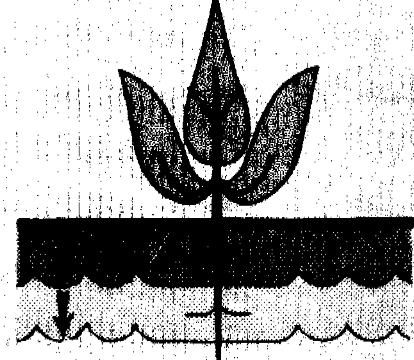
State of Knowledge in LAND TREATMENT OF WASTEWATER



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SYMPOSIUM COORDINATOR: H.L. McKIM

State of Knowledge in

LAND TREATMENT OF WASTEWATER

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H.L. McKim
Symposium Coordinator

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PREFACE

Many countries have been applying wastewater to the soil for centuries. In almost all instances the nutrients in the effluent from agricultural and domestic sources have been used for their fertilizer value. This alone can make the concept of land treatment of wastewater a cost-effective alternative to other forms of treatment of these waste products. Agricultural crops produced at land treatment sites are of high quality and can be used as an animal food source. When wastewater is applied to various forest species, yields may increase by a factor of two. This increase in yield may provide an alternative energy source to replace diminishing oil, gas and coal supplies.

In the United States several land treatment systems have been in operation since the turn of the century. In the past ten years, however, Federal legislation has not only given municipalities added incentive to utilize land treatment but it has also mandated that land treatment be one of the treatment alternatives that is considered prior to applying for federal construction grants. If land treatment is the selected method its percolating water must meet prescribed water quality standards and it must be the most cost-effective method for treating the wastewater.

The objectives of this Symposium are to summarize the state of knowledge of the practical aspects of the treatment of wastewater by land application and to identify the suitable approaches for the design of such land treatment systems. The topics to be included are: site selection considerations, case studies of national and international concern, health effects of land treatment systems, pretreatment considerations, uses of wastewaters in agricultural and forest systems, monitoring, modeling and design criteria.

The Symposium Proceedings are published in two volumes. Volume I contains the invited papers presented and discussed at the conference. Volume 2 contains shorter papers about on-going research that were selected from the responses received following a call for abstracts.

The program committee wishes to acknowledge Mr. Noel Urban, Chief, Urban Studies and Management Section, Engineering Division, OCE for his support of the Symposium. Special thanks is offered to Mr. Ben Yamashita and Ms. Mary Ann Austin of the Public Affairs Office, Mr. Stephen L. Bowen and Mrs. Donna R. Murphy of the Publications Section and Mrs. Donna Gerow of the Word Processing Center for their assistance in preparation and layout of the proceedings. In addition, the Symposium coordinator is particularly indebted to the program committee for their assistance in coordination and review of the proceedings, to Mrs. Sandra Messersmith who helped in the overall planning of the Symposium, to Ms. Nancy Dumont who arranged for the Russian translations, and to CRREL management for their support and cooperation in hosting the Symposium.

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CASE STUDIES

A CASE STUDY OF SOIL PHOSPHORUS AND HEAVY METALS DUE TO EXTENDED EFFLUENT IRRIGATION

Mikkel R. Anderson, URS Co., Seattle, Wn.

A spray irrigation facility operated for fourteen years near Seattle, Washington was examined for soil enrichment of phosphorus, copper, zinc, lead, and chromium. Profiles were take in a naturally acid, sandy loam, in which pasture grasses were being grown with effluent from a trickling filter secondary treatment facility.

At the time of sampling and for a significant portion of the facility's life, the field was receiving extreme hydraulic loads of over 4 cm per day over the 18.6 hectares with no resting periods. The upstream treatment plant was continually exceeding design capacity. The field operated with no storage during rainy or freezing periods. Effluent quality was monitored and mass loadings computed to compare with estimated enrichment values. Enrichment was calculated as the difference between the estimated mass of each constituent in the irrigated profiles and the control profiles. Analyses revealed definite enrichments of phosphorus, copper, zinc and lead. The response to chromium appeared indefinite due to the low level of chromium addition relative to soil background levels. Soil pH showed a one unit depression from the control to irrigated profiles with one surface value recorded at pH 4.5. Results for soil concentrations were compared to results reported as part of the Penn State effort and found to be comparable. Conclusions drawn indicated that despite severe hydraulic overloading and low soil pH, heavy metals and phosphorus were being readily removed by the soil profile.

BACKGROUND

This research reviewed the performance of the Lake Hills Sewage Treatment Plant near Seattle, Washington, a full-scale spray irrigation facility in a moderate, humid climate. This facility was erected as a temporary means of disposal until the completion of a major intercepter to this area. Because of project scheduling, this "temporary" facility served from 1959 to 1973.

The Lake Hills treatment facility consisted of primary sedimentation, trickling filtration, secondary sedimentation, disposal by pumping to a sprayfield utilizing seven sets of agricultural type-impact sprinklers. At the end of the service life, the spray area of 18.63 hectares was heavily overloaded hydraulically with flows reaching a peak of 22,710 cubic meters per day ($m^3/_d$) with peak daily averages around 15,000 $m^3/_d$ compared to a design flow of 9,500 $m^3/_d$. The application rate of 4.11 centimeters (cm) /day was greater than five times that suggested as desirable in the literature.[1] Despite this overloading, the field continued to operate satisfactorily from the standpoint of wastewater disposal and nuisance problems.

