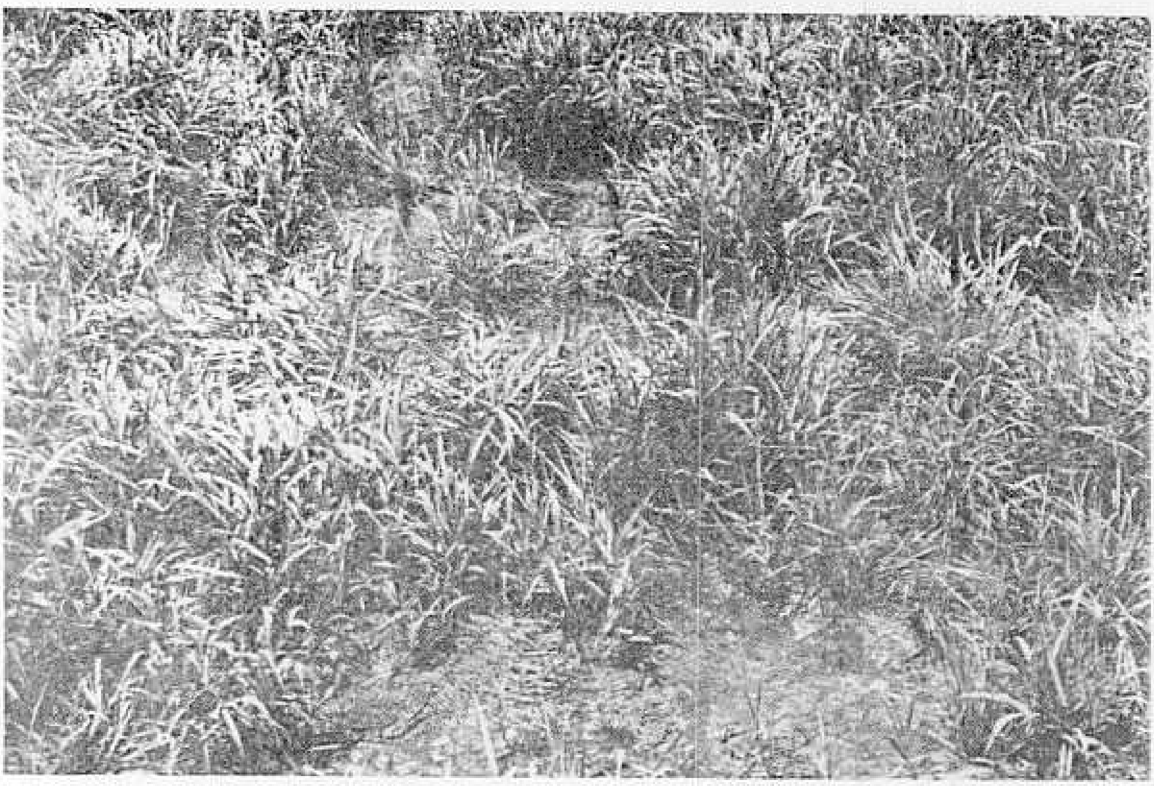

Figure 1. Close-up of an overland flow treatment site.

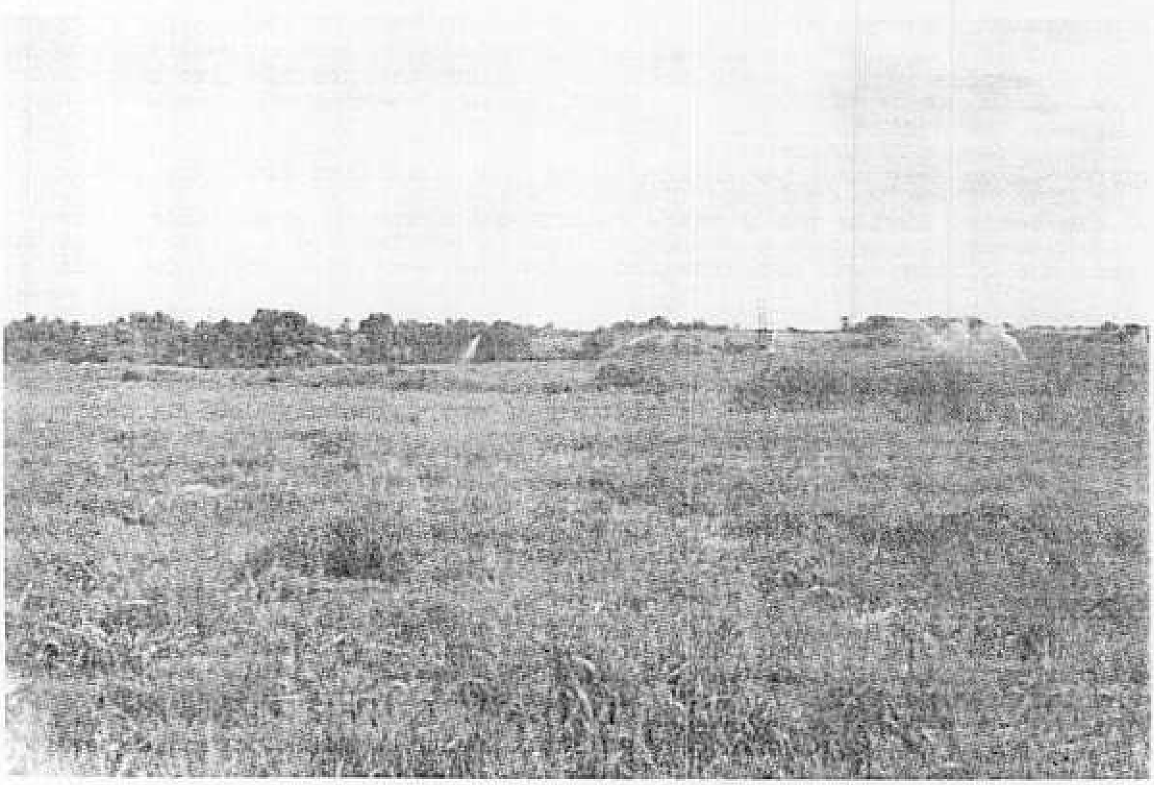

Figure 2. General view of an overland flow treatment site.

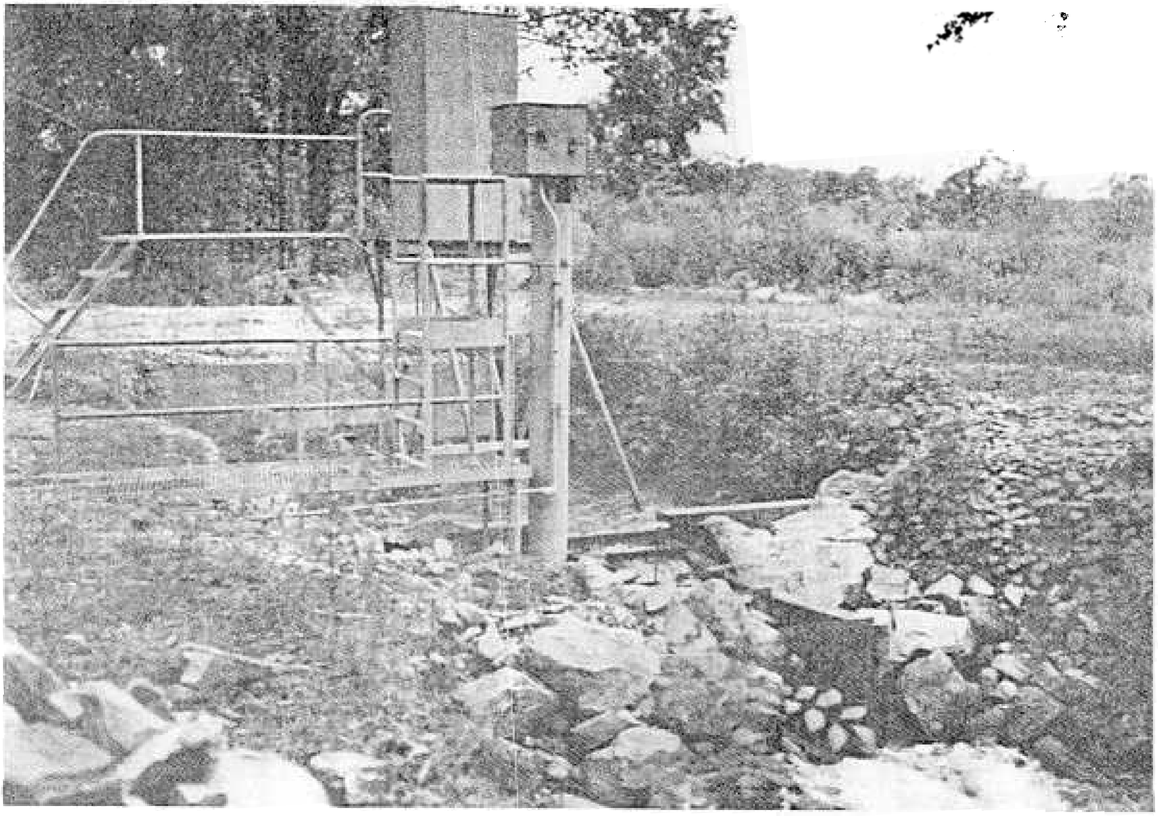

Figure 3. Water-monitoring site for treated water leaving the overland flow treatment site.

place, samples were randomly taken throughout the profile for later analyses. A central divider, consisting of an epoxy-paint-coated steel strip 5.1 cm deep and 3.05 m long, was then installed down the length of the box to divide the surface runoff and minimize channeling. However, the divider was not extended below the 4-cm soil depth for fear of obtaining an edge effect. The behavior of this soil during the experiment indicated that this fear was probably unfounded.

For the first 3 weeks, a mixture of approximately 0.05 percent sucrose-water was applied to the soil system in order to stimulate the grass cover and the microflora. By the end of this period, the grass had responded well and was giving uniform coverage of the model.

Wastewater Collection and Preparation

Secondarily treated wastewater was obtained from a small extended aeration treatment plant operated by the Scottish Inn, Inc., Vicksburg, Mississippi. The wastewater was collected with a small portable pump from near the chlorine inlet line and stored in two 208-liter epoxy-paint-coated drums for approximately 15 hours prior to application to the model. This allowed for dissipation of the residual chlorine, which approximated 4 ppm. The wastewater was pumped from the drums into two polyethylene barrels, adding equal volumes to each.

Since the wastewater was lower in nitrate and ammonium than most secondarily treated municipal wastewater and cannery waste, it was amended with ammonium nitrate to increase the final nitrogen concentration by 7.5 ppm for each component. The final concentration for nitrate plus ammonium was generally from 16 to 20 ppm nitrogen. In order to simulate the cannery waste in reduced organic components, 0.1 g of sucrose was added to each liter of influent to give a final chemical oxygen demand (COD) of approximately 200 mg/ℓ. The schedule involved running the model for 5 days, Monday through Friday. Due to additional weekend drying, 70 liters of wastewater were applied on Mondays, and 57 liters were applied on the other 4 days to approximate adding 6.35 cm of wastewater per week.

The wastewater was pumped from each of the two calibrated storage barrels into two Plexiglas trays, one for each side of the model. This was accomplished by means of a single peristaltic-action pump using a double head, with speed adjusted to give a flow rate of about 100 ml/min to each tray. This rate allowed for continuous application of the wastewater over a 5-hour period. The trays were located at the upslope end of the model and measured 0.76 m by 5.1 cm by 2.5 cm, with an overhanging lip on one side to allow for overflow onto the soil. Paper towels were cut in 7.5-cm-wide strips and placed along the length of the lip. The toweling acted as a hydrophilic wick and served to distribute the tray overflow more evenly. It was previously soaked to remove any soluble materials. Tests revealed that the paper towels did not significantly change the composition of the applied wastewater.

Wastewater Sampling and Analyses

Wastewater samples were collected from the inflow and outlet lines simultaneously, and both flow rate in milliliters per minute and time of collection were recorded at each location. Samples other than at

3.05 m were obtained from small pools at various locations on the model surface by means of a syringe. These sites were labeled for future collection. Bottom samples were also collected with a syringe through the rubber caps of the bottom ports. Amended and raw wastewater samples from the model were analyzed daily for nitrate and ammonium concentrations using Orion specific ion electrodes and an Orion Model 801 expanded-scale millivoltmeter. These readings were occasionally compared with results obtained using the chromotropic acid method for nitrate (61) and distillation-nesslerization for ammonium (65). Nitrite was determined with the sulfanilamide-N-1 naphthyl-ethylenediamine hydrochloride colorimetric analysis, using the Technicon Autoanalyzer II[b] (65). Total Kjeldahl nitrogen (TKN) determinations varied in frequency because of instrumentation problems but were made at least on a weekly basis, and COD tests (65) were performed regularly. Occasional analyses of the effluent from the soil system were made for phosphorus using the stannous chloride method (61), and for magnesium, calcium, potassium, sodium, manganese, lead, zinc, cadmium, nickel, and aluminum using a Perkin-Elmer Model 303 atomic absorption spectrophotometer. Measurements for pH were made for wastewater sampled at various locations within the experimental model using a glass electrode.

Soil and Plant Sampling and Analyses

Soil samples were also obtained at 0.84- and 2.29-m locations from each side of the model for compositional analyses. Each site was sampled from 0- to 7.5-cm and 7.5- to 15-cm depths. Moist soils were used for determining TKN and exchangeable ammonium, the latter being extracted with acidified sodium chloride. A composite of 20 samples from the soil surface was used to obtain a measure of TKN variability.

Upon drying the soil at 60 C for 48 hours, phosphorus was determined in an acidified salt extract using the previously described method. Atomic absorption was used for determining the magnesium, calcium, potassium, sodium, manganese, iron, aluminum, lead, zinc, cadmium, and nickel in the same extracts. Total readily available organic carbon was

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[b] Manufactured by Technicon Corp., Tarrytown, New York.

determined by wet digestion using the Walkley-Black method (19). Soil pH measurements were made on random samples obtained from the upper 5 cm of the model using a saturated paste.

The vegetation within the experimental model was almost exclusively Reed canary grass, which began to emerge from the soil shortly before initiation of the experiment. After 2 weeks of wastewater application when the grass had reached a height of about 30 cm, the grass on one side of the median divider of the model was cut to a height of 5 cm. This height was maintained weekly for a total of 4 weeks in order to allow for establishment of a thick algal slime layer and the study of pursuant changes in the model functioning. The other side of the box was maintained at a height of 30 cm during this period to serve as a partial control. After the 4 weeks, the grass was allowed to grow to the height of 30 cm to simulate normal regrowth of the grass after harvesting. Previous to regrowth, the model was on one occasion covered with a thick canvas cloth to prevent light from reaching the algal mat and thus to determine if photosynthesis by the algae had a significant effect on the function of the model. Complete regrowth of the grass on the cut side took about 2 weeks following the last cutting, and algal decay required an additional 2 weeks. Thus, for the final 2 weeks of experimentation, the model approached the ecological conditions that prevailed when the model was initially studied.

All grass cuttings were dried at 60 C for 96 hours and then ground for compositional analyses. Samples were analyzed for TKN, acid extractable ammonium and nitrate, phosphorus, magnesium, calcium, potassium, sodium, manganese, iron, aluminum, lead zinc, cadmium, and nickel, as previously described. Grass cuttings were obtained from different areas on the model in order to correlate changes in composition with the various locations.

Oxidation-Reduction Measurements

Oxidation-reduction potential measurements were monitored in the soil system of the model using bright platinum electrodes and a calomel reference electrode. The platinum electrodes were precalibrated in a saturated aqueous quinhydrone solution at a pH of 7.0 (25 C) before use, then inserted into the soil along with small wooden supports the day previous to monitoring. Electrode readings were recorded on a portable Orion Model 407 millivoltmeter. Random locations were sampled in the model at downslope distances of 0.6 and 2.4 m on both sides of the median divider. Two electrodes were placed less than 5 cm apart at each location, one at less than 5 mm into the soil surface just under or within the surface organic slime and the other at a depth of 13 mm within the soil. The four sites were monitored at random times over an 18-day period ending 1 week prior to the completion of experimentation. At this time, electrodes were installed at downslope distances of 0.8, 1.5, and 2.3 m and were monitored from Thursday through Monday of the final week in the manner previously described. The purpose of the latter measurements was to show the discrete fluctuations that occur during alternating periods of surface saturation and drainage on the model.

Experiment II

Model Construction and Location

The model used in Experiment II was quite similar to that used in Experiment I. The only significant difference was that the collection ports at the downslope end of the model were only at the bottom and the 15.2-cm soil surface levels. The model was located in the same greenhouse and under the same conditions as Experiment I.

Soil Collection, Preparation, and Grass Establishment

A low-permeability clay soil of the Susquehanna series was used in this study. Susquehanna soils occur in higher elevation areas of the Atlantic and Gulf Coastal Plains from Virginia to Texas, extending into western Tennessee, Arkansas, and Oklahoma. The soil was obtained in the DeSoto National Forest, 4 miles south of Camp Shelby in Forest County, Mississippi. The site was an eroded field supporting a sparse growth of grass and scattered pine trees and with clay subsoil exposed. The clay is hard and stiff when dry and highly plastic and sticky when wet but shows a granular structure on partial drying. Permeability to water in the field is estimated to be less than 0.15 cm per hour.

Surface soil to a depth of 15 cm was removed in a moist condition (28 percent water content) and stored in covered containers to prevent drying. Surface litter and grass tufts were removed before collection.

The soil was a clay loam according to U. S. Department of Agriculture textural classes, with 36 percent clay, 40 percent silt, and 24 percent sand. According to the Unified Soil Classification System, the soil was a heavy clay (CH) with 90 percent fines, a liquid limit of 65 percent, and a plasticity index of 37 percent. The soil was acidic with a pH at 5.0 for a 1:1 soil-water suspension. The cation exchange capacity was 25 milliequivalents per 100 g of soil and 35 percent saturated with bases (14).

The soil was sieved through a 1.27-cm wire screen and spread over the entire bottom of the model to a depth of 2.5 cm. Then, hardwood blocks weighted to 0.35 kg/cm^2 were placed without dropping or vibrating on the surface to compact this soil to a 1.25-cm-thick layer. After compacting, the surface was scarified to allow for better adhesion with the next layer. This procedure was continued until the compacted depth approximated 12.7 cm. Then, digested sludge (Table 1) from the Belzoni, Mississippi, Wastewater Treatment Plant was incorporated into the top compacted 2.5 cm on a 6 percent dry weight basis.

The model was then seeded with a 5:2:2:1 mixture of Reed canary grass, perennial rye grass (Lolium perenne L.), Kentucky 31 tall fescue (Festuca arundinacea), and Bermuda grass (Cynodon dactylon L.) at a rate of 38.5 $g/m.^2$ Once established, the upper layer was compacted in the same fashion as previously described.

Wastewater Collection and Preparation

Secondarily treated effluent was collected from the Scottish Inn site at the same time and in the same manner as in Experiment I. Preliminary tests showed that the nitrate and ammonium nitrogen levels were very low, approximately 1 to 2 ppm, so a solution of NH_4NO_3 was added to the effluent to increase the nitrogen concentration to 16 to 20 ppm. In

Table 1

Chemical Characteristics of Dried, Digested Sewage Sludge From a Municipal Trickling Filter Plant, Belzoni, Mississippi

Constituent	Total	Extractable NH_4OAC	Extractable H_2O
	Concentration, ppm		
Cd	5	0.66	0.30
Cu	212	2.20	0.44
Pb	197	0.50	0.19
Mn	304	40.95	4.80
Zn	1954	136.95	10.62
Al	562	6.56	5.22
K	753	169.16	42.54
Na	180	80.92	57.73
Mg	2382	250.16	46.29

TKN = 1.02 percent

Total P = 0.41 percent

addition, a heavy-metals solution consisting of lead acetate and zinc, manganese, copper, cadmium, and nickel chlorides was also added to boost the concentrations of each to approximately 0.3 ppm.

Wastewater Application, Sampling, and Analyses

A 1.27-cm (61-liters total) application of supplemented wastewater was put on the model each day over a 6-hour interval. These applications were made Monday through Thursday making a 4-day wet and 3-day dry cycle.

The runoff and subflow were collected daily in polyethylene carboys from each half of the model, and volumes were recorded. The supplemented wastewater, runoff, and subflow were analyzed daily for nitrates and ammonia as described in Experiment I.

Surface liquid samples were collected by use of a syringe along both sides of the model at distances of 0.76, 1.52, and 2.29 m, and subflow liquid samples were also taken along the length of the box. Nitrate, ammonium, and TKN; total phosphorus, potassium, calcium, magnesium, cadmium, copper, manganese, nickel, and zinc; and the COD were determined on these samples as previously described.

Soil and Plant Sampling and Analyses

The grass was cut to a height of 2.5 cm after 2 months of growth. By that time, the grass had reached a height of about 48 cm at the upper end of the box and 15.24 to 22.86 cm at the lower end. The box was divided into eight sections before harvesting, four 76.2 by 76.2 cm sections on each side of the median divider.

The grass was dried at 60 C for 48 hours and then analyzed for ammonium N, and TKN and total phosphorus, potassium, calcium, magnesium, cadmium, copper, manganese, nickel, and zinc according to the procedures described for wastewater samples of Experiment I.

After 9 weeks of application, soil samples were collected from several sites at downslope locations of 0, 0.76, 1.52, 2.29, and 3.05 m, at a depth of 0 to 2.5 cm, after removal of the surface organic layer. Moist soil samples were tested for TKN and exchangeable ammonium. The elements analyzed for the grass were also determined for soil extracts. All values were made on an oven-dry soil weight basis.

Oxidation-Reduction Measurements

Redox measurements were monitored as those in Experiment I had been during the 5-day period of the final week of experimentation, except the locations were 0.5, 0.8, 1.5, and 2.3 m down the length of the model.

RESULTS AND DISCUSSION

Analysis of Test Results from Experiment I

Nitrogen removal in overland flow treatment of cannery wastewater is quite good, and denitrification has been suggested as a significant component in the process (36). Cannery wastewater, however, is normally quite high in organic and ammonium nitrogen and requires nitrification before denitrification can proceed. In addition, denitrification requires the absence of oxygen, while nitrification and plant growth require oxygen. Thus, an enigma concerning nitrification and denitrification in overland flow systems exists.

The process of overland flow treatment of wastewater has been studied in large and small field plots, and the number of nitrification and denitrification studies of small and large sizes is myriad (4,15,26, 36,57). For this study, however, a compromise between a well-controlled bench-scale study and a less-controlled field-scale study was used. A 1.52- by 3.05-m model located in the greenhouse offered the opportunity for relatively controlled conditions, intensive sampling, on-site instrumentation, and access during inclement weather.

During this initial 12-week study, attempts to learn of the treatment efficiency, range in variation under existing conditions, and mode of model functioning were made. A balance sheet for nitrogen was attempted by partitioning data into soil, plant, and runoff liquid categories. Although it was realized that N^{15} studies would have to be made for a more quantitative accounting of nitrogen, it was felt that a qualitative accounting, interpreted in relation to the conditions of the model, would be best in this phase of the study.

Account of Nitrogen in the System

The model, having a soil volume of 0.698 m^3 and a downslope length of 3.05 m, represented a highly impervious soil which had previously been irrigated with cannery wastewater under field conditions for 8 years. Its nitrogen content was 0.179 percent. Yet, after 12 weeks of fortified wastewater application to the soil surface, an average of

91.4 percent of the applied nitrogen had been either immobilized or volatilized, with both ammonium and nitrate being removed rather well (Table 2). Including organic nitrogen, 81.7 percent of the total nitrogen was removed. Table 3 gives average percent removal efficiencies for nitrate, ammonium, and organic nitrogen after passage of the fortified effluent over the length of the model. Tables 4-6 and Figure 4 show how these efficiencies fluctuated with time. All values used in calculations for the efficiencies, including concentration and volume fluctuations, are given in Appendix I, Tables I-1 through I-4. Figures 5-7 show the concentration fluctuation for nitrate, ammonium, and TKN in the applied wastewater and runoff, and Figure 8 depicts the losses in wastewater volume after passage over the model.

During the first 4 weeks of analysis, nearly all wastewater nitrate was removed. Significant but lower losses of ammonium also occurred. The relatively high organic nitrogen values were thought to be stimulated by the addition of excess sucrose to the wastewater (0.35 g/ℓ) during the first 5 days of the experiment. At that time, COD values approximating cannery waste were being sought. A sugar concentration of 0.1 g/ℓ in the wastewater gave an appropriate COD (approximately 225 mg/ℓ), and this ratio was maintained throughout the remainder of the experiment (Table I-5). The decrease in nitrate treatment efficiency during the fifth through tenth weeks appeared to be related to an increase in algal growth and will be discussed in the following section. Concentration of ammonium and organic nitrogen decreased during the fifth through the eighth weeks. This reflects the great decrease of these nitrogen forms from the half of the model on which algae were not allowed to develop. The side that was covered with the algal crust continued to give high TKN values.

Although the reasons for these differences are not fully understood at this time, several hypotheses are given in the following two sections, which include physical separation of the surface liquid and nitrogen fixation by the algal layer. Removal efficiency variations did not appear to be dependent on temperature fluctuations in the greenhouse during the course of experimentation (Table I-12).

Table 2

Average Nitrate, Ammonium, and Organic Nitrogen[a] Values for Applied Wastewater, Runoff, and Subflow From the Cannery Wastewater Model

Liquid	Volume liters	Nitrogen Source	Nitrogen Concentration mg/ℓ	Nitrogen, g
Wastewater	3212	Nitrate N	9.70	31.16
		Ammonium N	8.24	26.47
		Organic N	4.16	13.36
		Total N	22.10	70.99
Runoff	1474	Nitrate N	1.96	2.89
		Ammonium N	1.31	1.93
		Organic N	4.89	7.21
		Total N	8.16	12.03
Subflow	233	Nitrate N	0.37	0.09
		Ammonium N	0.16	0.04
		Organic N	3.60[b]	0.84
		Total N	4.13	0.97

a Represents TKN values excluding ammonium.
b Based on one measurement.

Table 3

Average Percent Removal Efficiencies From the Cannery Wastewater Model for Nitrate, Ammonium, and Organic Nitrogen During a 12-Week Application Period

	Percent Removal
Nitrate N	90.4
Ammonium N	92.6
Organic N	39.8
Total N	81.7

Table 4

Nitrate Nitrogen Removal Efficiencies for the Cannery Wastewater Model

Day	Percent Removal for Cited Week												
	1	2	3	4	5	6	7	8	9	10	11	12	Mean
Mon		99.91		98.47	89.30	83.23	98.46	94.97	86.84	89.01	100.00	97.44	93.76
Tue		98.78	95.42	96.38	90.26	86.81		79.64	71.13	75.14	85.13	87.90	86.66
Wed		100.00	96.11		86.70	83.83	80.73	79.80	81.75	83.45	88.75		86.79
Thu	98.87	98.37	96.77	92.44	98.16	92.30	78.03		81.05		87.50		91.50
Fri	100.00	98.06	96.67	96.14			66.10	80.32	81.65		92.92		88.98
Mean**	99.43	99.02	96.25	95.86	91.11	86.54	80.83	83.68	80.48	82.53	90.86	92.67	

** Significantly different at the 0.01 level using the F test.

Table 5

Ammonium Nitrogen Removal Efficiencies for the Cannery Wastewater Model

Day	Percent Removal for Cited Week												
	1	2	3	4	5	6	7	8	9	10	11	12	Mean
Mon		99.52		84.70	98.09	92.44	99.59	98.51	90.54	91.69	100.00	99.10	95.42
Tue		93.78	94.03	88.16	91.42	91.81		95.06	97.64	81.25	90.56	94.86	91.86
Wed		87.65	86.70		91.97	89.21	94.04	91.43	88.99	86.92	89.36		89.59
Thu		84.72	74.96	96.99	96.75		94.67		90.45		91.55		90.02
Fri	92.00	81.82	84.43	74.23			92.75	94.29	87.83		96.26		87.95
Mean N.S.	92.00	89.50	85.03	86.02	94.56	91.16	95.26	94.83	91.09	86.62	93.55	96.98	

N.S. Not significantly different at the 0.01 level using the F test.

Table 6

TKN Removal Efficiencies for the Cannery Wastewater Model

Day	Percent Removal for Cited Week												
	1	2	3	4	5	6	7	8	9	10	11	12	Mean
Mon				73.74	86.07			92.96		85.69	99.99		87.69
Tue			62.03	59.05	68.03			83.05		64.79	84.20		70.19
Wed		73.57	73.89	46.35				83.53		76.79	82.41	92.84	75.63
Thu		69.25	58.63	51.76							82.85		65.62
Fri	58.84	35.35	68.87										54.35
Mean**	58.84	59.39	65.86	57.73	77.05			86.52		75.76	87.36	92.84	

** Significantly different at the 0.01 level using the F test.

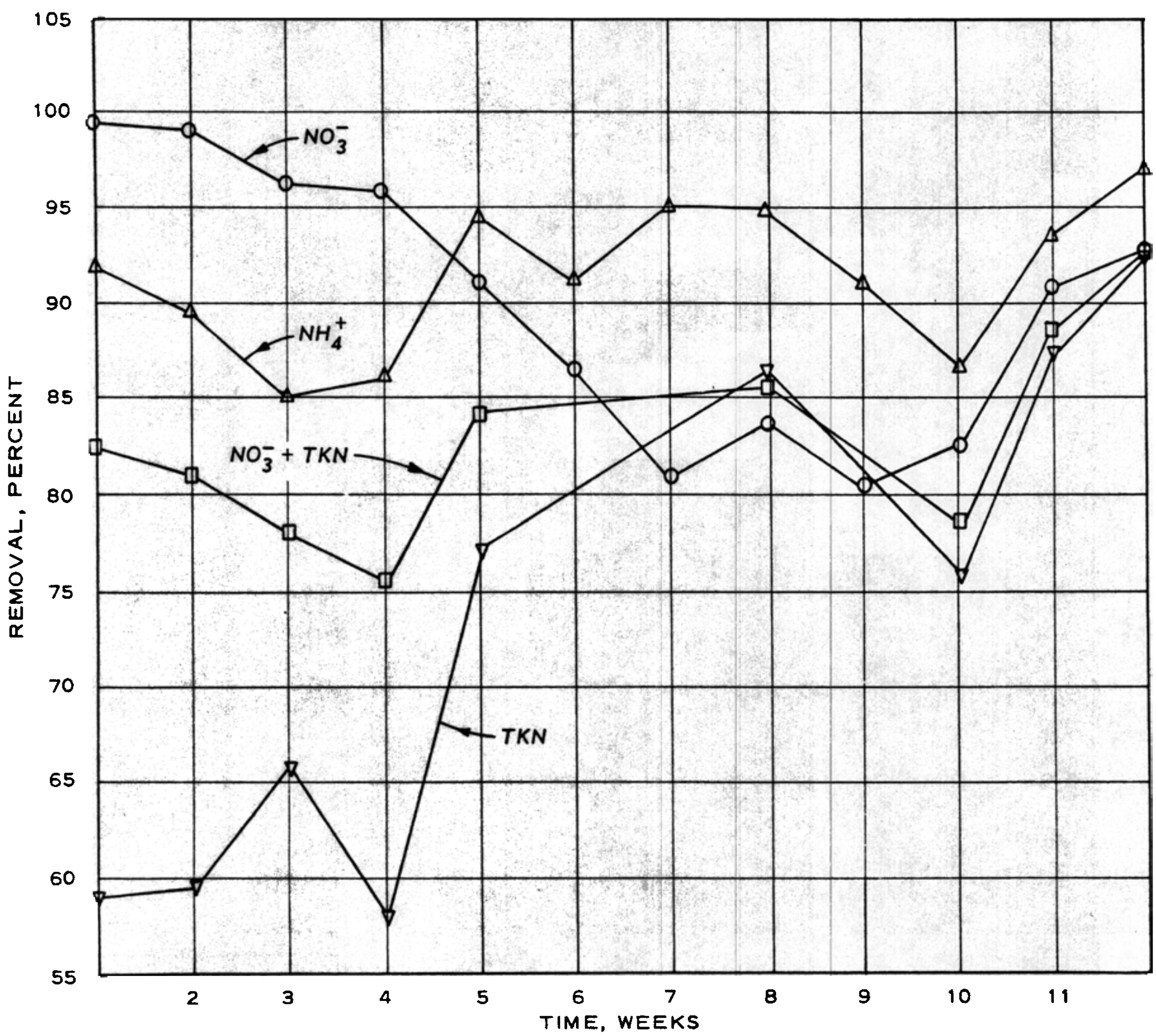

Figure 4. Nitrogen treatment efficiency of the cannery wastewater model during the study period.

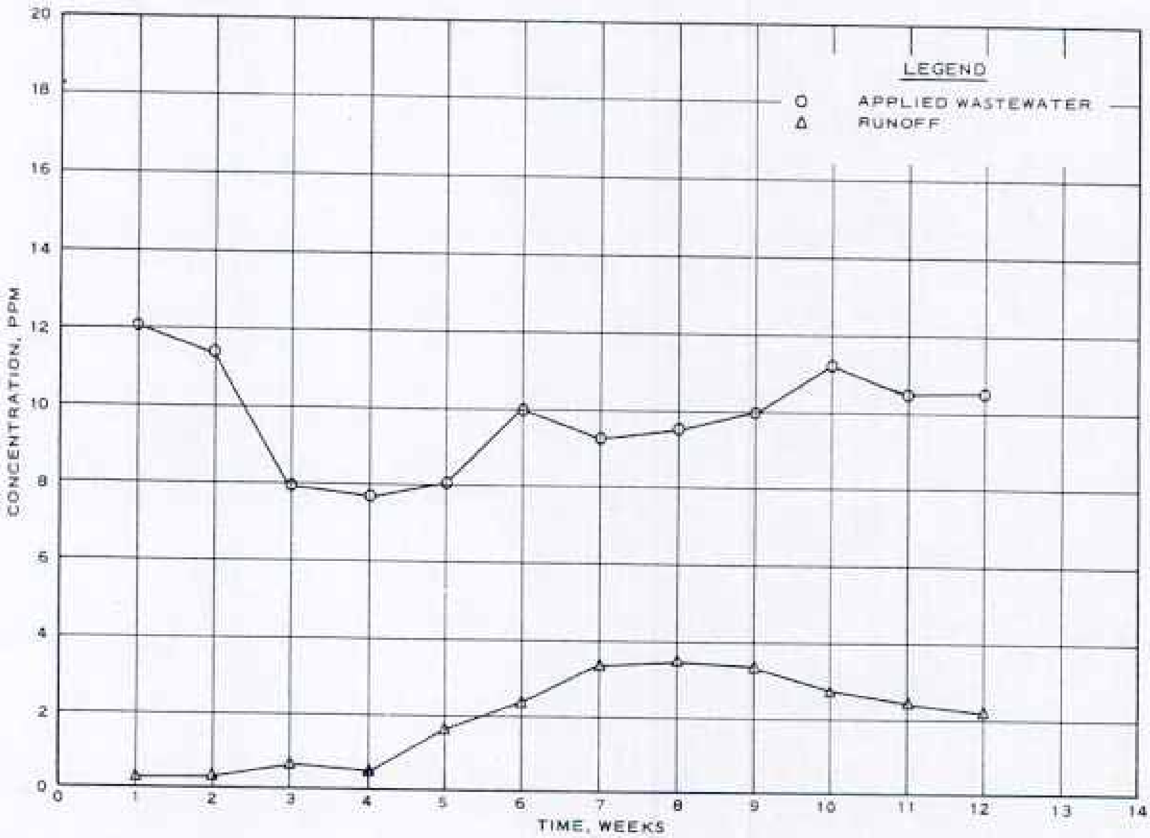

Figure 5. Nitrate concentrations in wastewater and runoff of the cannery wastewater model.

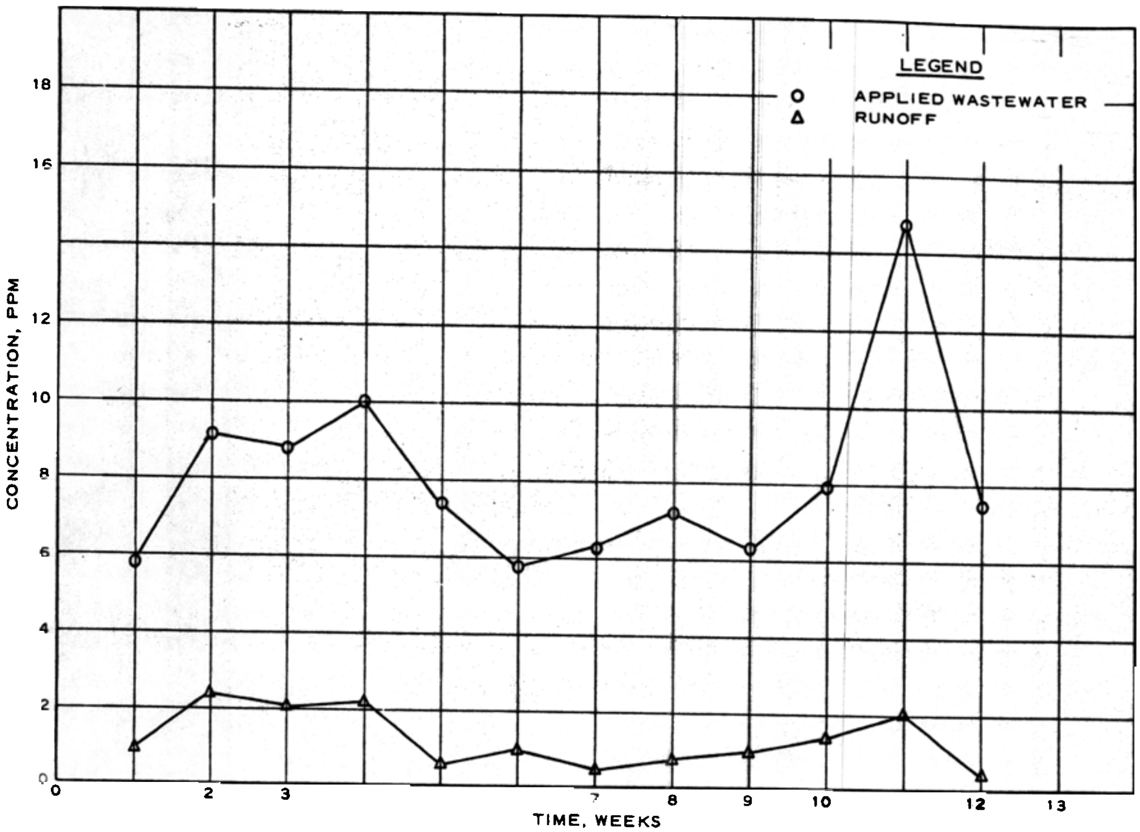

Figure 6. Ammonium concentration in wastewater and runoff of the cannery wastewater model.

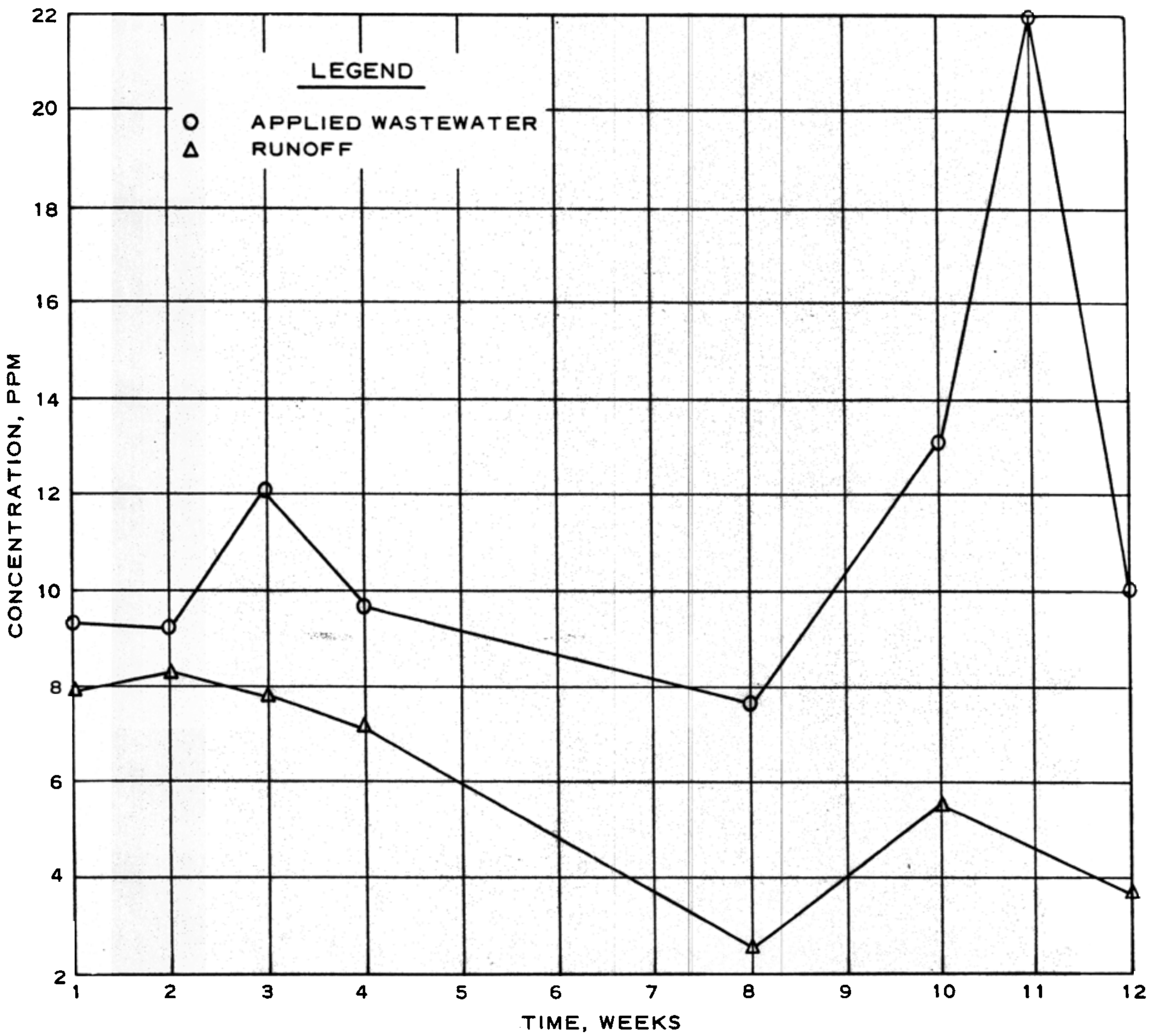

Figure 7. TKN concentration in wastewater and runoff of the cannery wastewater model.