The specific objectives of this research were to analyze the possible buildup and penetration in the soil of the heavy metal ions of copper, zinc, chromium, and lead; and the biological stimulant phosphorus. Columns of soil were sampled with depth in two irrigated and two adjacent, non-irrigated profiles. Concentrations of constituents were determined and compared with depth for analysis of profile penetration, and compared with effluent application rates to estimate long-term removal effectiveness.

INSTALLATION CHARACTERISTICS

The Lake Hills spray field is located on a deposit of Lynden loamy sand, a well-drained, moderately acid, rich reddish-brown, loamy sand with considerable coarse sand and a scattering of shot pellets. The upper ten inch subsoil is a moderately acid, yellowish-brown, loamy sand. These materials are slightly acid to neutral.

The total area of the sprayfield was 21 hectares. However, only about 18.63 hectares were actually utilized for irrigation. The site is naturally quite flat with a maximum elevation variation of no more than six in 1000. The winter water table is approximately one meter under the surface.

Vegetal cover at the site was primarily coarse pasture grasses. At the time of sampling, hydraulic loading was severe enough to have resulted in formation of slime mats in the immediate vicinity of the sprinkler heads, and the vegetation was sparse to non-existent in these areas. Elsewhere within the influence of the sprinklers and subsequent runoff, the grass was generally quite lush. Crop growth was luxurious enough to permit up to three cuttings of hay annually when timing of rains would permit farm machinery to operate adequately. During periods of high rainfall and plant flows, runoff from the field would pond on part of the nearby, well-used King County Marymoor Park necessitating the shutdown of adjacent sets of sprinklers for health hazard considerations.

The disposal site was divided into seven "fields", with a maximum design application rate of 11.5 cubic meters per hectare (m^3/ha) or 0.58 centimeters per day. Information regarding design application schedules was not available. Automatic valves were installed to allow remote control of field irrigation from the plant, but abandoned due to cold weather malfunctions. Although no field operation records exist, it appears that the manual selection of irrigation fields was somewhat arbitrary with the exceptions of selective field drying for harvesting and full field operation during freezing weather.

The effluent pump station was designed for 15,400 liters per minute maximum, utilizing three pumps with a combined output of 181 kilowatts with all seven fields operating. In an effort to lower the system head, increase pump output, and reduce maintenance, the nozzle orifice portions of the sprinklers were removed. At first, this was only practiced during wet weather flows, but towards the end of the life of the installation, the nozzles were left out all the time. This practice aggravated conditions of hydraulic overloading by adding the problems of compaction due to droplet impact and reduced the effective area of primary application. As a result, localized conditions of overland flow existed within the vicinity of the application circle, and standing water was common.

SAMPLING METHODS

Effluent Samples

A concentrations taken in 1971 and 1972, and 3) examinations by the author of samples collected coincident with field sampling efforts in 1973. A summary of effluent quality appears in Table 1. All tests were run according to Standard Methods.[2]

Soil Sampling

Sampling was carried out on two paired profiles, one irrigated and one control each exposed by hand digging. Profiles 2 and 3 were one pair, 4 and 5 the other. 2 and 5 being the irrigated profiles and 3 and 4 being control. Sites indicated as control were established in areas where influence from wastewater spraying would be minimal, but with close proximity to the sites in irrigated soils.

The samples were air dried for 48 hours under a forced-draft hood. The samples were then ground in an attempt to separate any fine material adhering to larger particles. The grindings were passed over a 0.27 mm plastic sieve. These portions were utilized for analysis of copper, zinc, lead, chromium, and phosphorus.

Soil pH was measured according to procedures developed by Black.[3]

Soil sample digestrates were analyzed with an atomic absorption spectrophotometer for metal contents. Digestion was a modification of the procedure suggested by Jackson[4] utilizing, HF, HNO_3, $HClO_4$ and HCl acids.

Total phosphorus determinations were performed by the vanadomolybdophosphoric acid colorimetric method per Standard Methods[2]

TABLE 1

LAKE HILLS STP EFFLUENT QUALITY

Parameter	Mean Concentration	Number of Samples
Biochemical Oxygen Demand	51 mg/l	120[2]
Suspended Solids	34 mg/l	120[2]
Copper	0.28 mg/l	10[3]
Zinc	0.37 mg/l	10[3]
Lead	0.05 mg/l	10[3]
Chromium	0.02 mg/l	10[3]
Phosphorus (as P)	3.6 mg/l	7[1]
pH	6.8	7[2]

Source Key:

(1) Field samples and analyses, 1973.

(2) Seattle METRO operating records, courtesy Mr. Gary Isaac, 1962-1973.

(3) Seattle METRO laboratory data, courtesy Mr. Jim Hinman, 1971-1972.

RESULTS

pH

Soil pH measured in water is displayed in Figure 1. The two irrigated soil profiles show roughly a one pH unit drop relative to the control soils at all depths. Measured values for the control soils are equal to or slightly higher than values measured for the Lynden series in 1938. What appears to be the case is that medium to slightly acid soil has become very strongly to strongly acidic at all depths.