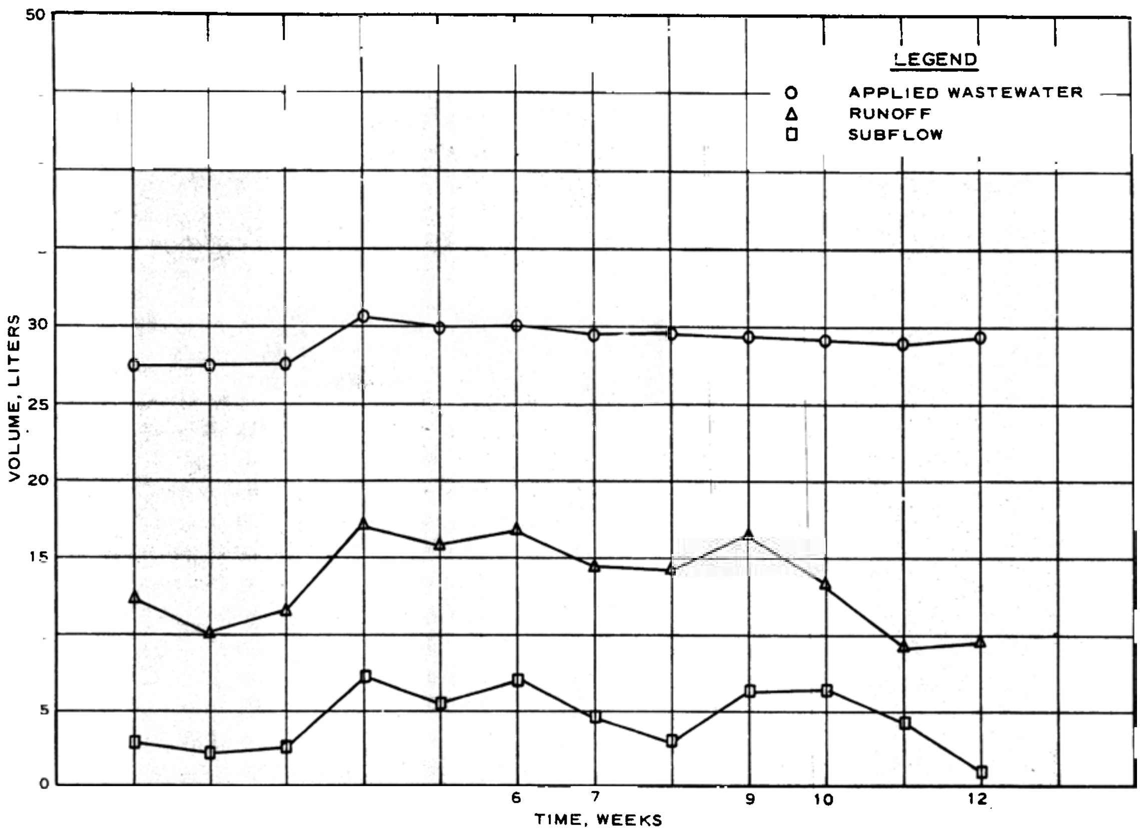

Figure 8. Daily liquid volumes for the cannery wastewater model.

Grass cut from the model was analyzed for TKN, phosphorus, and certain macro- and microelements (Table 7). Based on the TKN data, 53.9 g of nitrogen were incorporated into 1544 g (3320 kg/ha) of grass tissue (dry weight) harvested during the experiment. If there was no incorporation of soil organic matter nitrogen, this would account for 75.9 percent of the 70.99 g of total nitrogen added to the applied wastewater. Reed Canary grass has been shown to be effective in removing nitrate nitrogen from soil (33). However, mineralization and utilization of nitrogen from the soil most likely added significantly to the plant nitrogen. The percent of total nitrogen in the grass tended to decrease slightly with greater distance from the applied wastewater source. Values for the 0- to 0.76-m, 0.76- to 2.29-m, and 2.29- to 3.05-m sections were 39.4, 34.0, and 26.6 percent, respectively.

The soil was also analyzed for organic nitrogen, ammonium nitrogen, organic carbon, and 12 other elements (Table 8). The final analyses were made with soil obtained from 0- to 7.5-cm and 7.5- to 15-cm depths at four different sites. Additionally, seven TKN analyses were made using a composite sample representing 20 different soil locations within the model. Initial soil nitrogen, determined from random samples obtained from the soil profile prior to growth of the dormant grass, was 0.179 percent on an oven-dried soil weight basis. Final nitrogen concentrations in the soil gave an average value of 0.217 percent, but levels in the upper half of the profile were three times those in the lower half. Also, concentrations in the upslope half of the model were nearly three times those from the downslope sampling areas. Free ammonium nitrogen in final soil analyses constituted a negligible fraction of the total soil nitrogen (5.6 g) and failed to show a concentration pattern. Soil nitrogen calculations indicated an increase of 0.038 percent during the course of the experiment. However, using a calculated bulk density of 1.33, this small change accounts for an increase of 211.6 g in nitrogen. This value is of course unrealistic since only about 71 g of nitrogen were added. Yet, it emphasizes the previously stated problem concerning the construction of a nitrogen balance sheet without the use of N^{15}. This is especially true in a

Table 7

Elemental Analysis of Reed Canary Grass From the Cannery Wastewater Model

Element	Percent Analysis	Total, g	kg/ha
N	3.49	53.89	115.97
P	0.32	4.94	10.63
K	3.13	48.33	104.01
Ca	0.20	3.09	6.65
Mg	0.14	2.16	4.65
Na	0.05	0.77	1.66
Mn	0.009	0.140	0.30
Zn	0.005	0.077	0.17
Fe	0.012	0.185	0.40
Al	0.007	0.108	0.23
Cd	0.00		
Ni	0.00		
Pb	0.00		
Cu	0.00		

Table 8

Elemental Analysis of Soil Extract From the Cannery Wastewater Model after Completion of the Experiment

Element	Percent Analysis	Total, g[a]
C	3.670	20,433
N	0.217[b]	1,208
K	0.046	256
Ca	0.760	4,231
Mg	0.053	295
Na	0.049	273
Mn	0.003	17
Zn	0.000	
Fe	0.000	
Al	0.000	
Cd	0.000	
Ni	0.000	
Pb	0.000	
Cu	0.000	

[a] Estimated on the basis of a bulk density obtained of 1.33 and a soil volume of 697,687.5 cm^3.

[b] Moist soil used in chemical analyses.

soil that already has a high nitrogen content.

Although the soil system could not be properly evaluated, a better picture was realized by comparing the more precisely measured soil water and grass data. The amount of nitrogen applied was 70.99 g (Table 2). Of this, 17.1 percent (12.14 g) was accounted for in the runoff, and 75.9 percent (53.9 g) was calculated as being removed from the system by the Reed canary grass. This leaves a deficit of only 7.0 percent (4.97 g), which could have either remained in the soil or been denitrified. It is most likely that a major portion of the plant nitrogen came from the soil nitrogen rather than the applied nitrogen, and an amount considerably greater than 7.0 percent would have been denitrified. Yet, it is not possible to determine the fate of this nitrogen deficit or whether grass nitrogen values actually reflect the true deficit. However, it would appear that the soil system should have reached a dynamic equilibrium in terms of nitrogen incorporation. As will be pointed out in the discussion on Eh studies in the model, the conditions for significant denitrification were definitely present.

Physicochemical Inhibition by an Algal Layer on Nitrogen Removal

Figures 9-12 represent schematics of the ways the surface area of an overland flow system might function. The aerobic-anaerobic double-layer concept of Figures 9 and 10 has already been discussed in the literature review, and it is considered to be the means by which nitrification and denitrification occur in marsh and rice paddy soils (49). When an abnormally high oxygen demand is placed on the system, a situation similar to that shown in Figures 11 and 12 might occur. Figure 11 depicts what may happen when an actively respiring filamentous algal layer rests within the surface liquid of an overland flow system, acting as a physical barrier by impeding the rate of nitrate and ammonium diffusion between the surface liquid and the liquid flowing beneath the algae. This may also allow for greater biochemical interaction, thus retarding both oxygen and nitrate from reaching the diverse microbial population at the soil surface. Oxygen would be eliminated by biological respiration within and beneath the crust. Figure 12 shows a situation

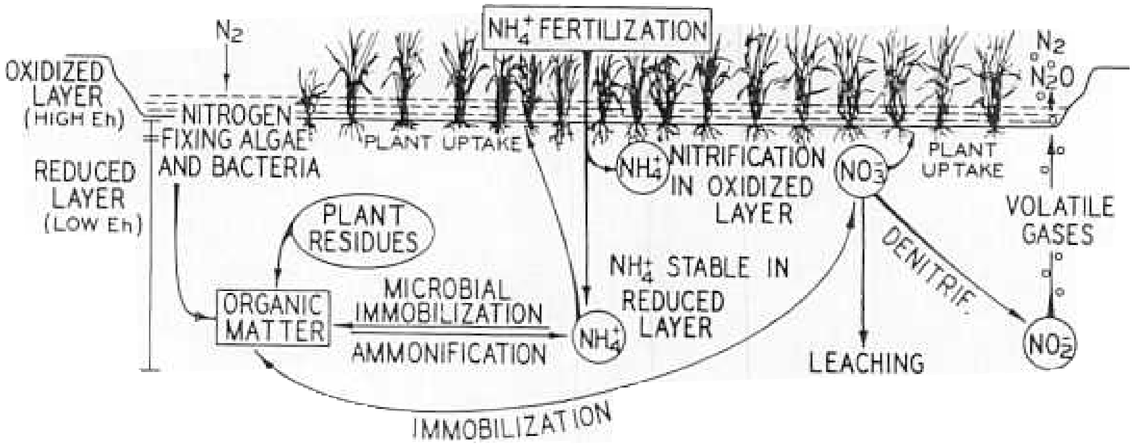

Figure 9. Nitrogen transformations in flooded soils (after Patrick and Mikkelsen (50)).

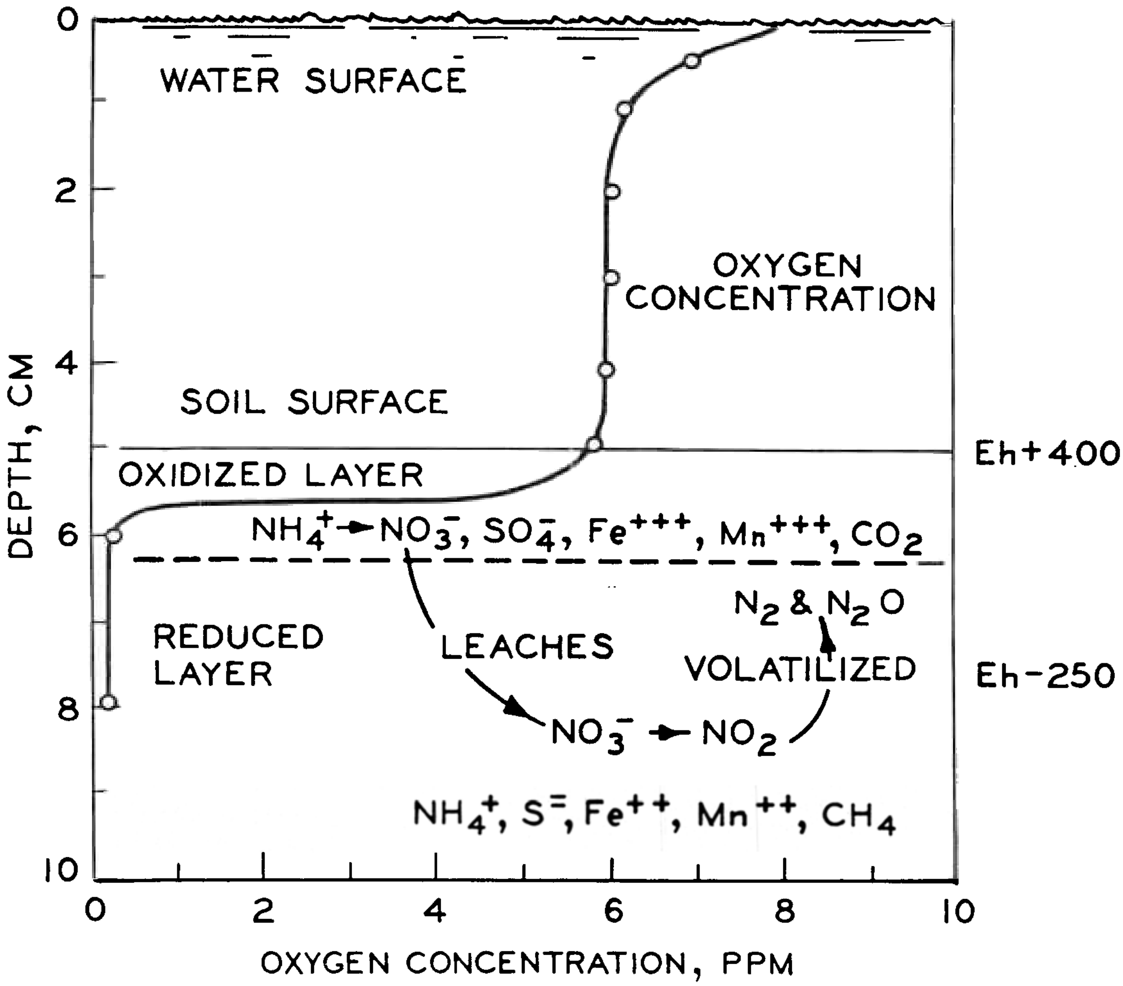

Figure 10. Oxygen profile in flood soil (after Patrick and Mikkelsen (50)).

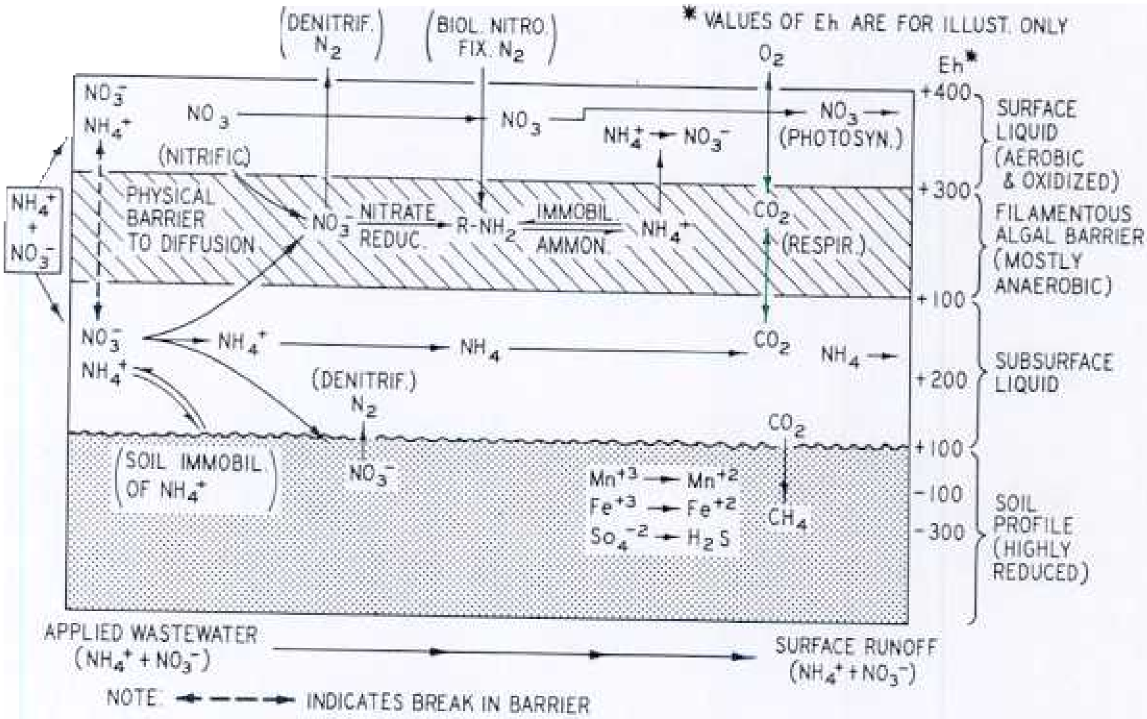

Figure 11. Cross section of a flooded soil.

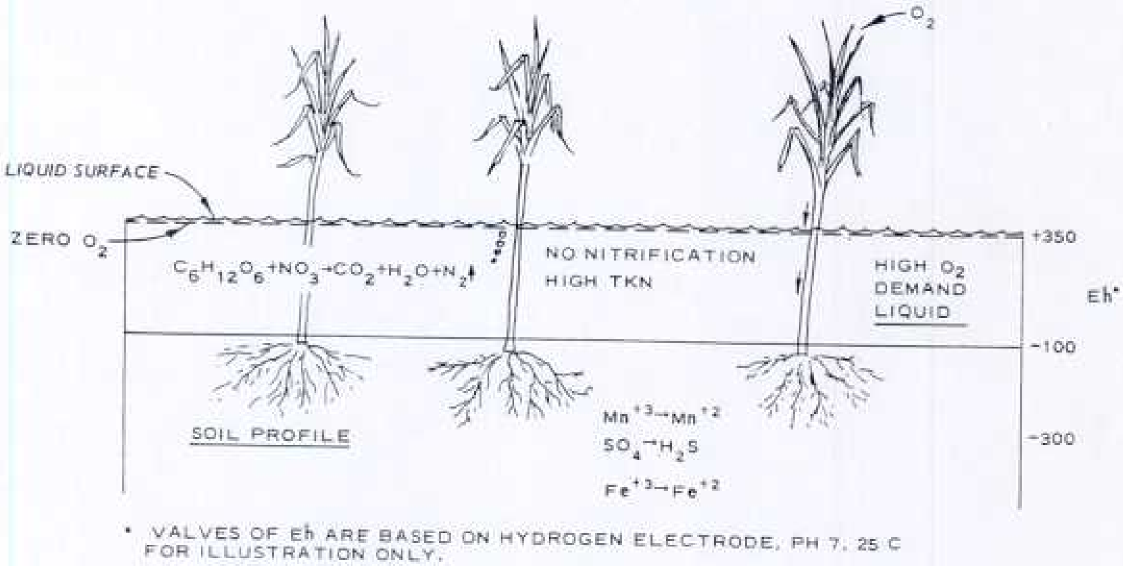

Figure 12. An overland flow surface with the aerobic zone remaining in the liquid flowing over the soil profile.

in which accentuated microbial metabolism in the surface liquid, due to a high level of reduced organic matter therein, almost eliminates an aerobic zone.

In order to study these systems, the grass was cut to a height of about 5 cm on one-half of the model at the end of the second week (Appendix I, Figure I-1A). This resulted in the development of a thick, tough, slimy algal crust on the surface exposed to greatest light (Figures I-1B and I-1C). Only an inconspicuous "bacterial" slime was observed on the shaded side (Figure I-1D). Optical and scanning electron microscopic analysis of the thick slime showed that it was composed primarily of tightly packed parallel filaments belonging to species of the blue-green alga, Oscillatoria. The blue-green algal genus Anabaena became prominent around the margins of pools, especially at the downslope end of the box, where it produced very thick, dark mucilaginous masses. Planktonic forms of algae, protozoans, and worms were more abundant in association with the blue-green algal mat (Table I-6).

During establishment of the algal slime, organic nitrogen values in the runoff dropped conspicuously, and ammonium nitrogen values also decreased noticeably. Conversely, nitrate removal efficiency decreased markedly on the light-exposed side of the model that supported the algal crust, and this trend persisted in the presence of the algae. Nitrate reduction was also observed to be slightly hindered at this time on the shaded half of the model, upon which the grass was left at a height of about 30 cm. Once the algal crust was fairly well established, ammonium levels also increased in runoff on the side exhibiting the algae, while only trace levels were generally observed on the side lacking the algae.

At the end of the sixth week, the grass was allowed to regrow on the cut side of the box, although noticeable disappearance of the algal mat was not observed until about the end of the ninth week. During this period of grass regrowth, nitrate reduction efficiency stabilized and then increased at a substantial rate. However, ammonium nitrogen in the surface runoff reached its highest levels at this time, averaging 4 to 5 ppm in effluent from the surface exhibiting algal decay. Ammonium concentrations continued in trace amounts in surface liquid from the other half of the model. This ammonium was thought to be derived from mineralization of nitrogen in the algae. The control side elicited complete removal of nitrate by the twelfth week, whereas the side on which the slime had developed continued to improve in this respect until the culmination of the experiment. During the final week, total nitrogen removal efficiency was greater than for any other week. Thus, the inhibition was apparently not correlative with permanent functional damage.

The rather dramatic decrease in nitrate and ammonium removal during the period of algal growth on the model gave strong support to the hypothesis that the algal layer did not increase nitrogen treatment efficiency by enhancing denitrification. Nitrogen fixation by certain algae could accentuate the buildup of amino nitrogen (69), but how this would be correlative with poorer nitrate removal is unknown at this time. Further work is needed in this area.

The half of the model that displayed luxurious grass growth and no algal mat probably functioned similarly to the aerobic-anaerobic double-layer system depicted in Figure 10. Treatment on this half was superior to that on the side with the surficial algae, but before and after the algal crust developed, treatment was similar over the entire model. <u>These observations lead to the hypothesis that a system that functions similarly to a rice paddy is better than one in which a blue-green algal layer is prevalent</u>. One possibility as to why the filamentous algae caused a drop in treatment efficiency is that it physically separated the nitrate and ammonium concentrations from the soil (Figure 11). Engler[c] has shown that denitrification does not readily occur in floodwater over a soil but in the soil. When water from over an actively denitrifying system was separated from the soil, no denitrification occurred in the liquid. Yet, when the water was again placed over the soil, denitrification occurred. The algae could act as a medium for denitrification by supplying anaerobic microsites. However, the algal barrier could also have blocked the movement of ammonium from the anaerobic soil layer to the aerated surface, which could overshadow

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[c] Personal communication, Dr. R. M. Engler, OES-WES, Vicksburg, Miss.

nitrogen loss attributable to the algae. Additionally, the nitrifying population on the aerobic surface side of the barrier could have remained small among the algae-associated microbial populations, and accordingly, rapid nitrification may not have been occurring even when ammonium was present.

The experimental basis for the above hypotheses is that on several occasions, liquid collected with a syringe from below the algal layer was compared with the surface liquid for nitrate and ammonium concentrations. The nitrate concentration decreased threefold in liquid sampled from beneath the algae, while the ammonium concentration remained about the same throughout. This indicates that the previously described concept could be operative.

In summary, it could be concluded that for nitrification and denitrification to function on an overland flow system, there must be an aerobic and an anaerobic layer in the soil. Extensive blockage to diffusion or nitrogen transformations between these layers must not occur.

Assuming that this conclusion is correct and an aerobic-anaerobic layer is required in the soil, an interesting hypothesis concerning the need for rest periods for overland flow systems can be made. If the system uses a highly organic wastewater (e.g., cannery waste), the demand for oxygen might become so great that the aerobic layer in the soil would become quite thin or the aerated zone might move into the surface liquid as depicted in Figure 12. The absence of a well defined aerobic zone could cause great reduction in both nitrification and denitrification. This would correspondingly result in a deterioration of treatment efficiency. A rest period under aerobic conditions would reduce the oxygen demand of the soil surface and stimulate nitrification. When wastewater was reapplied to the system, the thickness of the aerobic layer would decrease, and this would result in an increased removal rate for nitrate. These types of responses have been observed in the field (36), and it will be interesting in future research to see if the proposed hypothesis of double-layer disappearance is in fact true. <u>If it is true, it would follow that a wastewater such as secondarily treated municipal effluent with sufficient but not excessive BOD might be treated</u>

better on an overland flow site than cannery waste. However, the authors caution that these are only hypotheses extrapolated from limited data and literature, and they should not be considered proven. They are discussed instead to give the reader the scope of possibilities for nitrogen treatment on an overland flow system.

Biochemical Inhibition by an Algal Layer on Nitrogen Removal

In addition to the barrier concept discussed previously, it was suspected that algae could inhibit denitrification by two other methods. One is by the simple production of oxygen during photosynthesis which could inhibit denitrification (WES Miscellaneous Paper Y-73-1). Jannasch noted this in experiments with the green alga *Chlorella* (30). The other method is by the production of a toxic product or excretion of a by-product that would accumulate to toxic levels under anaerobic conditions.

In order to test whether oxygen production by photosynthesis and the associated aerobic conditions were reducing nitrate removal efficiency on the algal-covered side of the model, a cover was placed over the model on the afternoon before a wastewater application was made. This would eliminate any production of oxygen the following day and during the application period. No significant change in either ammonium or nitrate treatment occurred. This indicates that the poor nitrogen treatment was not significantly related to oxygen production. Perhaps the extensive proliferation of organisms within the algae was depleting the oxygen as rapidly as it was being produced. For additional evidence, redox electrode potentials at the algae-soil interface and at 1.2- to 1.5-cm soil depths were compared with similar depths on the side lacking the algae. These electrodes were monitored for 3 weeks (Figure 13; Table I-7). With the exception of one abnormally high value obtained beneath the algal crust during the first week, the redox values from the two sides were surprisingly similar. Additionally, the potentials were generally in the reduced range during the wastewater application periods and in the oxidized range during the weekend drying period. This is also in agreement with the redox potential findings

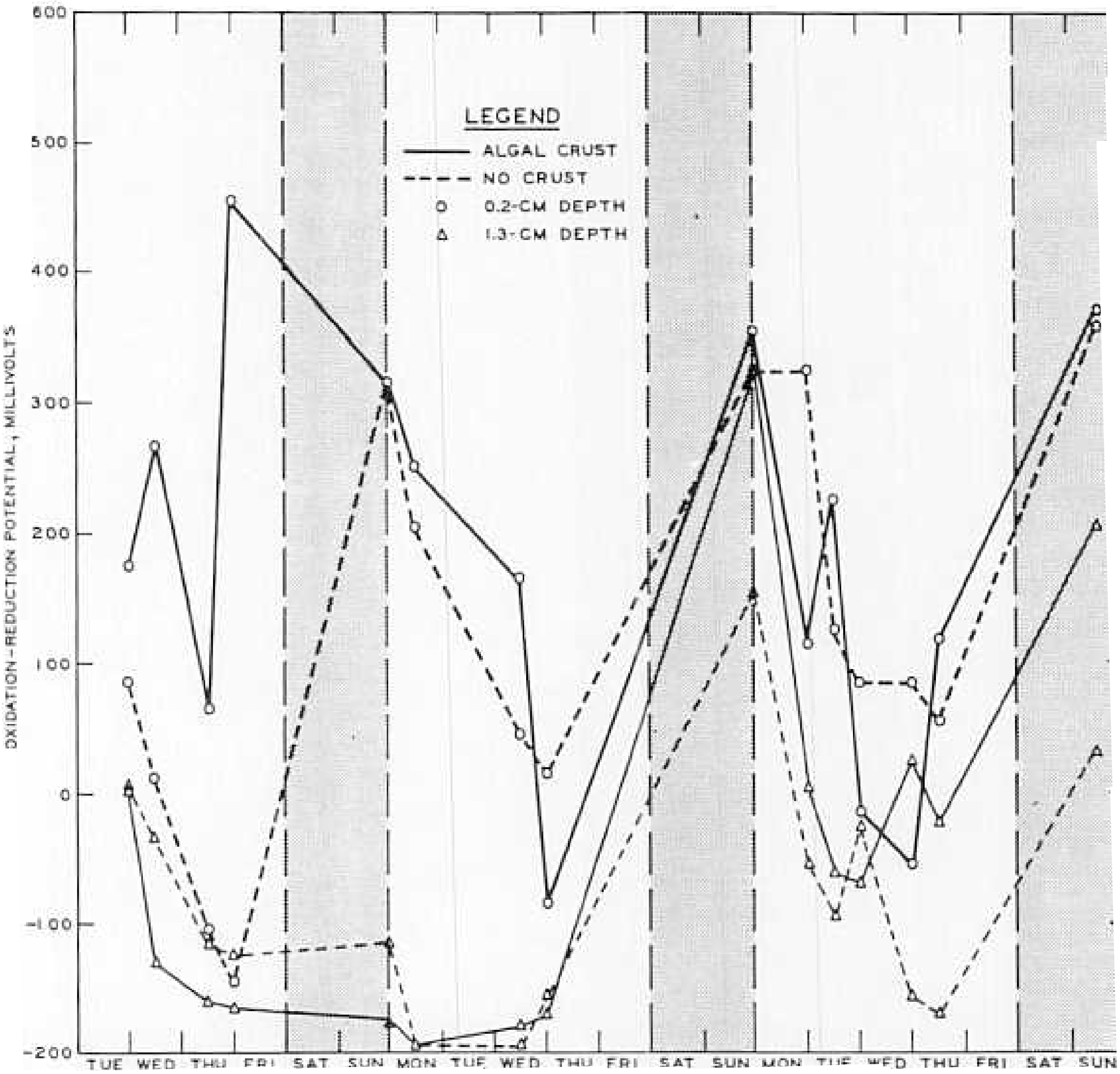

Figure 13. Adjusted oxidation-reduction potentials within the cannery wastewater model over a 3-week period, showing application times (light), drying times (dark), and soil pH (5.0).

described in the following section. From the results of these two studies it was concluded that extensive anaerobic conditions probably persisted in the presence of the algae, and thus aeration by algal photosynthesis was not a significant factor in the decline of nitrogen treatment efficiency.

Many species of the blue-green alga Anabaena are known nitrogen-fixers, and some are also considered to be the most toxic of the blue-green algae, whereby the toxin is released only by living cells. Other inhibitive substances are also known to be produced by various blue-green algae and other organisms which are in close association with them (52,69).

To test for an inhibitory effect, the blue-green algal slime was collected from the surface of the model. A centrifuged cell extract of the identified genera, Oscillatoria, Anabaena, and Cylindrospermum, was then added to fortified wastewater in flasks and bubbled with nitrogen gas to remove oxygen. No inhibition of denitrification appeared to occur when compared with controls lacking algal extract or to which boiled extract was added. The removal values were 25.7 percent without algal extract, 39.1 percent with algal extract, and 40.6 percent with boiled extract after 3.5 days of incubation at room temperature. Yet, no conclusions should be drawn from this data due to small variation in nitrogen removal, lack of significant growth of Anabaena at the time of sampling, and limited experimentation. It can only be stated that inhibition of nitrate reduction by algae still appears to be a viable possibility and should be investigated more thoroughly.

Oxidation-Reduction Potential Study

In order to further establish whether conditions for denitrification were actually occurring very near the soil surface and how these varied over the rest period, bright platinum electrodes were placed at 0.2- to 0.5- and at 1.2- to 1.5-cm depths at six locations on the model. Electrode measurements were then made from Thursday through Monday (Figure 14; Table I-8). This allowed for a 2-day wet, 2-day dry, and 1-day wet cycle, respectively. The soil pH was 7.0, and the effluent was normally close to neutrality (Table I-9). Under

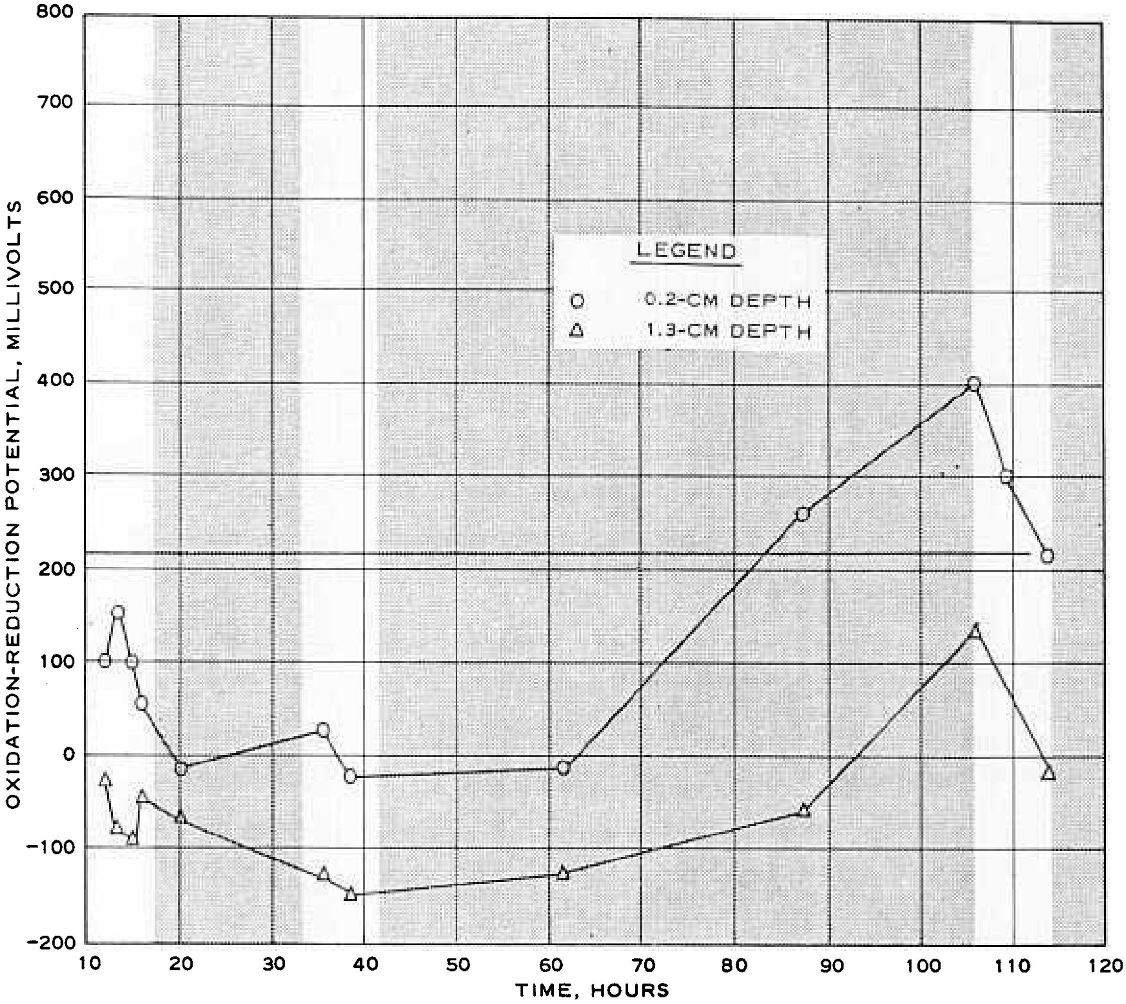

Figure 14. Adjusted oxidation-reduction potentials within the cannery wastewater model during a wastewater application and drying cycle. Stippled areas represent the rest periods.

these conditions, an Eh of approximately +220 mv or less would cause the nitrate to be unstable (49).

It was noted that even during the weekend, the Eh at 1.3 cm remained in the range of nitrate instability. This indicated that at the 1.3-cm depth constant denitrification was occurring. Thus, any nitrate that reached that depth would be denitrified. However, ammonium at that depth would be stable, for no nitrification would be occurring. The Eh of the 0.2- to 0.5-cm layer oscillated from reduced to oxidized to reduced conditions during the wetting-drying-wetting cycle. Unfortunately, the measurements were not near enough to the soil surface to discern the presence of an aerobic layer that might have been no more than a fraction of a millimeter in depth. Its presence was assumed, but proof of its existence in this oxygen-demanding system will have to await further experimentation. It was also noted that the 2- to 5-mm depth became reduced within a few hours after wastewater was reapplied on Monday.

A highly significant difference in Eh values existed with respect to both depth and time. In addition, Experiment II (Appendix II, Table II-5) and a parallel study (16) had similar results with highly significant depth and time differences. These findings show that conditions for denitrification definitely existed at the studied depths and near the soil surface during wastewater application and strongly support the hypothesis that denitrification is a significant process in overland flow treatment of wastewater.

An observation concerning Reed canary grass growth was made in conjunction with Eh values. Throughout the experiment, the grass growth was quite luxurious, despite the presence of Eh values too low for oxygen to exist in most of the soil profile. Thus, most likely the grass was readily transporting oxygen from the leaves to other parts of the plant as does the rice plant (7,8). Thin sections from the stem and crown of the Reed canary grass growing on the model showed large intercellular cavities in areas that were originally conductive bundles.

Effect of Downslope Distance on Nitrogen Treatment

In order to correlate downslope distance with losses of inorganic nitrogen, concentrations of nitrate and ammonium were monitored over the length of the model from designated surface sites and bottom ports during the course of experimentation. Surface concentration distributions showed a characteristic pattern (Figure 15), whereas subflow values were usually in trace amounts at all locations. When the same locations on the soil surface were repeatedly monitored during weeks 2 through 4, before the algal crust had become established, nearly half of the total nitrate and ammonium losses from the amended wastewater were incurred within the initial 30 cm.

Wijler and Delwiche (66) noted that the rate of denitrification is independent of the nitrate concentration in soil. Ideally then, it would be expected that a constant rate of decrease in concentration as the slope distance increases would occur. The nonlinear slope obtained could be partly explained by loss in runoff of applied wastewater volume through evapotranspiration, which averaged 54 percent (44 to 68 percent) through the 12-week period (Figure 8; Table I-4) Provided that no other factors were involved, the true rate of nitrate reduction should parallel this initial slope, excluding concentration due to the evaporation.

Since the soil had an average C/N ratio of 16.9 and the wastewater had a sucrose concentration of 150 mg/ℓ, immobilization of mineral forms of nitrogen by both microbes and plants seemed evident (1). This assimilation into living tissue would also accentuate the initial uptake of ammonium nitrate. The near zero slopes shown by surface flow concentrations from the lower half of the model probably indicate a nonlinearity in removal rate at concentrations below about 1 ppm. A similar finding was reported by Law, et al. (36) for a cannery overland flow system. They found 15.0, 0.7, and 1.0 ppm total nitrogen after 0, 12.19, and 30.48 m of overland flow, respectively.

When the same areas on the model were monitored during the ninth week, the algal crust was present on half of the model, and the graph

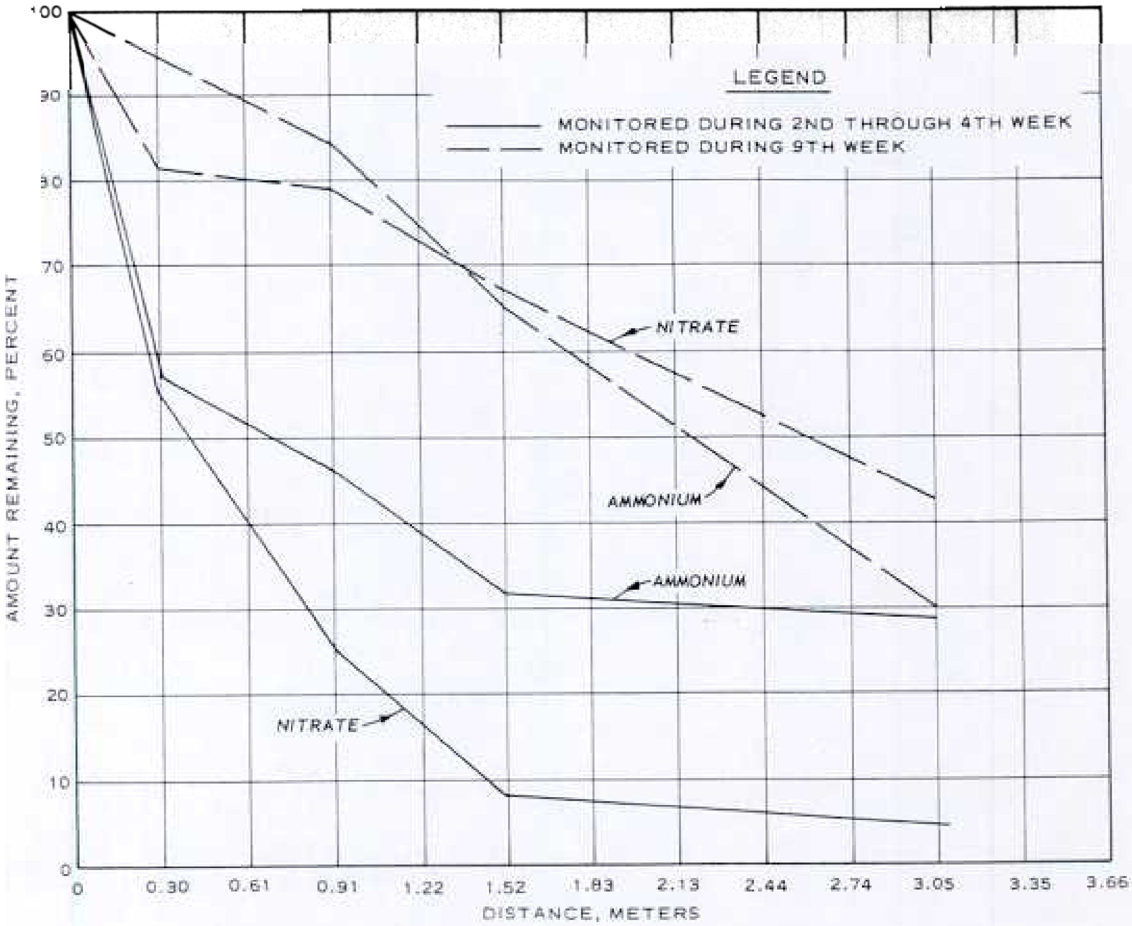

Figure 15. Nitrate and ammonium remaining in the surface liquid of the cannery wastewater model.

showed a slower rate of removal (Figure 15). However, nitrate was reduced by nearly 20 percent within the first 30 cm. Loss of volume was 48.8 percent as compared with a 55 percent average loss during the course of the earlier analyses, but this could not account for the drastic change in slope.