The significance of the low pH values is prelimarily the increased solubility of metal ions which would result. Numerous researchers, including Leeper[5] and Chaney[6] have cited reports showing little to no reduction in yield of crops at relatively high metals levels and a pH of near neutral, but noticeable toxicity when the pH was dropped to around 5.5 with all else held equal. These same researchers have recommended that soil pH be kept at about 6.5 to avoid excessive crop uptake of heavy metals. At Lake Hills only the deeper samples in the control profiles had pH near or above 6.5.

At Lake Hills, if increased mobility of ions due to low pH occured, it should have resulted in higher values of ions in the lower portion of the profiles where the pH was higher. However, such a condition was not observed. The pH depression was not significant enough to promote increased downward mobility of the specific ions examined at this site.

Lead

As shown in Figure 2, lead values were low, both in the soil and the effluent. This was reassuring considering lead's potential toxicity to humans.

The irrigated profiles of soil showed a moderate but consistent enrichment over the control soils. As in the zinc and copper determinations, all profiles tended toward a common value below 60 centimeters. In light of the fact that lead concentrations in the upper portion of a plant are usually lower than in the roots, even in highly contaminated soils, it appears that lead buildup presented no threat at Lake Hills either from the standpoint of public health or crop health. The only suggestion the analyses of lead raises, is the avoidance of edible root crops on fields to be irrigated with wastewater.

Zinc

The major source of zinc in Lake Hills water was corrosion of zinc galvanized water pipes. With the Seattle area's soft, corrosive water, this is a relatively rapid process.[7] The water applied to the sprayfield was richer in zinc than any metal measured.

Zinc displayed an obvious enrichment due to irrigation as displayed in Figure 3. Both irrigated profiles showed nearly identical values at depths to 40 centimeters. Enrichment of the first 30 centimeters was the most pronounced, with zinc values estimated at more than twice that of the control soils' upper 30 centimeters of depth. The second 30 centimeters of soil also showed enrichment, but this appeared due to higher values in the upper layer being reflected into the upper portions of the underlying layer. However, the deepest sample values tended toward a common value. Using the analogy of an ion exchange bed, it does not appear that zinc had "broken through" at depths greater than one meter.

The significance of 200 micrograms per gram in the upper soil is probably one of plant pathology rather than public health. Although zinc enrichment was definitely occurring, the fourteen years (1959 to 1973) of operation did not result in phytotoxic buildups of zinc.

From the standpoint of zinc, it appears that the Lake Hills sprayfield was removing zinc effectively without toxic buildup. The acid pH of the soil did not appear to promote excessive mobility to lower levels of the profile. It appears that the sprayfield could operate for an additional 20 years or so without extreme ramifications resulting from zinc buildup, but the question of sublethal crop toxicity would still need to be investigated.

Copper

The results of copper determinations showed a similar, but less pronounced pattern to that of zinc. This is not surprising as copper was being applied at a lesser rate (0.28 milligrams per liter Cu vs. 0.37 ppm Zn) although of the same magnitude as zinc. Plumbing corrosion plays the key role as a source.

As displayed in Figure 4, the maximum value for a soil sample was 132 micrograms/gram, occurring at the five centimeter depth in profile two. This was also the depth where maximum zinc values were recorded, emphasizing the similarity of the two profiles. Copper values also showed no "breakthrough", again tending toward natural levels below 60 centimeters in depth and above the water table.

Chromium

The results of the chromium analyses are confusing compared to the results of the other heavy metals. The possibility of unknown difficulties in the laboratory analyses cannot be ruled out. Profile two shows a substantial, albeit erratic, enrichment over control profile three, but irrigated profile four and control profile five reveal almost identical values for the first and second 30 centimeters. To further cloud the results, no profile showed any tendency to coincide at a common value below 60 centimeters.

It appears that chromium values in the irrigated and non-irrigated profiles are so close that no pattern is apparent. This is an analogy to a low "signal" to "noise" ratio.

Phosphorus

Christman[1] estimated that efficiencies in the range of 99% could be expected for phosphorus removal. At Penn State, Hook, Kardos and Sooper[8] reported a mean removal rate over seven years of spray application of 98.3% when effluent was sprayed on crops grown on clay loam and sandy loam soils.

The analyses at Lake Hills indicates a definite buildup of phosphorus in the irrigated profiles. The irrigated profiles contained nearly 400% as much phosphorus as the control profiles in the first 30 centimeters, while the second 30 centimeters contained almost 500% as much.

Calculations of Enrichment

The total mass present in the irrigated profiles minus the mass in the control profiles was taken to represent the enrichment due to irrigation. These values were calculated for the 0 to 30 centimeter depth and 30 to 60 centimeter depth, reported herein as 30 centimeter and 60 centimeter respectively. These computed values are displayed in Table 2.

For the purpose of comparison, Table 3 displays reported values for metals in wastewater and soil versus the Lake Hills values. From the values, it appears that the Lake Hills effluent was enriched, primarily by water supply corrosion products, to greater levels than the effluent reported by Sidle, et al,[9] except for the element lead. Correspondingly, the values for the soil profiles show a similar pattern of concentrations, although neither exceeds the values reported as "common" in agricultural soils.

Table 4 compares the estimated applied pollutant loading with the estimated soil enrichment. Several observations are possible.