The rather dramatic reduction of nitrogen in the first 30 cm of the model was probably the result of accentuated microbial immobilization in this area. The amount of sucrose added to the wastewater during the experiment was an order of magnitude lower than the concentration prior to wastewater application, and much of this carbon probably was readily removed by the soil system, thus causing a severe nitrogen deficit near the source of the applied wastewater. This is indicated by influent COD values obtained during the initial 2 weeks (Table I-5). However, the amount of easily degradable carbon in the soil near the source probably had decreased significantly by the ninth week. This is reflected by a decrease in the initial rate of nitrate and ammonium removal and their resultant lower final concentrations. The question remains, however, whether or not the slope of the concentration decline would have approached zero at approximately 1 ppm. It is interesting that in Experiment II and another study with previously untreated Susquehanna clay that the concentration did go to zero within the limits of the experimentation. It could be that, in new systems with low levels of nitrogen, very high nitrogen removal is possible, whereas in older equilibrated systems, a concentration lower than 1 or 2 ppm total nitrogen could not be reached.

Effect of Flow Rate on Nitrogen Treatment

It was observed that treatment efficiency was quite responsive to changes in application and runoff flow rates, which were monitored continuously during the final 6 weeks of the experiment. This seemed intuitively reasonable since the liquid would remain in contact longer with reactive sites. In order to quantify this effect, a simple linear regression analysis was made for nitrate and ammonium concentration fluctuations in the runoff with changes in flow rate (Figures 16 and 17).

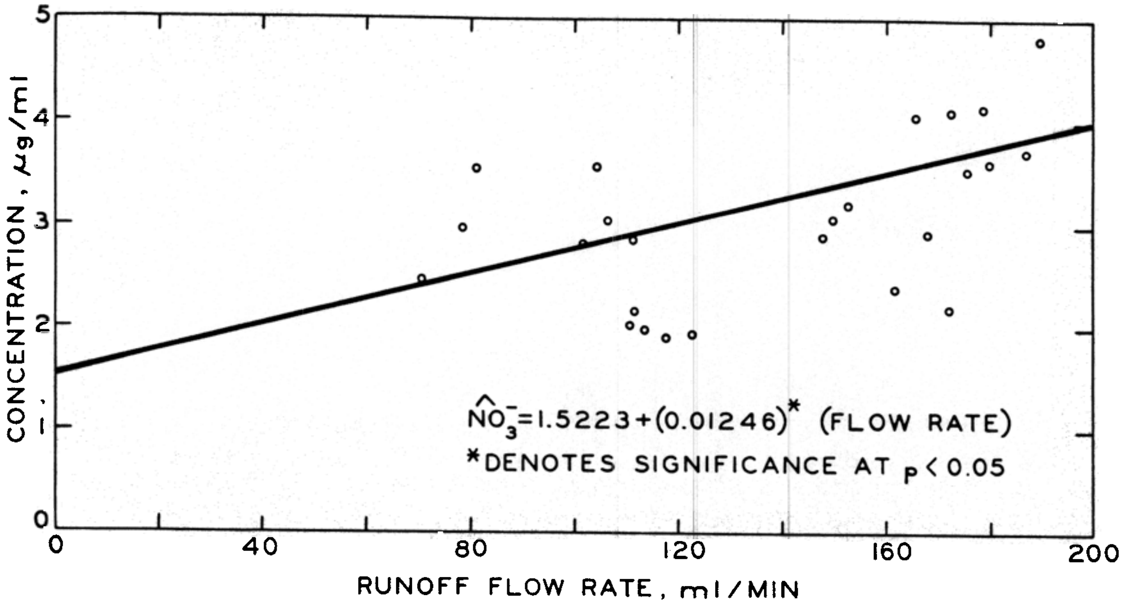

Figure 16. Simple linear regression analysis for nitrate in surface runoff.

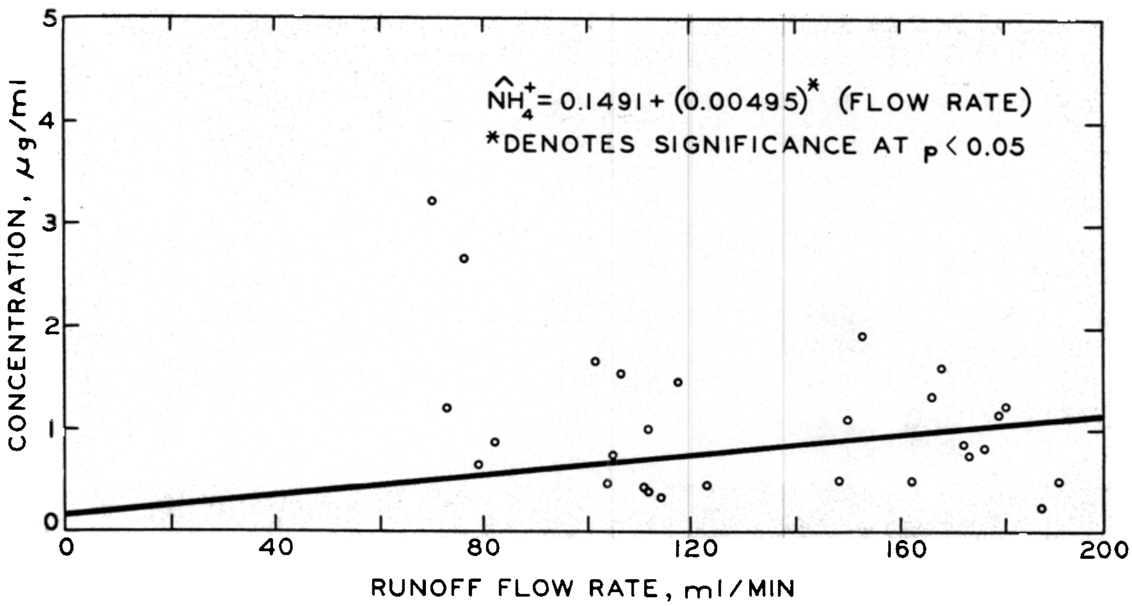

Figure 17. Simple linear regression analysis for ammonium in surface runoff.

Despite the observed fluctuation with time, a good correlation was found using 6 weeks of flow rate data (Table I-10). For each milliliter per minute increase in flow rate, nitrate was found to increase by 0.013 μg/ml and ammonium by 0.005 μg/ml in the runoff, using the simple linear regression formula. It was noticed that ammonium fluctuations were more time-dependent than flow-rate-dependent.

These findings are quite interesting and point out that many factors are involved in the performance of an overland flow system. For instance, the overland flow rate would vary with slope, and longer treatment areas or lower rates of application might be required for steeper slopes. These effects and interactions of slope and application rate are some of the more interesting phenomena that will have to be investigated in more detail in the future.

Account of COD and
Major and Minor Elements

The wastewater was fortified only with ammonium nitrate and sucrose. The sugar raised the COD of the effluent to an average value of 237.4 mg/ℓ, although values were as high as 630 during the first 2 days of experimentation (Table I-5). Average COD for the runoff was 116.6, which indicated a removal efficiency of 76.5 percent. This is a rather substantial reduction when the fact is considered that the treatment distance for the surface liquid was only 3 m. A similar system at Paris, Texas (36), reduced COD by about 92 percent (750 to 60 mg/ℓ) and BOD by nearly 99 percent within a 2 percent downslope distance of approximately 85 m. However, runoff COD values for the experimental model were only about twice those obtained for the field plots. Thus, substantial COD reduction might be realized within a very short initial overland flow distance.

Phosphorus was depleted from an average wastewater concentration of about 17 to 4.9 mg/ℓ in the runoff volume. This gave an average removal of 89.4 percent (Table I-11). This value is surprisingly high for an overland flow system, even with the more ideal conditions of the experimental model. More analyses are needed to confirm the initial data. The influent phosphorus amounted to approximately 55 g after the

12 weeks of application. Only 7.3 and 0.6 g of this were recovered in the runoff liquid and grass (Table 7), respectively. Therefore, the major part of the phosphorus appeared to be held by the soil. Calcium, magnesium, potassium, sodium, and manganese increased in concentration in the runoff. The higher runoff concentration for sodium was due to wastewater evaporation, but elution of the other elements from the soil system seemed to be the reason for their increase. The highly significant increase of manganese in runoff and subflow liquids could have been due to the low oxidation-reduction potential of the soil. Manganese is readily reduced to the more soluble manganous ion at potentials of less than +300 mv. Other heavy metals tested for were in trace amounts in both runoff and subflow. Significant increases of calcium and magnesium in grass tissues from the control half of the model, where the grass was allowed to grow to a greater height, probably reflect greater maturity of this grass resulting from smaller harvests. Both elements appeared to be readily available in the soil profile as was previously mentioned.

Analysis of Test Results from Experiment II

Nitrogen Treatment

The sludge-amended soil model in this experiment was studied to evaluate the effect of treatment of secondarily treated effluent on a new overland flow system with increased organic content. It was also used to investigate the compatibility of overland flow with digested sewage sludge disposal. The rationale for using a 1.52- by 3.05-m model in a greenhouse was the same as that discussed for Experiment I.

Daily concentrations of nitrate, ammonium, and TKN effluent; runoff; and subflow and liquid volume are presented in Appendix II, Tables II-1 - II-4. The percent nitrate removed from the wastewater after passage over 3.05 m of sludge-amended Susquehanna clay is presented in Figure 18 and Table 9. The statistical data of Experiment II were handled in the manner described for the previous experiment and are considered nonsignificant at the 0.1 level. The runoff liquid had a highly significant increase in percent nitrate nitrogen removal, ranging from 50 to

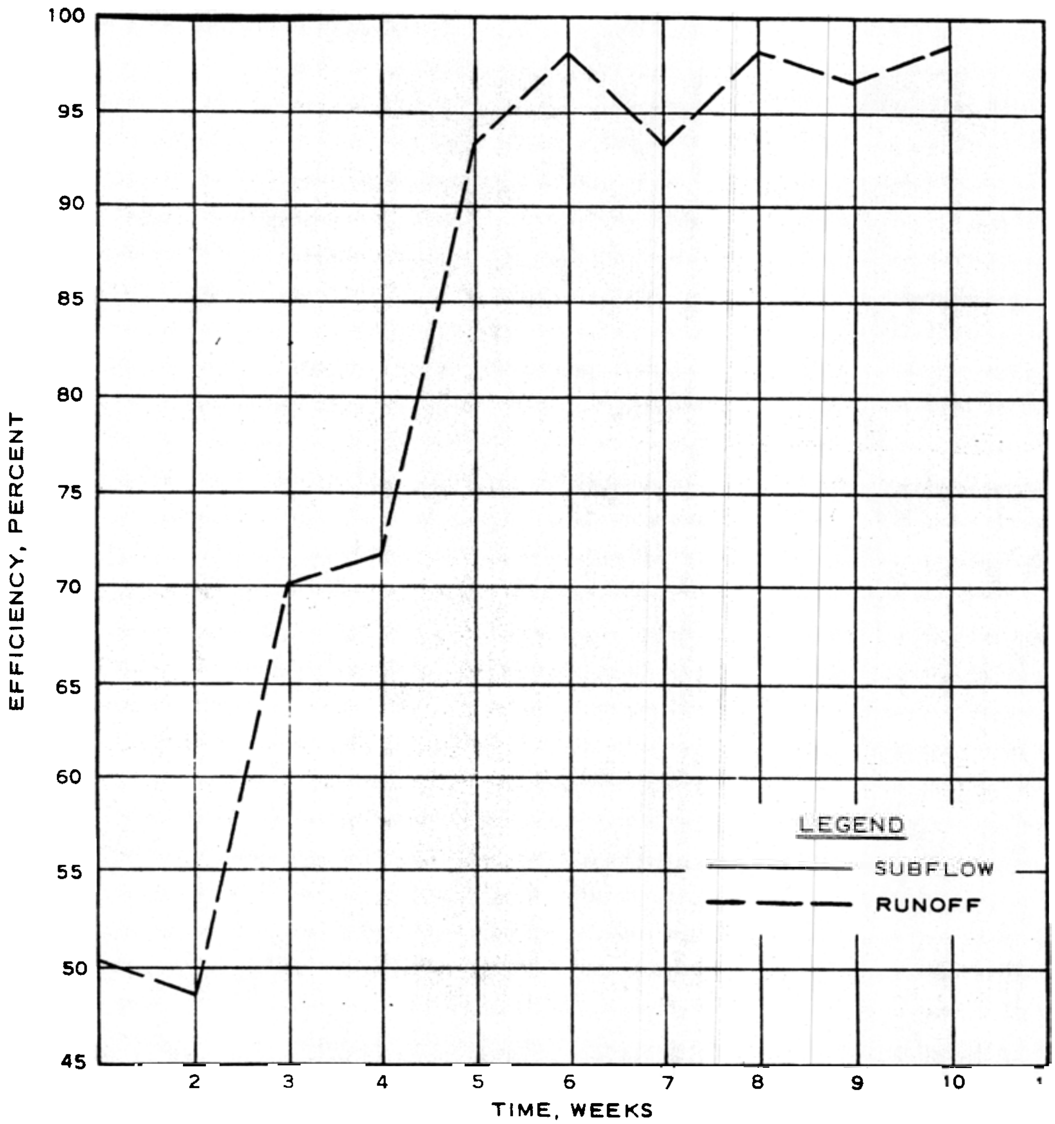

Figure 18. Efficiency on nitrate removal by the sludge-amended Susquehanna clay model.

Table 9

Nitrate Nitrogen Removal Efficiencies From Runoff and Subflow of the Sludge-Amended Susquehanna Model

Liquid	Day	Efficiency, percent, for Cited Week 1	2	3	4	5	6	7	8	9	10
Runoff	Mon			73.32	71.10	98.31	98.19	92.49	100.00	100.00	99.79
	Tue			61.80	69.70	91.12	98.59	87.90	98.54	98.86	99.28
	Wed		52.15	68.73	79.13	88.12	98.10	95.38	97.18	94.83	93.75
	Thu		45.46	76.55	68.10	95.99	97.64	96.73	96.93	92.99	100.00
	Fri	50.61									
	Mean**	50.61	48.81	70.10	71.76	93.39	98.13	93.13	98.16	96.67	98.21
Subflow	Mon			99.80	100.00	100.00	100.00	100.00	100.00	100.00	100.00
	Tue			100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00
	Wed		99.55	99.72	100.00	100.00	100.00	100.00	100.00	100.00	100.00
	Thu		100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00
	Fri	100.00									
	Mean[NS]	100.00	99.78	99.88	100.00	100.00	100.00	100.00	100.00	100.00	100.00

** Highly significant difference at the 0.01 level using the F test.
NS Not significantly different at the 0.1 level using the F test.

93 percent during the first through fourth weeks, while no significant changes were noted during the remainder of the experiment. The subflow contained essentially no detectable nitrate nitrogen during the entire experiment. Ammonium removal followed a similar but less dramatic highly significant change during the first 3 weeks of the experiment (Table 10). However, except for this short period, no detectable ammonium was observed in the runoff or subflow samples. TKN removal data are presented in Table 11; unfortunately, they are not as extensive as the nitrate and ammonium data. For the TKN removal in runoff, a significant effect of time exists, with the removal percentage declining in weeks 7 and 8. TKN percent removal from subflow through the first 5 weeks was nearly complete, and thus further analyses were deleted. Since the TKN analyses used in this study did not measure the nitrate, the summation of nitrogen removed as nitrate and TKN is quite meaningful. These data are presented in Table 12. Although there was a significant increase in treatment percent removal during the first 4 weeks of the experiment, the increase was related to percent nitrate removal, and the total removal was more constant after that time. This effect of total percent removal being more uniform than TKN, NO_3, or NH_4 removal, was even more pronounced than in Experiment I. Average combined nitrogen values for the experiment indicated that 18.1 percent nitrate and 10.6 percent TKN (3.7 ammonium and 6.9 organic nitrogen) had remained in the runoff. However, only 2 percent nitrate, 0 percent ammonium, and 8.8 percent organic nitrogen were found in the runoff during the last week of the experiment. The lag in nitrogen removal efficiency could have resulted from a number of factors. It could have been a simple biological response, dependent on the time necessary for both plants and microbes to become active. This hypothesis is supported to some extent by the fact that, in a parallel study with Susquehanna clay in a 6.10-m-long model, a similar lag in treatment efficiency was observed (16). The lag could also have been caused or intensified by channeling on the model surface during the first 2 weeks. This hypothesis is supported by the fact that one side of the model received most of the liquid during this period and removed significantly less nitrogen than did the other side. If the

Table 10

Ammonium Nitrogen Removal Efficiencies From Runoff and Subflow of the Sludge-Amended Susquehanna Clay Model

Liquid	Day	Efficiency, percent, for Cited Week									
		1	2	3	4	5	6	7	8	9	10
Runoff	Mon		89.51	91.91	100.00	100.00	100.00	100.00	100.00	100.00	100.00
	Tue		89.67	92.06	100.00	100.00	100.00	100.00	100.00	100.00	100.00
	Wed		84.35	95.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00
	Thu		76.81	100.00	98.63	100.00	100.00	100.00	100.00	100.00	100.00
	Fri	83.93									
	Mean**	83.93	85.06	94.74	99.66	100.00	100.00	100.00	100.00	100.00	100.00
Subflow	Mon		100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00
	Tue		100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00
	Wed		100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00
	Thu		100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00
	Fri	100.00									
		100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00

** Highly significant difference at the 0.01 level using the F test.
NS Not significantly different at the 0.1 level using the F test.

Table 11

TKN Removal Efficiencies From Runoff and Subflow of the Sludge-Amended Susquehanna Clay Model

Liquid	Day	Efficiency, percent, for Cited Week									
		1	2	3	4	5	6	7	8	9	10
Runoff	Mon			89.59					93.44	96.09	
	Tue			90.78	87.85	95.86			79.11	93.72	
	Wed							85.66	75.43	89.41	
	Thu				94.38				82.35	85.55	
	Fri										
	Mean**			90.19	91.12	95.86		85.66	82.58	91.19	
Subflow	Mon			99.53							
	Tue			100.00	100.00	100.00					
	Wed		99.20								
	Thu				99.83						
	Fri										
	Mean NS		99.20	99.76	99.92	100.0(					

** Significantly different at the 0.1 level using the F test.
NS Not significantly different at the 0.1 level using the F test.

Table 12

Nitrate Plus TKN Removal Efficiencies From Runoff and Subflow of the Sludge-Amended Susquehanna Clay Model

Liquid	Day	Efficiency, percent, for Cited Week									
		1	2	3	4	5	6	7	8	9	10
Runoff	Mon			79.91	60.85				95.76	97.09	
	Tue			76.15	81.66	94.00			87.54	95.00	
	Wed							90.27	85.98	91.54	
	Thu				80.88				89.51	89.27	
	Fri										
	Mean**			78.03	81.27	94.00		90.27	89.70	93.23	
Subflow	Mon			99.69	100.00						
	Tue			100.00	100.00	100.00					
	Wed		99.38								
	Thu				99.93						
	Fri										
	Mean NS		99.38	99.85	99.98	100.00					

** Highly significant difference at the 0.01 level using the F test.
NS Not significantly different at the 0.1 level using the F test.

surface reaction of the model was similar to that of a trickling filter and was mass-transfer-limiting, the channeling and concomitant reduced surface contact would have lowered the treatment efficiency significantly. The surface of the model was leveled during the second week of treatment, increasing both surface contact and treatment efficiency.

However, the most surprising observation was that the treatment efficiency was very high after only 5 weeks. It was originally feared that the ammonium and nitrate removals might never be high on the relatively short (3.05-m) slope. It can also be seen in Figure 19 and in Table II-4 that the runoff varied from greater than two-thirds to greater than one-third the applied volume and that the subflow never exceeded one-sixth the applied volume. Extensive upwelling was not occurring, for the model was dammed at 1.52 m and no downslope surface discharge

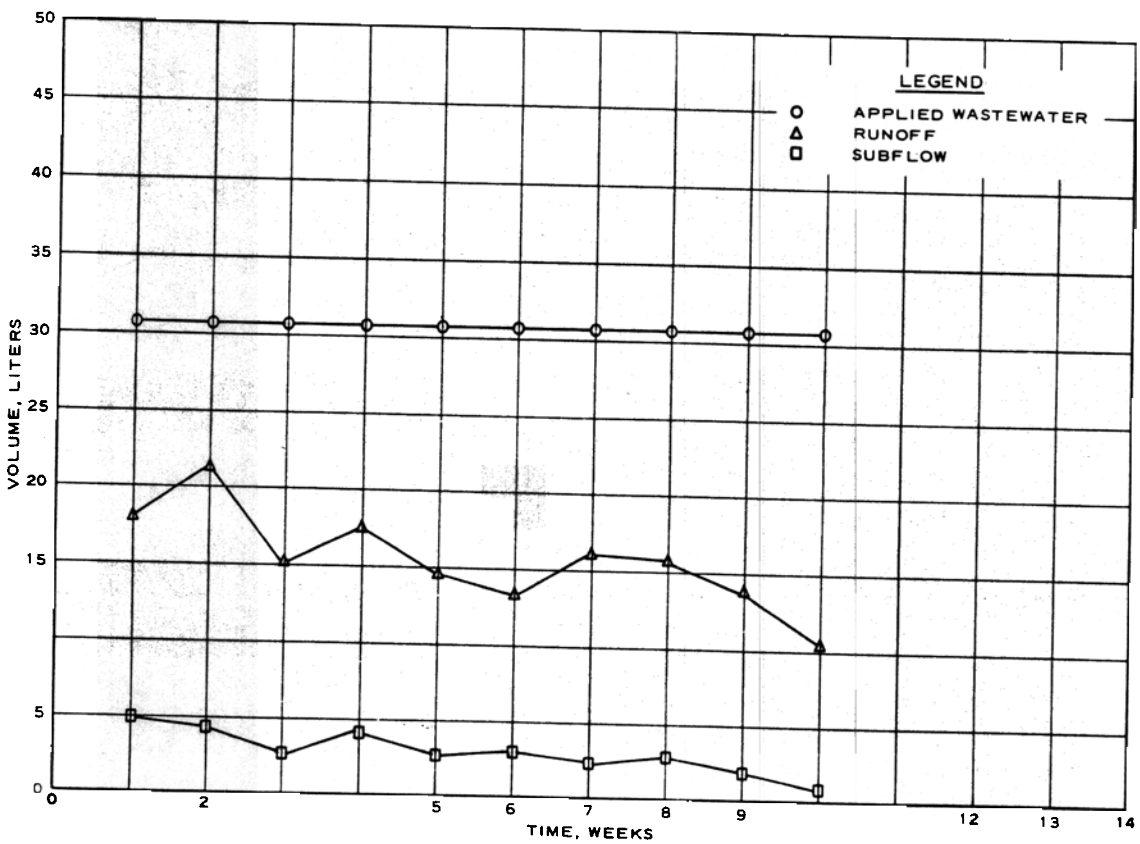

Figure 19. Daily liquid volumes for the sludge-amended model.

occurred. The same was found to be true in an experiment with untreated Susquehanna clay. These results indicate that the treatment was truly a surface phenomenon.

Another interesting point was that the amendment of sludge into the surface of the Susquehanna clay did not significantly change its treatment efficiency from that of an unamended Susquehanna clay in a model that was treated in a very similar manner (16). The unamended clay also showed nearly 100 percent removal of nitrate at 6.10 m after 3 weeks, and after the fourth week nitrate and ammonium were essentially removed after passage over 3.05 m of surface.

A nitrogen balance sheet is presented in Table 13. An amount equivalent to 30 percent of the nitrogen applied was removed by the

Table 13

Nitrogen Balance Sheet for the Susquehanna Clay Sludge-Amended Model Irrigated With Secondary Effluent for 10 Weeks

Component	Percent of Total Applied Nitrogen
Runoff N	12
Grass N	30
Soil N	34
Unaccounted for N	24

plants, 34 percent remained in the soil, 12 percent remained in runoff, and 24 percent was unaccounted for. It is hypothesized that the 24 percent unaccounted for was lost by denitrification. Unfortunately, this hypothesis cannot be quantitatively supported since N^{15} was not used and the TKN measurements of soil nitrogen are not accurate enough to detect small changes in soil nitrogen percentages. However, the oxidation-reduction measurements made over a 5-day period that encompassed a wetting-drying cycle strongly support the hypothesis that significant denitrification was occurring (Figure 20; Table II-5). The horizontal line at +320 mv represents the potential at

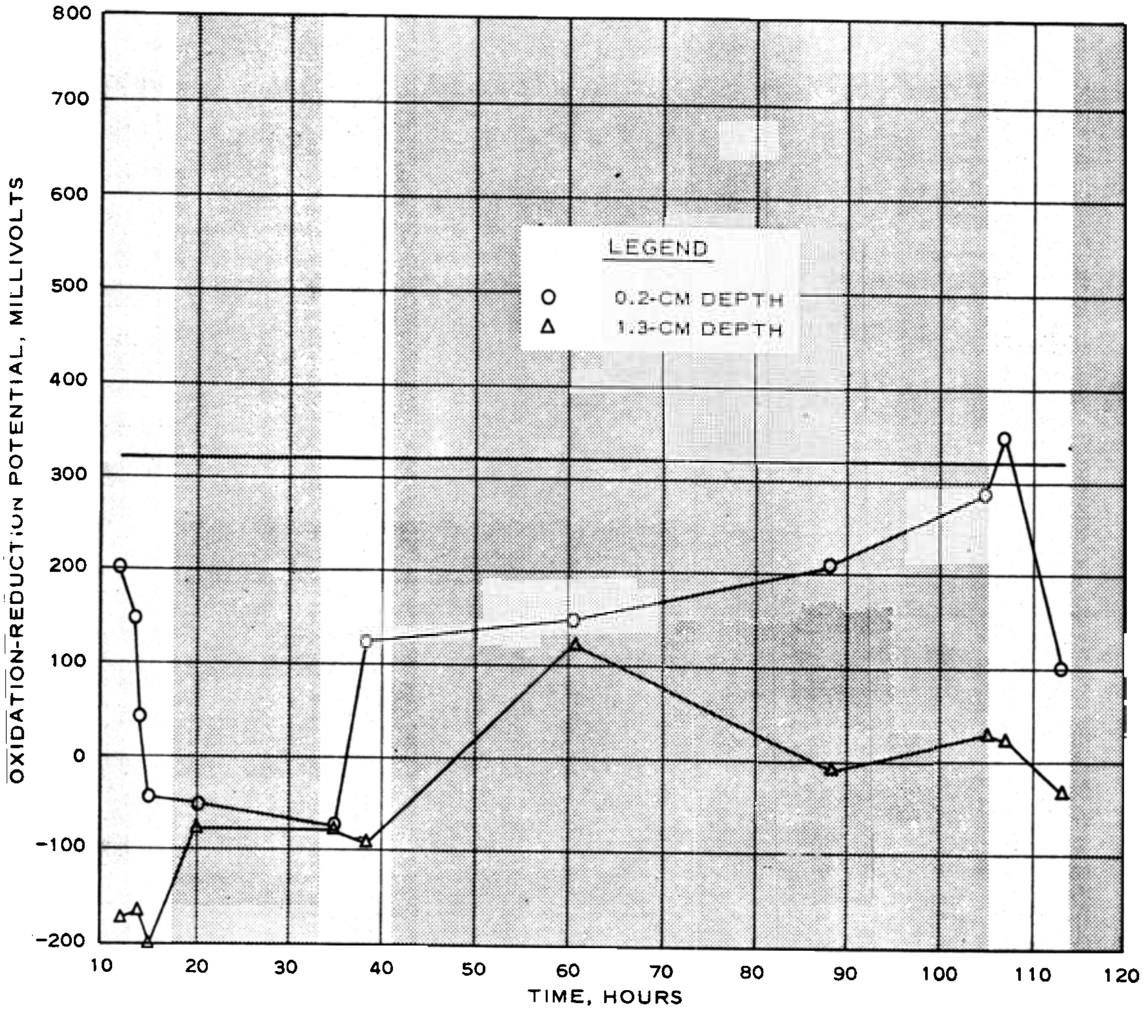

Figure 20. Adjusted oxidation-reduction potentials within the sludge amended model during a wastewater application and drying cycle. Stippled areas represent the rest periods.

which nitrate instability occurs at a pH of 5 and a temperature of 25 C (49). As can be seen, the Eh at the 1.3-cm depth did not reach +320 mv even after a weekend of drying. Another point of significant interest is that the Eh of the soil at depths of only 3 to 5 mm was in the denitrifying range during the wetting period and in the nitrifying range during the weekend drying period. The Eh of the 3- to 5-mm surface layer declined into the denitrifying range within only a few hours after wastewater application, even after a weekend drying period. The accuracy of this redox curve is not only supported by similar results in Experiment I (Figure 14), but also by the previously mentioned parallel study (16).

In hypothesizing the functioning of this model, it should be remembered that it was operated 4 days and allowed to dry for 3 days during most of the experiment. Under these conditions, significant nitrification could occur within the system. The nitrate formed during this period would have been subjected to denitrification when wastewater was reapplied or when it diffused downward to the anaerobic zone during the drying period. This mechanism could also account for the loss of nitrogen from the applied sludge, without its noticeable appearance in the runoff or subflow. These hypotheses seem even more reasonable when the facts are considered that denitrification was also thought to be a major mechanism in the loss of nitrogen from cannery wastewaters at the Campbell Soup Company's Paris, Texas, plant (36) and that this system is similar to a rice paddy system. As shown in Figure 9, a rice paddy normally has an "aerobic-anaerobic double layer" that functions to the detriment of a rice farmer by converting ammonium to nitrate in the aerobic layer and denitrifying it in the anaerobic layer (50). It is hypothesized that the overland flow system would be even more efficient in eliciting the gaseous loss of nitrogen than a rice paddy system because most rice fields are kept flooded rather than cycled through flooding and drying periods. As a result, nitrate is only formed in a thin, surficial aerobic zone, and its formation rate is dependent on the diffusion of other forms of nitrogen from the lower soil profile. In the cycled overland flow system, the surface aerobic layer would be thickened during the dry periods, and the larger quantity of nitrate formed at this time would then be subject to denitrification upon wastewater application.

In order to better understand the mode of disappearance of the combined nitrogen, the concentrations of nitrate and ammonium in overland flow were monitored down the length of the model (Table 14). Nitrate failed to show the initial rapid decrease that was observed for the more mature cannery wastewater model of Experiment I. The most

Table 14

Removal of Major Nutrients from Wastewater With Distance Along Susquehanna Clay Amended With Sewage Sludge After 10 Weeks

Liquid	Distance, m	Concentration, ppm, of Cited Nutrient			
		NO_3	NH_3	K	P
Wastewater	0.00	8.8	7.2	6.23	31.10
Surface Liquid	0.76	8.6**	3.9**	4.24**	28.59
	1.52	4.65	.47	1.92	25.66
	2.29	1.15	0.0	0.56	25.92
	3.05	1.05	0.0	0.13	21.60
Subflow	0.76	0.0 NS	0.0 NS	0.56	0.0
	1.52	0.0	0.0	0.26	0.0
	2.29	0.0	0.0	0.26	0.0
	3.05	0.0	0.0	0.28	0.0

** Significantly different at 0.01 level using F test for values in respective columns.

NS Not significantly different at 0.1 level using F test for values in respective columns.

rapid nitrate loss was observed at about half the downslope distance, near where luxuriant grass ceased to grow. Thus, a newly developed system on soil deficient in nitrogen may show much greater potential than one that has reached a nitrogen equilibrium. The rapid nitrate loss also indicates that different processes may be involved in nitrogen loss from a new versus an old system. Ammonium loss followed a pattern similar to that observed in Experiment I.

The results of this study show that, within a relatively short time period, a very high degree of nitrogen treatment was achieved by overland flow on a clay soil that had been amended with digested sewage sludge. However, the establishment of the mechanisms involved have not yet been proven conclusively, and the development of criteria for design and operation of such systems must await further study.

Macroelement Treatment

The removal efficiencies of phosphorus, potassium, calcium, magnesium, and sodium in overland flow of the wastewater are presented in Table 15; daily concentrations of these elements are presented in Tables II-6 - II-10. Due to man power limitations, these elements

Table 15

Removal Efficiency of Macroelements From Wastewater on Susquehanna Clay Amended With Sewage Sludge

Element	Removal Efficiency, percent, for Cited Week						
	5	6	7	8	9	10	Mean
P				47.7	91.1		69.4
K	97.8	98.5	98.2	91.5	94.5		96.1
Ca	72.7	73.7	80.8	67.0	83.8		75.6
Mg	75.1	68.3	59.6	-26.0	2.3		35.8
Na	67.4	46.9	45.6	39.5	51.8		50.2

were not sampled during the first 4 weeks of the study, but were sampled routinely after this time. Although the phosphorus in the runoff varied considerably and was not measured extensively, the mean removal of 69 percent is in moderate agreement with that of the previously mentioned parallel study where 67 and 33 percent of the phosphorus remained in the runoff after passage over 3 and 6 m of Susquehanna clay, respectively. This low treatment for phosphorus is still surprising since the Susquehanna clay had a pH of 5 and high iron and aluminum contents (43). It is hypothesized that this low removal was a result of poor contact of the liquid with the compacted clay soil and the higher solubility of phosphorus under reduced conditions (49). Additionally, the microbial cell mass in the upper few millimeters of soil was perhaps so intense that it may have limited the contact of liquid with reactive zones on or within the soil. It has already been pointed out that reducing conditions existed in the soil profile at depths of 0.5 cm under flooded conditions and at depths greater than 1.3 cm most of the time. Numerous studies have confirmed the increased solubility of phosphorus under reduced as opposed to oxidized conditions (50,51,53). One of the primary means of phosphorus release under reduced conditions is associated with the breakdown of ferric oxide-ferric phosphate coatings of clay colloids (49,68). Thus, phosphorus is not only less effectively removed by the soil, but is also released from insoluble forms within the soil under reduced conditions.

The authors believe that phosphorus removal may well be the limiting factor in an overland flow treatment system. However, phosphorus removal with distance down the length of the model (Table 14) was basically linear in this and the parallel study, giving support to the hypothesis that increased slope length could raise the phosphorus removal efficiency to greater than 99 percent. This is, of course, an extrapolation beyond the limits of the present data. Thomas[d] claims that soil treatment systems may require up to 4 months before phosphorus removal efficiency stabilizes. With the use of primary municipal wastewater, he found this to be around 50 percent (220-440 kg/ha/yr).

Treatment might be increased by combining a shallow, rapid infiltration system with the overland flow system. The shallow, rapid infiltration bed could greatly reduce the phosphorus concentration and convert the nitrogen to nitrate, while the overland flow could polish the nitrogen and heavy-metal removal. This combination might be particularly good where small volumes are involved.

Potassium was removed quite extensively (96 percent) throughout

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[d] Personal communication, R. E. Thomas, EPA, Water Quality Control Branch, Ada, Okla.

the 5-week period, and it was primarily removed on the upper half of the model (Table 14). The removal was probably an exchange phenomenon, with a considerable amount going into plant tissue.

Sodium, an element similar to potassium, was only 50 percent removed (Table 15). However, the concentration of sodium was always an order of magnitude higher than either potassium or the combined concentrations of calcium and magnesium. Calcium was moderately removed (75 percent), but magnesium showed poor removal (35 percent) (Table 15). The apparently poorer magnesium removal from the wastewater may have resulted from the leaching of magnesium from the sewage sludge, which initially contained 2380 and 660 ppm magnesium and calcium, respectively. However, removal of both probably resulted from mass-action displacement by the more concentrated sodium (55). A prolonged application of a liquid with a sodium to calcium plus magnesium ratio as high as these effluents had would most likely result in both soil structure and plant osmotic damage problems (12,27). The high sodium content of this wastewater probably resulted from the fact that restaurant waste constituted a high percentage of the wastewater. Concentration fluctuations down the length of the model for calcium, magnesium, and sodium are given in Table 16. The results indicate that high evapotranspiration may accentuate the sodium concentration problem.

Minor Element Treatment

Using 5 weeks of data, removal percentage means for manganese, copper, lead, and cadmium were greater than 90 percent, while those for zinc and nickel were 75 and 83 percent, respectively (Table 17). Daily concentrations of these elements are presented in Tables II-1 - II-16. During the downslope monitoring of heavy-metal concentrations in the overland flow, manganese, lead, nickel, and cadmium were greatly reduced within the first quarter of this distance (Table 18). At this time, though, copper, as well as zinc, was not greatly reduced in this same distance, and zinc even appeared in much of the subflow.

The grass was also analyzed for the amended elements. Table 19 gives the removal percentages of various macro- and microelements by the grass, based on both the total amount of each applied in the wastewater

Table 16

Removal of Macroelements With Distance Along Susquehanna Clay Amended With Sewage Sludge and Irrigated With Secondary Effluent for 10 Weeks

Liquid	Distance, m	Concentration, ppm, of Cited Element Ca	Mg	Na
Wastewater	0.00	3.85	1.56	87.5
Surface Liquid	0.76	2.60**	1.54^{NS}	86.3
	1.52	2.15	1.53	94.05
	2.29	1.70	1.54	99.75
	3.05	1.63		87.5
Subflow	0.76	1.80		5.65
	1.52	1.40	1.87	3.05
	2.29	1.26	1.28	2.45
	3.05			

** Significantly different at the 0.01 level using the F test.
NS Not significantly different at the 0.1 level using the F test.

Table 17

Removal Efficiency of Minor Elements From Wastewater on Susquehanna Clay Amended With Sewage Sludge

Element	Removal Efficiency, percent, for Cited Week 5	6	7	8	9	Mean
Zn	80.26	74.85	74.41	[illegible]	77.86	74.61
Mn	90.82	96.79	96.24	[illegible]	96.95	95.47
Cu	89.17	91.29	100.00	[illegible]	96.32	92.36
Ni	62.87	75.61	100.00	[illegible]	89.70	83.24
Pb	95.06	90.45	79.14	[illegible]	89.70	90.32
Cd	96.06	97.46	89.71	[illegible]	96.97	95.45

Table 18

Removal of Minor Elements With Distance Along Susquehanna Clay Amended With Sewage Sludge and Irrigated With Secondary Effluent for 10 Weeks

Liquid	Distance, m	Concentration, ppm, of Cited Element					
		Zn	Mn	Cu	Ni	Pb	Cd
Wastewater		0.20	0.23	0.19	0.15	0.19	0.22
Surface Liquid	0.76	0.05NS	0.0**	.08**	.10NS	0.0**	.04**
	1.52	0.06	0.0	0.07	0.0	0.02	0.0
	2.29	0.05	0.0	0.06	0.04	0.01	0.0
	3.05	0.07	0.0	0.03	0.03	0.04	0.0
Subflow	0.76	0.14	0.0	0.0	0.04	0.03	0.0
	1.52	0.10	0.0	0.01	0.0	0.02	0.0
	2.29	0.06	0.0	0.01	0.0	0.03	0.0
	3.05	0.0	0.0	0.0	0.0	0.0	0.0

** Significantly different at the 0.01 level using the F test.
NS Not significantly different at the 0.1 level using the F test.

Table 19

Removal Percentage of Major and Minor Elements by Grass on Susquehanna Clay Amended With Sewage Sludge and Irrigated With Secondary Effluent for 10 Weeks

Element	Removal, percent		Element	Removal, percent	
	Of Total Applied	Of Total Remaining in Model		Of Total Applied	Of Total Remaining in Model
P	7.27	10.17	Zn	18.80	26.53
K	66.95	69.26	Mn	10.37	11.11
Ca	14.36	22.18	Cu	3.88	6.83
Mg	25.36	58.12	Ni	0.98	1.12
Na	3.06	6.29	Cd	0.51	0.54

as well as on the total retained by the model, excluding runoff loss. Table 20 shows how these elements varied in concentration in grass collected from even downslope quadrants. In agreement with the overland flow values, the high concentration of zinc in plant material did not vary down the slope. The zinc was most likely being supplied by the sludge rather than the wastewater. Additionally, in the parallel study on unamended Susquehanna clay, the zinc concentration in plants varied between the much lower values of 75 to 150 ppm.

The appearance of copper in wastewater at the 1.5- and 2.3-m distances at this particular sampling time was probably due to

Table 20

Nutrients in Grass Harvested From Susquehanna Clay Amended With Sewage Sludge and Irrigated With Secondary Effluent for 10 Weeks

Component	Concentrations, percent, for Various Downslope Distances, m				
	0.76	1.52	2.29	3.05	Mean
TKN††	2.115	1.99	1.695	1.36	1.79
P[NS]	0.44	0.385	0.345	0.345	0.3785
K**	1.46	1.145	1.115	0.825	1.13625
Ca[NS]	0.1818	0.2041	0.28325	0.28185	0.23775
Mg[NS]	0.41	0.382	0.3785	0.3665	0.38425
Na[NS]	0.935	0.87	0.71	0.755	0.8175
Zn[NS]	225.5	200.5	206	205.5	209.37
Mn[NS]	74.55	64.45	59.2	72	67.55
Cu[NS]	17.45	13.9	17.0	10.1	14.61
Ni[NS]	3.6	0.85	1.3	0.95	1.675
Cd*	7.05	0.8	2.85	0	2.675
	Kg/ha				
Yield**	2505	1235	1121	1066	1482

** Significantly different at the 0.01 level using the F test.
†† Significantly different at the 0.10 level using the F test.
NS Not significantly different at the 0.10 level using the F test

experimental variation, for normally 92 percent removal was achieved. In addition, the copper concentration of the sludge was not extremely high (212 ppm). Manganese was removed more efficiently than expected, with 100 percent removal in the initial quarter of the overland flow distance. However, a considerable amount of manganese was incorporated into the plants down the slope of the model, averaging 67 ppm. This reflected the moderately high (304-ppm) concentration of manganese in the sludge. Cadmium also showed a significantly higher uptake by the grass, probably reflecting the 100 percent removal efficiency obtained for surface liquid within the upper half of the model. Total cadmium removal was 95 percent. On the other hand, relatively low nickel removal was reflected in the low concentrations found in the plant tissue.