All the heavy metals, except chromium, appeared to be enriched in the soil in roughly the same magnitude as they were applied. This might be expected, as these elements should be bound readily by soil processes. However, it is interesting to note that this apparent enrichment occurred in spite of the low soil pH measured. Various authors have cautioned that too low a pH (below 6.0) would promote metal ion mobility. This doesn't seem to have occurred in the Lake Hills soil which implies other metal binding mechanisms are at work that aren't as pH sensitive, or perhaps a seasonal lowering of soil pH occurs at Lake Hills which corresponded to the sampling period. The author feels the latter is more plausible, but in any event the native soil is naturally acidic, and yet no increased heavy metal ion mobility was apparent.

The element phosphorus appeared to have also been readily bound in the soil judging from the sample profiles, but enrichment calculations showed about twice the phosphorus applied. Some losses could be expected due to crop uptake and leaching. The latter mechanism might be significant due to the

TABLE 2

VALUES OF SOIL ENRICHMENT

Parameter	Mean Irrigated (metric tons) 30 cm/60 cm	Mean Control (metric tons) 30 cm/60 cm	Enrichment (metric tons) 30 cm/60 cm
Lead	0.7/0.2	0.4/0.1	0.3/0.1
Zinc	9.8/4.1	3.0/2.2	6.8/1.9
Copper	3.8/1.7	1.3/0.7	2.5/1.0
Chromium	3.7/1.8	2.2/1.6	1.5/0.2
Phosphorus	240/130	64/27	176/103

Note: Values are in total tons for 30 cm of depth over 18.63 ha.

TABLE 3

COMPARISON OF LITERATURE VALUES

WITH LAKE HILLS DATA

	Wastewater Applied mg/l				Soil (1st 30 cm) ug/g			
Element	Lake Hills	1	2	3	Lake Hills Irrigated	4	5	Lake Hills Control
Cu	.28	.02 to 5.9	0.07	0.45	6.7	20	1.0	2.7
Zn	.37	.02 to 20	0.20	1.74	16.6	50	3.1	6.3
Pb	.05	.02 to 20	0.14	0.04	1.2	10	3.3	0.8

Source Key:

(1) Menzie and Chaney,[10] Waste characteristics.

(2) Sidle, Hook, and Kardos.[9]

(3) Dangel,[7] Values for "standing tap water".

(4) Allaway,[11] Common soil values without wastewater.

(5) Same as (2), except for 5 cm/wk plots, extracted with 0.1 N HCl.

TABLE 4

ESTIMATED MASS LOADING VERSUS ESTIMATED ENRICHMENT

Parameter	Total Lifetime Mass Loading (metric tons)	Estimated Enrichment (metric tons)
Lead	1.45	0.4
Zinc	11.0	8.7
Copper	8.3	3.5
Chromium	0.6	1.7
Phosphorus	107	279

Note: Values represent total estimates mass applied or enriched during the life of the sprayfield of 18.63 hectares in area.

low soil pH, but no successful leachate samples were collected to verify or deny the significance of leaching.

The chromium analyses showed no clearly definable pattern. Whereas the uppermost samples indicated a definite enrichment, the next two deeper samples in the profile, 5 and 10 centimeters, showed the control soils increasing and irrigated plots decreasing. The remarkable difference in the chromium profile versus the other metals may have been due to problems in the sampling or analytical procedures, but the lead, which paralleled the chromium in every way throughout the sampling process, showed no such scattering of values. The enrichment of chromium appears to be low relative to backround levels making patterns difficult to detect.

CONCLUSIONS

In general, the Lake Hills sprayfield was a success when viewed as a treatment and disposal process. It appears to have sustained adequate operation and removal of pollutants in the face of extreme overloads and lack of maintenance that would not be tolerated in conventional operations. Credit for the success seems to rest mostly with the choice of soil, the Lynden series, although the records are not clear whether this was a product of engineering judgment or political availability. The sandy nature of the soil allowed the system to equilibriate to a quasi-overland flow system when heavy sprinkler applications clogged the soil in the immediate vicinity of the applicators. It appears that the system removed heavy metals and phosphorus in excess of removals by secondary treatment, while building the soil organic levels and growing a lush hay crop.

Depression of soil pH such as was found at Lake Hills is a constant concern voiced with regard to metals toxicity. One of the corrective techniques is liming. Adjusting the soil acidity at Lake Hills would cost approximately $78,200. This is a significant expense and efforts should be made to accurately verify the need, as most soils on the western side of Washington State are naturally acidic. Further, liming as an acidity control aimed at metals retention has the drawback that in high rainfall areas, such as western Washington State, natural forces are constantly at work leaching out buffering compounds and lowering the soil pH. If a soil pH is raised with lime, and metals are stored in the upper layers as carbonates or phosphates, they may be released in rather large quantities to crops by the mere act of cessation of continual lime additions. In other words, the soils would require perpetual liming and monitoring. It appears wiser to approach the metals question at the source, such as pipe corrosion in the case of Lake Hills. Addition of lime to the water supply would control corrosion products, an economic benefit to every homeowner, and lower metals levels in the effluent and sludges. At that point, application rates or field life could be adjusted to match soil conditions, i.e., lower with lower pH. Results of this study suggest significant metals retention capacity even at relatively low pH. More data on metal-soil-crop relationships are required to utilize a metals balancing approach, however.