Removal of nutrients by the plants resulted in the recycling of the elements, a development which could be either beneficial or detrimental under existing conditions. The heavy metals present in the harvested grass, ranked from highest to lowest with respect to concentrations, were zinc, manganese, copper, nickel, and cadmium. The zinc and manganese removals by the grass were quite significant, whereas the removals of copper, nickel, and cadmium were so small in comparison to the percentages retained by the soil that their recycling was rather inconsequential.

The soil was analyzed for various nutrients and heavy metals both before initiation of the experiment and at five evenly distributed downslope sites after culmination of experimentation (Table 21). Following the 10-week application of amended wastewater, no significant increases of heavy metals in the soil were noted. In addition, the concentrations did not vary significantly with location on the slope. The plant response to heavy metals was measured more accurately and was more obvious in this study. However, on a long-term basis, considerable amounts of heavy metals may be immobilized in the soil without the metals becoming significantly more available to plants (38). Although it must be recognized that, after the ability of a soil to immobilize heavy metals has been exceeded, heavy metals may well be available at toxic levels to plants (38).

Table 21

Major Nutrients of Susquehanna Clay Amended With Sewage Sludge and Irrigated With Secondary Effluent for 10 Weeks

Component	Concentration, percent, at Various Downslope Distances, m						
	Original	0.03	0.76	1.52	2.29	3.05	Mean
NH^{NS}	0.005002	0.00107	0.00053	0.00054	0.000725	0.000415	0.000656
TKN^{NS}	0.100577	0.07363	0.06458	0.06123	0.06250	0.05825	0.06404
K^{NS}	0.013272	0.007150	0.007700	0.008500	0.007950	0.008300	0.007920
Ca^{NS}	0.10222	0.10000	0.07090	0.1590	0.01580	0.01600	0.04372
Mg^{NS}	0.05541	0.05035	0.04925	0.05240	0.05285	0.05205	0.05138
Na**	0.00108	0.08480	0.03885	0.03950	0.05040	0.05160	0.05303
Zn^{NS}	0.00442	0.00185	0.00135	0.00335	0.00240	0.00285	0.00236
Mn^{NS}	0.0093	0.00115	0.00095	0.00085	0.00105	0.00080	0.00096
Cu^{NS}	0.00127	0.00010	0.00010	0.00010	0.00010	0.00010	0.00010
Ni^{NS}	0.0	0.00010	0.00010	0.00010	0.00010	0.00010	0.00010
Pb^{NS}	0.00118	0.00010	0.00010	0.00010	0.00010	0.00010	0.00010
Cd^{NS}	0.0	0.00010	0.00010	0.00010	0.00010	0.00010	0.00010

** Significantly different at the 0.01 level using the F test.
NS Not significantly different at the 0.10 level using the F test.

CONCLUSIONS AND RECOMMENDATIONS

Conclusions

1. Overland flow is an effective means of removing nitrogen from secondarily treated wastewater (pp 30-37, 57-64).

2. Algal crust probably hinders rather than promotes nitrogen removal (pp 31, 41-49).

3. Reed canary grass is capable of removing considerable quantities of nitrogen from an overland flow system (pp 39-40).

4. Nitrification and denitrification in an overland flow system appear to function in a manner similar to that observed in rice fields. Nitrification occurs in a surface "oxidized" layer, and denitrification occurs in the underlying "reduced" layer of an "oxidized-reduced double layer" in the upper soil profile (pp 49-51, 65-68).

5. The thickness of the soil aerobic zone rapidly fluctuates in response to flooding, and diminishes with duration of flooding and organic load in the applied water (pp 47-51, 65-67).

6. Anaerobic conditions seem to be prevalent at greater than a few millimeters soil depth during flooding of overland flow systems, but aerobic conditions can be attained after a 2-day rest period (pp 49-51, 65-67).

7. Accumulation of easily degraded organic material on the soil surface may cause the oxidized layer to be at or in the surface liquid above the soil, which would result in the need for a rest period (pp 41-46).

8. Generally, nitrogen removal should be higher if the nitrogen is in the nitrate form, due to the prevalence of reduced conditions in an overland flow system (pp 47-51, 65-67).

9. Plant species that can tolerate reduced soil conditions must be used in overland flow systems. Reed canary grass functions quite well under these conditions (p 51).

    Contact of the wastewater with the soil and its associated large microbial population may be essential to proper system functioning (p 45).

    Proper functioning of an overland flow system seems to require free diffusion between the oxidized and reduced zones in the soil (pp 41-46).

12. Nitrogen treatment efficiency is highly related to the rate of application and slope length (pp 52-56, 68-69).

A new overland flow system may require a 4- to 5-week period before nitrogen removal efficiencies approach maximum values (pp 64-65).

14. Sewage sludge incorporated into the upper few centimeters of the soil on a 6 percent by weight basis does not greatly help or hinder advanced treatment of wastewater by overland flow (pp 64-65, 67, 69).

Heavy metals in the wastewater appear to be removed quite effectively within the first meter of downslope distance (pp 71-73).

16. Phosphorus removal appears to be the short-term limiting factor for overland flow (pp 69-70).

17. Models 3.05 and 6.10 m long are effective in studying overland flow, especially with respect to nitrogen treatment (pp 20, 30-31, 52).

Recommendations*

1. In areas that have soils of low permeability, advanced treatment of wastewater by overland flow should be considered.

2. Slopes of an overland flow system should be made as smooth as possible so that the thinnest and most uniform flow of water will occur. Gullying and ponding should be prevented to the maximum degree practicable.

3. An overland flow system 50 to 100 m in length and at a 2-8 percent slope should give good nitrogen treatment of wastewater.

4. A daily application of 1 to 2 cm of wastewater over a 5- to 6-hr period should give an adequate residence time at the given length and slope of treatment area.

5. Overland flow length should increase with an increase in slope or rate of application to allow sufficient residence time on the soil for treatment.

6. A rest period of 2 days per week may or may not be necessary or adequate, depending upon the quality of the wastewater, the weather, and the condition of treatment slopes. Runoff monitoring should provide the clue as to when the system is in need of a rest period.

7. Vegetation should be selected that can tolerate semiflooded surface conditions and nearly continuous reduced conditions below a few centimeters depth. Reed canary grass, rice, and freshwater sedges are examples of such vegetation.

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* These interim recommendations are based on both the literature review and experimental results of this study.

8. At this time, overland flow is not recommended for forested areas.

9. Water-quality monitoring stations should be installed in areas where the treated water from several slopes enters a natural stream or lake.

10. Phosphorus treatment efficiency should be checked carefully, and the application rates should be adjusted to maintain the desired efficiency.

11. A method of application should be used which provides for uniform distribution.

12. Slopes should be sprayed at night as well as during the day to minimize peak pumping.

13. Slopes should be cycled so that wastewater quality and treatment condition can average out over the treatment area.

Operation and treatment information should be logged so that system design can be refined.

Research on overland flow for the advanced treatment of wastewater should be continued to develop a clearer understanding of the principles involved, to determine the effectiveness of the processes under varying conditions, and to improve criteria for design and operation of facilities.

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APPENDIX I

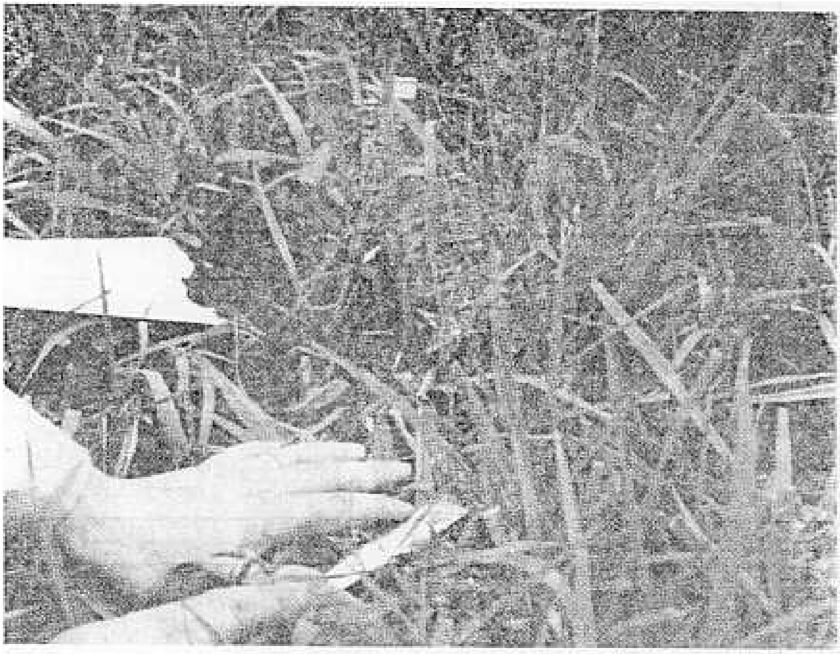
Data From the

Cannery Wastewater Model

A. Soil surface without algal growth in high grass on overland flow model

B. Algal mat on cut-grass side of overland flow model

Figure I-1



C. Algal layer covering soil on model with grass cut

D. View of the cut and uncut sides of the model

Figure I-1

Table I-1

Daily Concentration of Nitrate on the Cannery Wastewater Model

Liquid	Day	Concentration of NO_3, ppm, for Cited Week												
		1	2	3	4	5	6	7	8	9	10	11	12	Mean
Wastewater	Mon		11.25	7.40	7.62	6.10	5.50	7.25	8.60	9.80	8.40	7.45	10.20	8.14
	Tue		12.00	9.07	6.30	8.10	10.10	8.60	8.85	11.35	9.30	10.20	13.15	9.73
	Wed		12.00	8.00	11.00	8.60	8.90	10.10	10.05	9.60	9.35	12.80	9.55	10.00
	Thu	11.50	10.70	7.3	6.80	9.15	15.25	10.00	9.70	10.50	11.75	12.30	9.15	10.34
	Fri	12.50	10.90	7.45	6.40			10.10	10.20	8.60	17.00	9.40		10.28
	Mean	12.00	11.37	7.84	7.62	7.99	9.94	9.21	9.48	9.97	11.16	10.43	10.51	9.7
Runoff	Mon		0.35		0.23	2.08	1.65	1.95	2.00	2.20	1.91	0.0	2.02	1.44
	Tue		0.40	0.76	0.40	1.08	2.40	2.39	3.55	3.72	3.20	2.45	2.90	2.11
	Wed		0.0	0.75		1.78	2.70	2.90	4.15	3.07	2.82	2.47	1.50	2.21
	Thu	0.37	0.40	0.40	0.72	0.32	2.05	4.10	3.61	4.05		3.50	2.17	1.91
	Fri	0.0	0.40	0.47	0.40			4.80	3.55	2.92	3.56	2.85		2.11
	Mean	0.18	0.24	0.60	0.44	1.45	2.20	3.23	3.37	3.19	2.55	2.25	2.15	

Table I-2

Daily Concentration of Ammonium on the Cannery Wastewater Model

Liquid	Day	Concentration of NH_4, ppm, for Cited Week												
		1	2	3	4	5	6	7	8	9	10	11	12	Mean
Wastewater	Mon		9.20	10.45	8.97	5.35	4.50	6.50	6.90	5.60	9.40	22.00	7.30	8.74
	Tue		9.70	5.50	13.90	6.75	7.90	5.90	8.65	5.45	7.50	21.80	7.00	9.10
	Wed		9.10	6.85	8.35	5.90	10.50	6.10	6.70	6.40	7.50	14.35	7.40	8.10
	Thu		7.50	11.00	10.80	11.50	0.0	7.80	6.20	6.85	8.00	8.55	8.00	7.84
	Fri	5.70	10.00	9.55	8.10			5.00	7.60	7.00	7.00	6.70		7.41
	Mean	5.70	9.10	8.67	10.02	7.38	5.73	6.26	7.31	6.26	7.88	14.68	7.43	
Runoff	Mon		1.58			0.35	0.60	0.47	0.35	0.85	1.47		0.44	0.76
	Tue		1.90	0.63	2.45	0.79	1.15	0.49	0.83	0.25	1.90	3.20	0.63	1.29
	Wed		2.42	1.82		0.70	2.20	0.52	1.17	1.11	1.64	2.65	0.48	1.47
	Thu		2.75	4.45	0.51	0.73	0.17	0.75	1.23	1.31	1.18	1.52	0.41	1.36
	Fri	1.00	3.42	2.67	3.17			0.49	0.76	1.60	0.85	1.00		1.66
	Mean	1.00	2.41	2.04	2.21	0.64	1.03	0.55	0.87	1.02	1.41	2.09	0.49	

Table I-3

Daily Concentration for TKN on the Cannery Wastewater Model

Liquid	Day	Concentration of TKN, ppm, for Cited Week												
		1	2	3	4	5	6	7	8	9	10	11	12	Mean
Wastewater	Mon				10.25				7.25		14.60	28.30		15.10
	Tue			10.90	9.25				8.25		11.90	27.70		13.60
	Wed		8.40	12.15	9.00				7.32		13.60	18.60	9.95	11.29
	Thu		10.10	11.35	10.20						12.20	13.00		11.37
	Fri	9.30	9.10	13.55										10.65
	Mean	9.30	9.20	11.99	9.67				7.61		13.07	21.90	9.95	
Runoff	Mon				6.75				2.30		5.00	0.15		3.55
	Tue			7.15	6.45				2.75		6.20	7.25		5.96
	Wed		6.07	7.90	7.50				2.55		5.80	5.85	3.65	5.62
	Thu		7.55	8.05	7.75						4.75	5.40		6.70
	Fri	7.85	11.10	8.15										9.03
	Mean	7.85	8.24	7.81	7.11				2.53		5.44	4.66	3.65	

Table I-4

Volume of Applied and Runoff Liquid for the Cannery Wastewater Model

Liquid	Day	Liquid Volume, ℓ, for Cited Week												
		1	2	3	4	5	6	7	8	9	10	11	12	Mean
Wastewater	Mon		27.50	27.50	41.00	35.00	35.00	35.00	35.00	35.00	34.00	34.50	34.00	34.00
	Tue		27.50	27.50	27.50	28.50	28.50	28.50	28.50	28.50	28.00	28.00	28.50	28.10
	Wed		27.50	27.50	27.50	28.50	28.50	27.50	27.50	27.50	28.00	27.50	27.50	27.70
	Thu	27.50	27.50	27.50	28.50	28.50	28.50	28.50	28.50	28.50	28.00	27.50	27.50	28.00
	Fri	27.50	27.50	27.50	28.50	28.50		28.50	28.50	28.50	28.00	27.50		28.10
	Mean	27.50	27.5	27.5	30.6	29.8	30.1	29.6	29.6	29.6	29.2	29.0	29.4	
Runoff	Mon		0.80	0.0	16.40	12.50	19.50	2.00	6.60	16.10	12.25	0.25	4.00	8.2
	Tue		10.00	14.25	15.75	19.50	15.50	17.15	13.75	22.30	18.00	15.25	14.25	16.0
	Wed		12.00	11.40	17.65	16.60	14.45	18.00	13.00	14.10	13.80	14.75	10.25	14.2
	Thu	10.00	12.00	16.40	18.05	14.50	17.25	14.30	21.20	14.00	12.70	10.55		14.6
	Fri	14.25	14.50	14.50	17.50	15.00		19.25	15.75	15.55	9.90	5.60		14.2
	Mean	12.1	9.9	11.3	17.1	15.6	16.7	14.3	14.1	16.4	13.3	9.3	9.5	

Table I-5

Daily Concentration of COD on the Cannery Wastewater Model

Liquid	Day	COD, ppm, for Cited Week												
		1	2	3	4	5	6	7	8	9	10	11	12	Mean
Wastewater	Mon				175.0									175.0
	Tue			175.0								201.0	129.5	168.5
	Wed		290.0	127.0					128.5			178.5		181.0
	Thu		320.0	146.5							533.0	167.0		291.6
	Fri	633.0	109.0											371.0
	Mean	633.0	239.7	149.5	175.0				128.5		533.0	182.2	129.5	
Runoff	Mon				127.0									127.0
	Tue			179.0								98.5	53.0	110.2
	Wed		97.0	122.0					71.0			77.0		91.8
	Thu		152.0	104.0							129.0	89.0		118.5
	Fri	147.5	131.0											139.3
	Mean	147.5	126.7	135.0	127.0				71.0		129.0	88.2	53.0	

Table I-6

Conspicuous Flora and Fauna[a] of the Cannery Wastewater Model

Description		Habitat[b]		
Common Name	Scientific Name	Soil Surface With Algal Crust	Shaded Soil Surface (Lacking Crust)	Soil Profile (Grass Roots)
I. Algae				
A. Blue-green algae	*Oscillatoria* sp.	A		
B. Blue-green algae	*Anabaena* sp.	A		
C. Blue-green algae	*Cylindrospermum* sp.	C		
D. Green algae (desmids)	*Closterium* sp.	A		
E. Green algae	*Chlorococcum* sp.	C		
F. Yellow-green algae	*Vaucheria* sp.		A	
G. Yellow-green algae (diatoms)	Pennales (*Navicula*, *Nitzschia*)	A	C	
II. Slime molds	Liceales *Dictydium* sp.	C		
III. Works				
A. Sewage worms	*Tubifex* sp.	A	C	A
B. Bristle worms		A		
IV. Insects (larvae and pupae)				
A. Marsh flies	Sciomyzidae		C	A
B. Biting midges[c]	Ceratopogonidae	A	C	
C. Midges[c]	Chironomidae	A	C	
D. Moth flies[c]	Psychodidae	A	C	
E. Unidentified dipteran "blood worms"		A	A	A
V. Water mites	Hydrachnidae	A		
VI. Snails (gastropods)	*Lymnaea* sp.	A		

[a] Excluding dominant microscopic forms which were not noticeably visible, e.g. bacteria, fungi, actinomycetes, protozonans, nematodes, and rotifers.

[b] A denotes abundant; C denotes common.

[c] Identification made from pupae and adults.

Table I-7

Oxidation-Reduction Measurements for an 18-Day Period, Monitored at an 2.4-m Downslope Distance During Three Wetting and Drying Cycles of the 9th Through 11th Weeks, at Soil Depths of 0.5 and 1.3 cm Within the Cannery Wastewater Model

			Oxidation-Reduction Potential for Cited Soil Surface Condition			
			Blue-Green Algal Crust (Grass Kept Short)		No Algal Crust (Tall Grass)	
Week	Day	Time[a]	0.5 cm	1.3 cm	0.5 cm	1.3 cm
9	Wed	a	+174	-1	+84	+4
		b	+256	-131	+9	-36
	Thu	b	+64	-161	-106	-116
	Fri	a	+454	-166	-146	-126
10	Mon	a	+314	-176	+314	-116
		b	+249	-196	+44	-196
	Wed	b	+164	-181	+44	-196
	Thu	a	-86	-171	+14	-156
11	Mon	a	+354	+324	+324	-154
	Tue	a	+114	+4	+324	-56
		b	+224	-61	+124	-96
	Wed	a	-16	-71	+84	-26
	Thu	a	-56	+24	+84	-156
		b	+119	-21	+54	-156
		b	+369	+204	+359	+34

[a] a denotes morning; b denotes afternoon.