It appears from observations at the Lake Hills site that spray irrigation will become overland flow whenever soil infiltration rates are exceeded due to rain, effluent peaking conditions, organic overloading or a combination. This situation need not constitute a failure other than by definition, as overland flow has been reported to be an effective treatment process on its own, especially with regard to biochemical oxygen demand and suspended solids. If buffer strips are to be required to help minimize risk from pathogens contained in aerosols, these strips can also serve as a backup process in the overland flow mode. Presumably, runoff interception and erosion control would be part of the design in any event, but locating these facilities on the outside of the buffer zone may allow savings by reducing storage requirements by utilizing flood irrigation or overland flow of buffer zones for storage. One key concern is the susceptibility of the crop to soil flooding. Generally, perennial grass crops are more tolerant and would lend themselves better to this concept than an annual such as corn.

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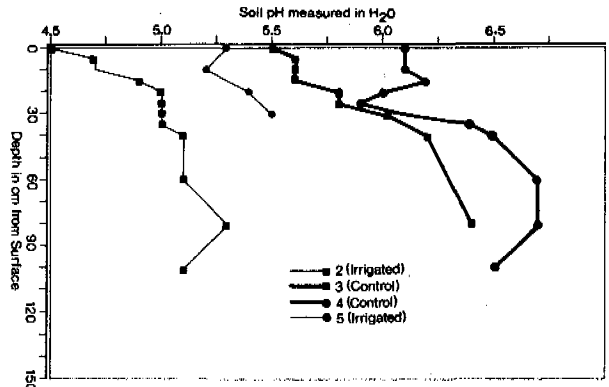