Table I-8

Oxidation-Reduction Measurements for a 5-Day Wetting and Drying Cycle, Monitored at Soil Depths of 0.5 and 1.3 cm During the 12th Week in the Cannery Wastewater Model

		Oxidation-Reduction Potential for Cited Soil Depth of Various Downslope Distances							
		0.76 m		1.52 m		2.29 m		Mean	
Day	Hours[a]	0.5 cm	1.3 cm	0.5 cm	1.3 cm	0.5 cm	1.3 cm	0.5 cm	1.3 cm
Thu	10	+112	-61	+147	-76	+99	-1	+119.0	-46.0
	11	+104	-66	+304	+104	+22	-36	+143.2	+66.0
	12	+184	-81	+99	-89	+22	+79	+101.5	-30.2
	13.5	+249	-86	+34	-114	+182	-41	+154.8	-80.2
	15	+194	-119	-11	-109	+119	-56	+100.7	-94.3
	16	+119	-99	-59	-111	+109	+72	+56.5	-46.0
	20	+14	-84	-56	-91	-1	-34	-14.3	-69.3
	Mean	+276.7	-84.9	+65.4	-69.2	+78.6	-2.4		
Fri	11.5	-76	-146	+139	-166	+24	-71	+29.0	-127.7
	14.5	-66	-156	-39	-131	+40	-166	-21.5	-151.0
	Mean	-71	-151.0	+50.3	-148.5	+32.0	-118.5		
Sat	13.5	-59	-94	+72	-181	-49	-106	-11.8	-126.8
	Mean	-59.0	-93.5	+71.5	-181.0	-48.5	-106.0		
Sun	15.5	+279	-221	+294	+74	+219	-21	+264.0	-56.0
	Mean	+279.0	-221.0	+294.0	+74.0	+219.0	-21.0		
Mon	10.0	+479	+72	+417	+389	+319	-46	+404.8	+138.2
	13.5	+288	-84	+294	+289	+319	+9	+300.2	+86.8
	18.0	+142	-121	+244	+119	+274	-49	+219.8	-16.8
	Mean	+302.7	-44.3	+318.2	+265.7	+304.0	+28.5		

[a] Based on 24-hour clock.

Table I-9

Wastewater and Soil pH Values for the

Cannery Wastewater Model

Sample	Model Location	Horizontal Distance, m	pH Value
Wastewater	Surface	0.00	7.3
			7.2
			7.5
	Bottom	3.05	7.2
Soil	Surface	0.00	6.9
			7.2

Table I-10

Flow Rates of Applied Wastewater and Runoff From 3.05 m of Clay Soil Supporting Reed Canary Grass, Previously Irrigated With Cannery Waste

		Flow Rate, $m\ell/min$ for Cited Week												
Liquid	Day	1	2	3	4	5	6	7	8	9	10	11	12	Mean
Wastewater	Mon					199	213	266	236	220	180	194	186	211.8
	Tue					199	189	262	236	216	200	143	165	201.3
	Wed					199	191	178	236	206	146	141	162	182.4
	Thu					195	191	236	236	204	144	169	179	194.3
	Fri					195		236	232	220	146	206		205.8
	Mean					197.4	196.0	235.6	235.2	213.2	163.2	170.6	173.0	
Runoff	Mon							123	114	172	118		111	127.6
	Tue							230	176	187	153	71	79	149.3
	Wed							148	179	150	102	77	104	126.7
	Thu							173	180	166	73	107	112	135.2
	Fri							190	105	168	82	112		131.4
	Mean							172.8	150.8	168.6	105.6	91.8	101.5	

Table I-11

Concentrations and Percent Removal for Major and Minor Elements in Overland Flow From the Cannery Wastewater Model

Element	Concentration (mg/ℓ) Applied Wastewater	Runoff	Percent Removal[a]
P	16.96	4.93	89.4
K	2.35	7.02	-41.9
Ca	22.50	61.00	-28.8
Mg	1.70	4.30	-20.1
Na	76.85	107.20	33.8
Mn	0.03	0.17	-169.2
Zn	0.10	0.08	62.2
Cu	0.02	0.02	53.0
Pb	0.02	0.02	53.0
Ni	0.00	0.00	
Cd	0.00	0.00	--

[a] Calculated on a weight basis.

Table I-12

Average Weekly Maximum and Minimum Greenhouse Temperatures During the Experiment

Temperature[a]	Time, Weeks											
	1	2	3	4	5	6	7	8	9	10	11	12
Maximum	30.4 (86.7)	31.2 (88.3)	30.2 (86.4)	30.0 (86.0)	27.6 (81.7)	31.0 (87.9)	28.5 (83.3)	30.3 (86.6)	31.5 (88.7)	30.2 (86.4)	30.1 (86.3)	31.1 (88.1)
Minimum	22.0 (71.6)	21.8 (71.3)	18.4 (65.1)	15.0 (59.0)	11.3 (52.3)	13.7 (56.7)	13.9 (57.0)	11.1 (52.0)	19.0 (66.3)	15.0 (59.0)	16.6 (61.9)	16.8 (62.3)

[a] Centigrade degrees; Fahrenheit degrees in parentheses.

APPENDIX II

Data From the Sludge Model

Table II-1

Removal of NO_3 by Susquehanna Clay Amended With Sewage Sludge

Liquid	Day	Concentration of NO_3, ppm, for Cited Week: 1	2	3	4	5	6	7	8	9	10	Mean
Effluent Waste-water	Mon			8.00	8.40	7.00	8.40	8.50	8.05	9.00	10.00	8.42
	Tue			8.00	9.40	8.90	9.10	10.60	9.35	9.10	13.20	9.71
	Wed		9.05	8.90	9.50	10.55	9.60	8.85	9.90	10.80	8.80	9.55
	Thu		8.00	9.20	17.00	11.00	8.90	10.00	11.65	11.80	9.20	10.75
	Fri	8.50										8.50
			8.52	8.52	11.07	9.36	9.00	9.49	9.74	10.17	10.30	
Runoff	Mon			6.20	5.00	0.86	0.70	2.25	0.0	0.0	0.41	1.93
	Tue			4.90	4.50	1.30	0.25	1.85	0.18	0.20	0.22	1.68
	Wed		5.00	5.00	3.35	2.00	0.34	0.75	0.45	0.92	1.05	2.10
	Thu		4.75	4.70	8.90	0.80	0.40	0.57	0.62	1.57	0.0	2.48
	Fri	4.95										4.95
			4.86	5.20	5.44	1.24	0.42	1.35	0.31	0.67	0.42	
Subflow	Mon			0.28	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.03
	Tue			0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Wed		0.25	0.25	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.05
	Thu		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Fri	0.0										0.0
	Mean	0.0	0.13	0.13	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Table II-2

Removal of NH_4 by Susquehanna Clay Amended With Sewage Sludge

Liquid	Day	Concentration of NH_4, ppm, for Cited Week: 1	2	3	4	5	6	7	8	9	10	Mean
Effluent Waste-water	Mon		16.25	4.70	9.40	7.00	8.50	7.30	10.75	22.75	6.70	10.37
	Tue		13.25	5.05	5.10	6.80	7.40	6.70	8.40	21.00	6.95	8.96
	Wed		11.75	4.90	9.20	6.50	7.90	7.25	7.95	11.90	7.20	8.28
	Thu		16.85	3.45	10.00	6.40	6.80	6.50	8.20	8.25	7.85	8.26
	Fri	10.00										10.00
	Mean	10.00	14.52	4.52	8.42	6.67	7.65	6.94	8.82	15.97	7.17	
Runoff	Mon		2.40	1.10	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.39
	Tue		1.15	0.61	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.20
	Wed		1.55	0.34	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.21
	Thu		2.60	0.0	0.25	0.0	0.0	0.0	0.0	0.0	0.0	0.32
	Fri	1.40										1.40
	Mean		1.92	0.51	0.06	0.0	0.0	0.0	0.0	0.0	0.0	
Subflow	Mon		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Tue		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Wed		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Thu		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Fri	0.0										0.0
	Mean	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Table II-3

Removal of TKN by Susquehanna Clay Amended With Sewage Sludge

Liquid	Day	TKN Concentration, ppm, for Cited Week										
		1	2	3	4	5	6	7	8	9	10	Mean
Wastewater	Mon			5.45	0.0				14.70	26.10		11.55
	Tue			7.85	5.35	7.60			12.20	27.40		12.08
	Wed		9.40					9.20	10.85	16.60		11.51
	Thu				13.30				12.10	11.80		12.40
	Fri											
	Mean		9.40	6.65	6.22	7.60		9.20	12.46	20.47		
Runoff	Mon			1.60	1.50	1.20	2.00		5.60	5.75		2.49
	Tue		3.40	1.13	1.00	0.60	1.00		3.75	3.50		2.05
	Wed			0.50	0.75	1.15	0.70	2.30	4.25	2.90		1.79
	Thu		2.50	0.0	1.20	1.15			3.65	3.25		1.96
	Fri											
	Mean		2.95	0.81	1.11	1.02	1.23	2.30	4.31	3.85		
Subflow	Mon		0.25	0.45	0.0	0.35	0.0					0.21
	Tue		0.70	0.0	0.0	0.0	0.05					0.15
	Wed		0.45	0.0	0.0	0.0	0.25					0.14
	Thu		0.15	0.0	0.15	0.35						0.16
	Fri											
	Mean				0.04	0.17	0.10					

Table II-4

Wastewater Applied to and Collected From Susquehanna Clay Amended With Sewage Sludge

Liquid	Day	Liquid Volume, ℓ, for Cited Week										
		1	2	3	4	5	6	7	8	9	10	Mean
Wastewater	Mon		30.50	30.50	30.50	30.50	30.50	30.50	30.50	30.50	30.50	30.50
	Tue		30.50	30.50	30.50	30.50	30.50	30.50	30.50	30.50	30.50	30.50
	Wed		30.50	30.50	30.50	30.50	30.50	30.50	30.50	30.50	30.50	30.50
	Thu		30.50	30.50	30.50	30.50	30.50	30.50	30.50	30.50	30.50	30.50
	Fri	30.50										30.50
	Mean	30.50	30.50	30.50	30.50	30.50	30.50		30.50	30.50	30.50	
	Mon		22.00	10.50			6.15	8.70	5.00	5.40	1.50	8.71
	Tue		18.20	19.00			15.50	20.60	20.70	15.00	9.00	17.29
	Wed		19.50	16.50			15.15	17.50	18.85	18.50	16.00	17.59
	Thu		23.85	14.00			16.00	16.90	17.65	16.00	15.00	17.13
	Fri	17.70										17.70
	Mean	17.70	20.89	15.00	17.52	14.47	13.20	15.92	15.55	13.72	10.38	
Subflow	Mon		2.40	1.75	3.95	1.25	1.95	1.30	1.50	0.20	0.20	1.61
	Tue		4.60	3.45	3.85	2.85	3.20	3.05	3.25	2.10	1.15	3.05
	Wed		5.00	2.85	4.20	3.80	3.45	2.20	3.10	3.00	1.15	3.19
	Thu		5.35	2.55	4.55	3.10	3.85	3.25	3.90	2.65	1.20	3.38
	Fri	4.90										4.90
	Mean	4.90	4.34	26.5	4.14	2.75	3.11	2.45	2.94	1.99	0.92	

Table II-5

Oxidation-Reduction Measurements for a 5-Day Wetting and Drying Cycle Monitored at Soil Depths of 0.5 and 1.3 cm During the 12th Week in Susquehanna Clay Amended With Sewage Sludge

Day	Hours[a]	Oxidation-Reduction Potential, mv, for Cited Soil Depth at Various Downslope Distances									
		0.30 m		0.76 m		1.52 m		2.29 m		Mean	
		0.5 cm	1.3 cm	0.5 cm	1.3 cm	0.5 cm	1.3 cm	0.5 cm	1.3 cm	0.5 cm	1.3 cm
Thu	12	369	-181	379	-176	154	-161	-96	-166	201.50	-171
	13.5	124	-181	129	-191	114	-131	219	-166	146.50	-167.25
	14	164	-186	-96	-201	29	-66	64	-206	40.25	-164.75
	15	254	-186	-236	-196	14	-216	-216	-201	-46	-199.75
	20	4	-166	-156	-66	164	124	-236	-201	-56	-77.25
	Mean	218.29	-173.14	-13.86	-136	67.57	-101.71	-111	-130.29		
Fri	35	144	-176	-186	134	-16	-61	-236	-226	-73.50	-82.25
	38.5	234	-176	-106	69	-16	-56	384	-211	124	-93.50
	Mean	189	-176	-146	101.5	-16	-58.5	74	-218.5		
Sat	61	-126	-196	-136	254	314	196	544	234	149	122
Sun	88.5	-66	154	479	-46	344	-136	74	-16	207.75	-11
Mon	105.5	319	-51	84	174	279	4	464	-26	286.50	25.25
	107.5	449	-61	179	164	199	4	574	-26	350.25	20.25
	113.5	219	-96	14	94	44	-46	114	-86	97.75	-35.50
	Mean	329	-69.33	92.33	144	174	-12.66	384	-46		

[a] Based on 24-hour clock.

Table II-10

Removal of Sodium From Susquehanna Clay

Amended With Sewage Sludge

Liquid	Day	Concentration of Sodium, ppm, for Cited Week 5	6	7	8	9
	Mon	101.40				66.35
Wastewater	Tue					87.60
	Wed	86.10	67.70	82.60	85.85	81.80
	Thu				107.70	100.00
	Mean	93.75	67.70	82.60	96.77	83.94
	Mon	89.95				93.65
Runoff	Tue					93.75
	Wed	68.75	70.95	77.30	81.50	69.55
	Thu				107.10	93.45
	Mean	79.35	70.95	77.30	94.30	87.60
	Mon	8.50				9.30
Subflow	Tue					18.90
	Wed	4.65	6.20	8.10	16.35	16.50
	Thu				20.55	17.60
	Mean	6.57	6.20	8.10	18.45	15.57

Table II-11

Removal of Zinc From Susquehanna Clay

Amended With Sewage Sludge

Liquid	Day	Concentration of Zinc, ppm, for Cited Week 5	6	7	8	9
	Mon	0.33				0.34
Wastewater	Tue					0.21
	Wed	0.32	0.37	0.53	0.32	0.13
	Thu				0.39	0.22
	Mean	0.32	0.37	0.53	0.36	0.23
	Mon	0.15				0.12
Runoff	Tue					0.07
	Wed	0.15	0.16	0.22	0.14	0.07
	Thu				0.21	0.08
	Mean	0.15	0.16	0.22	0.17	0.09
	Mon	0.06				0.10
Subflow	Tue					0.08
	Wed	0.08	0.12	0.13	0.24	0.14
	Thu				0.14	0.30
	Mean	0.07	0.12	0.13	0.19	0.16

Table II-12

Removal of Manganese From Susquehanna Clay

Amended With Sewage Sludge

Liquid	Day	Concentration of Manganese, ppm, for Cited Week 5	6	7	8	9
	Mon	0.22				0.26
Wastewater	Tue					0.17
	Wed	0.20	0.19	0.21	0.21	0.14
	Thu				0.23	0.18
	Mean	0.21	0.19	0.21	0.22	0.19
	Mon	0.07				0.01
Runoff	Tue					0.01
	Wed	0.01	0.01	0.01	0.0	0.01
	Thu				0.01	0.01
	Mean	0.04	0.01	0.01	0.00	0.01
	Mon	0.09				0.02
Subflow	Tue					0.02
	Wed	0.01	0.01	0.03	0.05	0.04
	Thu				0.04	0.01
	Mean	0.05	0.01	0.03	0.04	0.02

Table II-13

Removal of Copper From Susquehanna Clay

Amended With Sewage Sludge

Liquid	Day	Concentration of Copper, ppm; for Cited Week 5	6	7	8	9
	Mon	0.17				0.22
Wastewater	Tue					0.16
	Wed	0.09	0.07	0.20	0.02	0.07
	Thu				0.09	0.11
	Mean	0.13	0.07	0.20	0.06	0.14
	Mon	0.06				0.01
Runoff	Tue					0.01
	Wed	0.01	0.01	0.0	0.0	0.01
	Thu				0.03	0.01
	Mean	0.03	0.01	0.0	0.01	0.01
	Mon	0.01				0.01
Subflow	Tue					0.01
	Wed	0.01	0.01	0.0	0.0	0.01
	Thu				0.0	0.01
	Mean	0.01	0.01	0.0	0.0	0.01

Table II-14

Removal of Nickel From Susquehanna Clay Amended With Sewage Sludge

Liquid	Day	Concentration of Nickel, ppm, for Cited Week				
		5	6	7	8	9
	Mon					0.04
Wastewater	Tue					0.10
	Wed	0.04	0.05	0.14	0.10	0.06
	Thu				0.0	0.04
	Mean	0.04	0.05	0.14	0.05	0.06
	Mon					0.0
Runoff	Tue					0.01
	Wed	0.02	0.02	0.0	0.0	0.01
	Thu				0.02	0.01
	Mean	0.02	0.02	0.0	0.01	0.01
	Mon					0.0
Subflow	Tue					0.01
	Wed	0.02	0.02	0.0	0.0	0.01
	Thu				0.0	0.01
	Mean	0.02	0.02	0.0	0.0	0.01

Table II-15

Removal of Lead From Susquehanna Clay Amended With Sewage Sludge

Liquid	Day	Concentration of Lead, ppm, for Cited Week				
		5	6	7	8	9
	Mon	0.09				0.03
Wastewater	Tue					0.02
	Wed	0.25	0.22	0.11	0.08	0.03
	Thu				0.22	0.10
	Mean	0.17	0.22	0.11	0.15	0.05
	Mon	0.02				0.01
Runoff	Tue					0.01
	Wed	0.03	0.04	0.04	0.0	0.01
	Thu				0.01	0.01
	Mean	0.02	0.04	0.04	0.00	0.01
	Mon	0.01				0.01
Subflow	Tue					0.01
	Wed	0.01	0.01	0.0	0.01	0.01
	Thu				0.01	0.01
	Mean	0.01	0.01	0.0	0.01	0.01

Table II-16

Removal of Cadmium From Susquehanna Clay Amended With Sewage Sludge

Liquid	Day	Concentration of Cadmium, ppm, for Cited Week 5	6	7	8	9
	Mon	0.23				0.27
Wastewater	Tue					0.10
	Wed	0.22	0.24	0.23	0.24	0.13
	Thu				0.21	0.19
	Mean	0.23	0.24	0.23	0.23	0.17
	Mon	0.05				0.01
Runoff	Tue					0.01
	Wed	0.01	0.01	0.04	0.01	0.01
	Thu				0.01	0.01
	Mean	0.03	0.01	0.04	0.01	0.01
	Mon	0.01				0.01
Subflow	Tue					0.01
	Wed	0.0	0.01	0.01	0.01	0.01
	Thu				0.01	0.01
	Mean	0.0	0.01	0.01	0.01	0.01

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13. ABSTRACT

This study was limited to land treatment as a means of achieving advanced wastewater purification. Land treatment has the advantage of incorporating the recycling concept directly into its treatment mode, resulting in replacement rather than depletion of natural resources. Also, some form of control over ecologically damaging components is retained. This report presents results of a literature review on various methods of treating wastewater on land and also presents results of model tests of the overland flow method, with particular emphasis on nitrification and denitrification. Two types of soil systems for overland flow treatment of wastewater were investigated during these model tests. One soil was from an 8-year-old commercial cannery wastewater treatment site. The other was from an untreated natural site in a national forest that was low in indigenous soil organic matter; consequently, this latter system was amended with sludge in order to increase its organic matter content. Thus, both experimental soils represented soil systems that had more organic matter and biological activity than an average heavy clay soil. Overland flow on previously untreated, low-organic clay soil was investigated in an independent but parallel study that will be reported later. Based on the results of the model tests, the following conclusions were drawn: 1. Overland flow is an effective means of removing nitrogen from secondarily treated wastewater. 2. Algal crust probably hinders rather than promotes nitrogen removal. 3. Reed canary grass is capable of removing considerable quantities of nitrogen from an overland flow system. 4. Nitrification and denitrification in an overland flow system appear to function in a manner similar to that observed in rice fields. Nitrification occurs in a surface "oxidized" layer, and denitrification occurs in the underlying "reduced" layer of an "oxidized-reduced double layer" in the upper soil profile. 5. The thickness of the soil aerobic zone rapidly fluctuates in response to flooding, and diminishes with duration of flooding and organic load in the applied water. 6. Anaerobic conditions seem to be prevalent at greater than a few millimeters soil depth during flooding of overland flow systems, but aerobic conditions can be attained after a 2-day rest period. 7. Accumulation of easily degraded organic material on the soil surface may cause the oxidized layer to be at or in the surface liquid above the soil, which would result in the need for a rest period. 8. Generally, nitrogen removal should be higher if the nitrogen is in the nitrate form, due to the prevalence of reduced conditions in an overland flow system. 9. Plant species that can tolerate reduced soil conditions must be used in overland flow systems. Reed canary grass functions quite well under these conditions. 10. Contact of the wastewater with (Continued)

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13. ABSTRACT (Concluded)

the soil and its associated large microbial population may be essential to proper system functioning. 11. Proper functioning of an overland flow system seems to require free diffusion between the oxidized and reduced zones in the soil. 12. Nitrogen treatment efficiency is highly related to the rate of application and slope length. 13. A new overland flow system may require a 4- to 5-week period before nitrogen removal efficiencies approach maximum values. 14. Sewage sludge incorporated into the upper few centimeters of the soil on a 6 percent by weight basis does not greatly help or hinder advanced treatment of wastewater by overland flow. 15. Heavy metals in the wastewater appear to be removed quite effectively within the first meter of downslope distance. 16. Phosphorous removal appears to be the short-term limiting factor for overland flow. 17. Models 3.05 and 6.10 m long are effective in studying overland flow, especially with respect to nitrogen treatment. Several interim recommendations based on both the literature review and the experimental results of this study are given.

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