Figure 1
Soil pH vs Depth

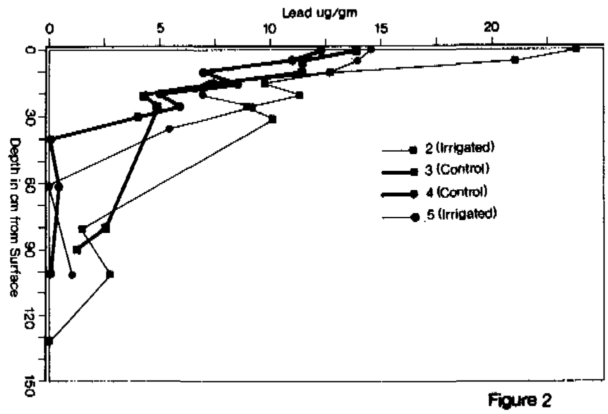

Figure 2
Lead vs Depth

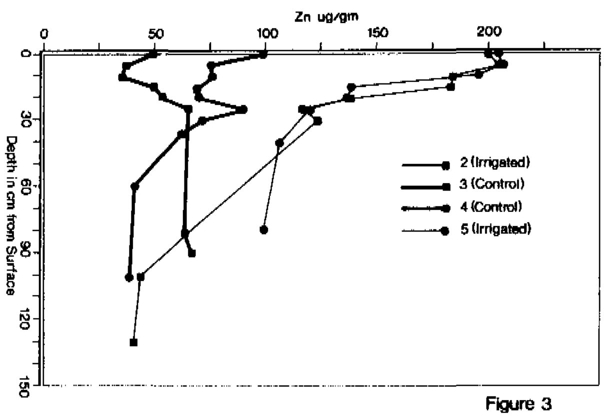

Figure 3
Zinc vs Depth

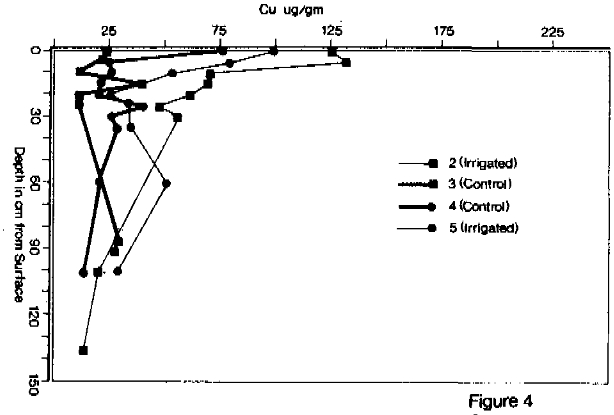

Figure 4
Copper vs Depth

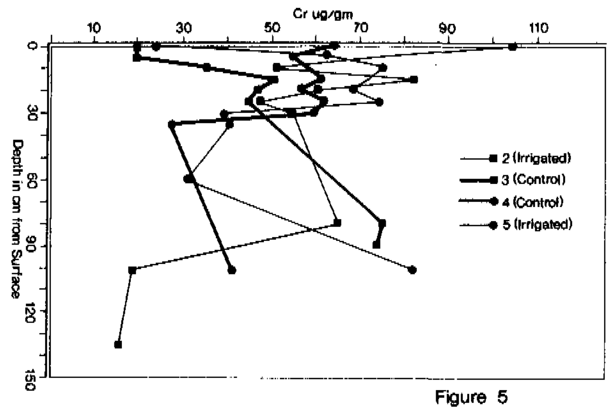

Figure 5
Chromium vs Depth

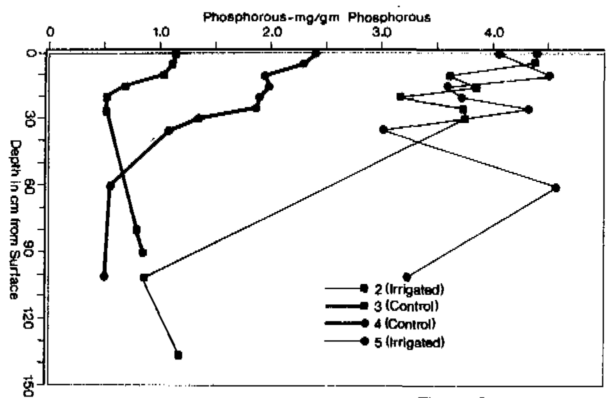

Figure 6
Phosphorous vs Depth

CASE STUDIES

DEMONSTRATION PLANT EVALUATION OF AN INFILTRATION-PERCOLATION SYSTEM FOR BOULDER, COLORADO

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This paper reviews the treatment performance of a demonstration infiltration-percolation system operated by the City of Boulder Sewage Treatment Facility. The study investigated the seasonal and climatic trends observed in the treatment efficiency of secondary effluent by the high-rate land treatment system during the first continuous year of operation.

The facility consisted of an infiltration-percolation basin of approximately 0.9 acre. Unchlorinated secondary wastewater effluent was applied twice a week at a loading rate of 100 ft/yr. The applied wastewater percolated through shallow loamy and clay sands covering alluvial sands and gravel. It was then collected 8 to 10 feet below the surface by underdrain piping and discharged into Boulder Creek.

The treatment performance of the infiltration-percolation system varied somewhat between summer and winter operation. The soil system demonstrated good capability for reducing the concentration of phosphates applied in the secondary effluent. However, maximum phosphate levels of about 1.5 mg/ℓ-P did leach through the system during the winter. Consistent removal of the majority of the refractory organic materials was observed throughout the study period. The maximum effluent COD concentration of 44 mg/ℓ occurred during the winter, and coincided with an abnormally high level of organics in the applied wastewater at that time. Organic and ammonium nitrogen removal by the infiltration-percolation basins was nearly complete. However, maximum leaching of ammonium through the soil also occurred during the winter. A nitrogen balance indicated that the basins stored ammonium nitrogen during the winter and discharged high levels of nitrate nitrogen in the spring.

INTRODUCTION

High rate infiltration-percolation is one of the treatment methods under consideration by the City of Boulder, Colorado, for upgrading the quality of wastewater to meet more stringent discharge requirements. As such, a pilot study was undertaken in the spring of 1976 to assess the capability of infiltration-percolation for providing efficient and reliable treatment in the context of the soil and climatic conditions which characterize the Boulder area. This paper discusses the performance of one infiltration-percolation bed during the first full year of steady state operation.

The City of Boulder is located 40 km (25 miles) northwest of Denver and is situated at the base of the Eastern Slope of the Rocky Mountains. The elevation of Boulder is 1655 m (5430 ft), and the climate is best described as semiarid. Inherent in this type of climate are lengthy periods of low humidity and little precipitation. The average annual precipitation in Boulder is approximately 48 cm (18.9 in.). Although the average annual temperature is about 11°C (52°F), large diurnal variations in temperature typically occur. The growing season generally extends from May 9 to October 8, with an annual average of 152 days above

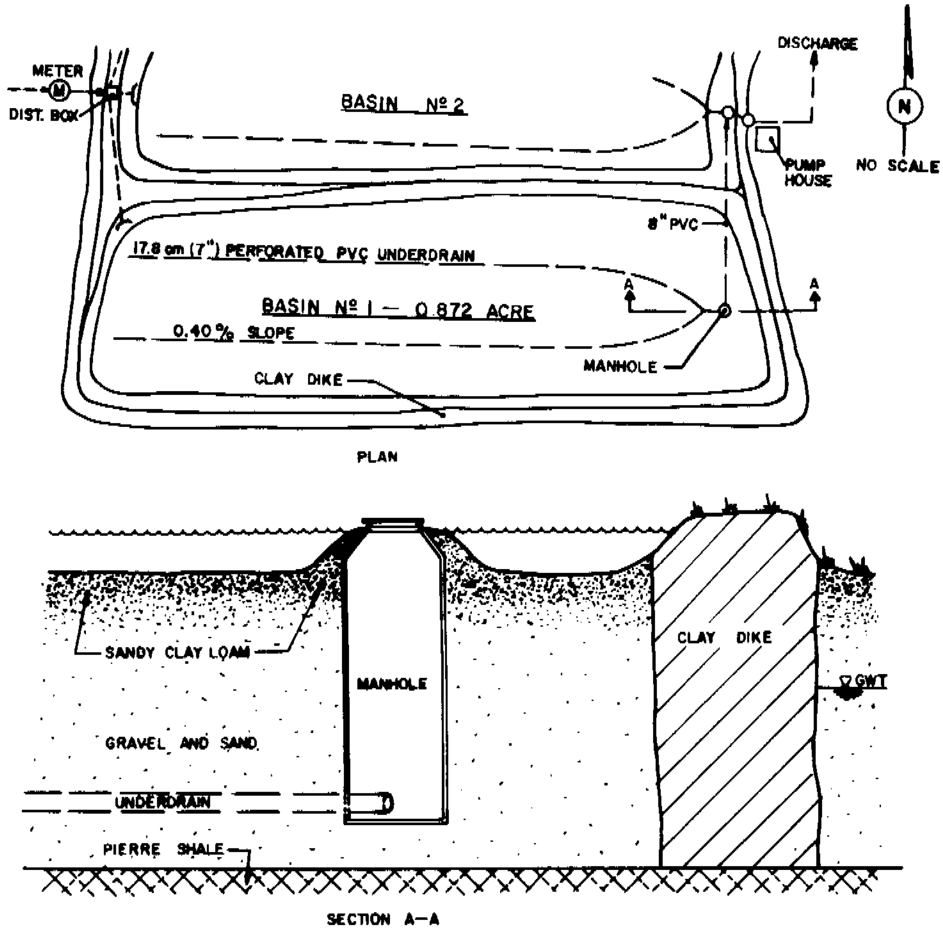

Figure 2 - Basin Layout

three days following loading. Sampling and analysis was performed during the first loading of each week. Samples were collected for analysis of the nitrogen forms at nine different times within the three days following basin loading, while the remaining constituents were monitored only once, at 24 hours after loading. Sampling took place after the percolate was collected by the underdrain system and prior to its pumped discharge to Boulder Creek.

System Performance

In evaluating the performance of the infiltration-percolation system, it was found that continuous operation of the basin was possible throughout the year. However, periods of freezing temperatures did produce a unique cycle within each loading sequence. During these periods, a layer of ice formed on the surface subsequent to basin loading. As the infiltration continued under the ice layer, the ice would eventually collapse and break apart. During the following basin loading the broken ice floated to the surface and the cycle was repeated.

Some seasonal variations were observed in both the system hydraulics and the treatment efficiency. Most of these observed differences were probably directly related to temperature effects, although seasonal variations in the influent wastewater characteristics, such as those indicated in Table I, may also have had some impact on the observed product quality variations. These variations have been presented and discussed for each of the measured wastewater constituents in the following sections of the paper.

Nitrogen. A complete analysis was performed for the typical nitrogen forms in the applied effluent and renovated water. The majority of the applied nitrogen existed in the ammonium and organic nitrogen forms. The annual average total nitrogen concentration applied to the basin was 16.5 mg/ℓ-N, of which 7.6 mg/ℓ was in the ammonium form, and 7.0 mg/ℓ in the organic nitrogen form. The seasonal variations of the applied nitrogen forms were

$0^{o}C\,(32^{o}F)$. Weather patterns occurring in this high plains region are basically dominated by four air masses. Cold artic air masses are responsible for short periods of extreme cold often experienced during the winter months. Periods of increased precipitation during the spring result from moist air masses which develop in the Gulf of Mexico. The summer months are commonly influenced by warm dry air from Mexico and the desert Southwest, causing periods of extreme warmth and dryness. However, the general weather patterns are influenced by moist Pacific air masses which move easterly across the Rocky Mountains.

DESCRIPTION OF SYSTEM AND OPERATION

The small infiltration-percolation basins evaluated in this study were constructed adjacent to Boulder Creek at the 75th Street Treatment Plant in Boulder. Since only one of three basins was operated continuously throughout the study period, discussion in this paper has been limited to the operation and performance of this basin. The 75th Street Treatment Plant is a 5.68×10^7 liter/day (15 MGD) secondary treatment system which consists of a standard rate trickling filter followed by chlorination. The influent to the infiltration-percolation basin was pumped from the secondary clarifiers prior to chlorination. A schematic flow diagram for the system is illustrated in Figure 1.

The demonstration infiltration-percolation basin was 0.872 acre in area and was comprised of a sandy clay loam surface which varied in depth from 38-66 cm (15-26 in.). The infiltration rate of this soil averaged 0.53 in/hr over the basin surface. The underlying soil consisted of well rounded gravel mixed with sand and silt loam to an approximate depth of 3.7 m (12 ft). This gravel layer was underlain with an impermeable shale layer. A clay dike was constructed around the perimeter of the basin to isolate the basin from the surrounding groundwater. This dike extended from the bedrock to 0.75 m (2.5 ft) above the ground surface. The normal groundwater table was typically 0.9-1.5 m (3-5 ft) below the ground surface in the area of the basin, but was lowered to approximately 2.4 m (8 ft) within the basin area by two 17.8 cm (7 in.) underdrains located at the 2.4 m (8 ft) depth. Figure 2 illustrates the basin layout in plan and cross-section.

Basin loading occurred twice each week for a six-week period, followed by a drying and scarification period between each loading cycle. The required length of the drying period between loadings was generally one week in the summer and autumn, and two weeks during colder periods. Each loading inundated the basin to a 30.5-35.5 cm (12-14 in.) depth, which was equivalent to an annual loading rate of 30.5 m (100 ft). All of the ponded water on the basin surface usually infiltrated the soil within

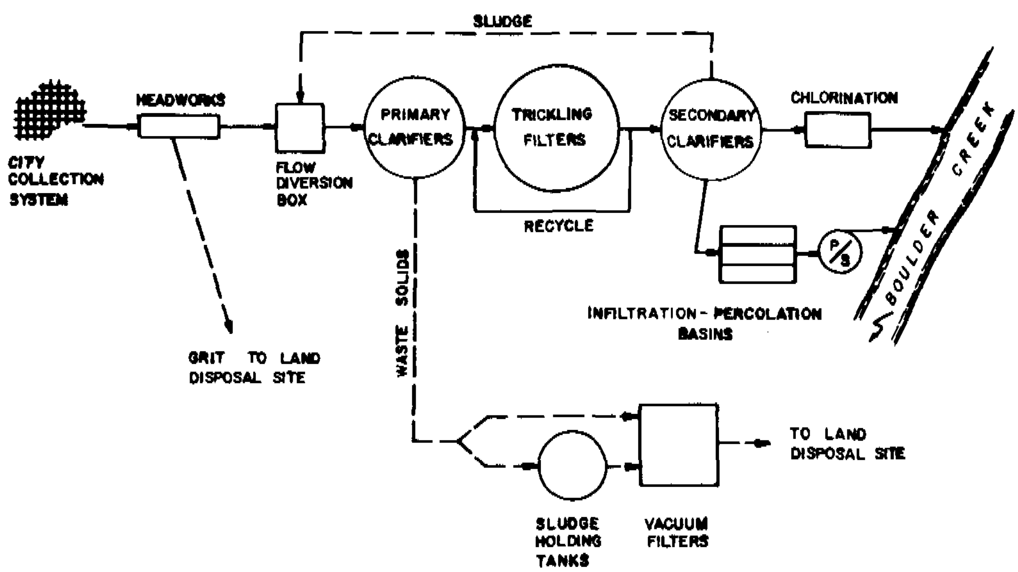

Figure 1 - Schematic Flow Diagram for the 75th Street Treatment Plant

Table I

Seasonal Averages for Constituent Concentration in Boulder Secondary Effluent Applied to Infiltration-Percolation Basins from 7/12/76 through 8/25/77

Constituent	Spring	Summer	Fall	Winter	Annual
Total Nitrogen[1]	13.8	13.2	14.6	24.5	16.5
Ammonia	7.3	6.0	5.3	11.7	7.6
Organic Nitrogen	4.7	4.7	6.6	12.2	7.0
Nitrate	1.8	2.5	2.7	0.6	1.9
Total Phosphate[2]	4.8	5.1	7.2	7.9	6.2
COD	58.0	58.1	72.1	118.3	76.6
Alkalinity[3]	128	118	125	164	134
Hardness[3]	132	154	119	112	129
Calcium	34.6	35.6	-	32.2	34.1
Magnesium	11.0	14.8	-	7.4	11.1
Chlorides	27.4	23.2	32.1	27.3	27.5
Suspended Solids	14.6	11.7	21.6	25.4	18.3
Total Solids	311	333	286	302	308

[1]All nitrogen concentrations in mg/ℓ as N
[2]mg/ℓ as P
[3]mg/ℓ as $CaCO_3$

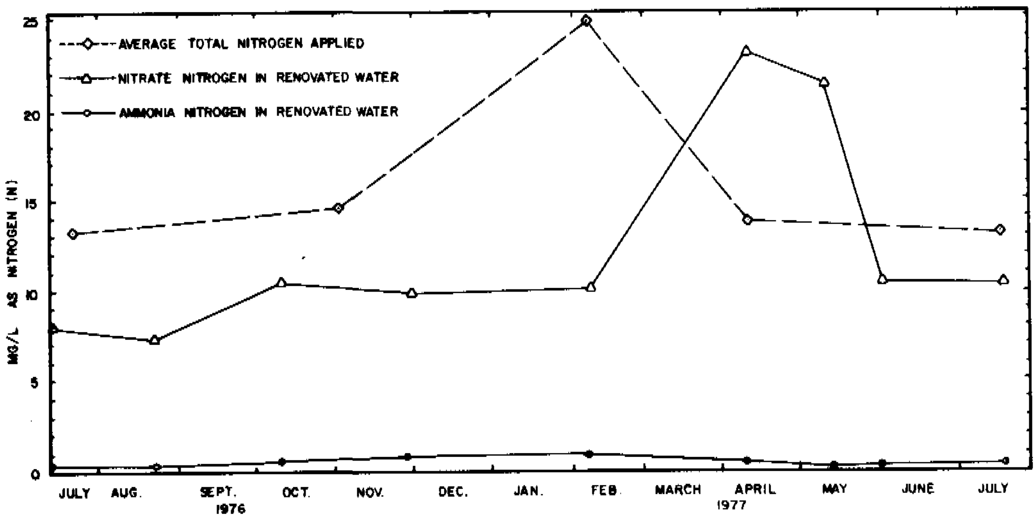

Figure 3 - Basin No. 1 Performance - Nitrogen

included in Table I. The nitrogen present in the renovated water was primarily in the nitrate form, with low levels of ammonium also detected. The organic nitrogen and nitrite concentrations were generally less than 0.5 mg/ℓ-N and 0.10 mg/ℓ-NO_2^--N, respectively. Figure 3 illustrates the nitrification performance of the basin by comparing the concentrations of the total nitrogen applied with the concentrations of nitrate and ammonium detected in the renovated water. Each point on the figure represents the average concentration of a six-week loading cycle, and corresponds to the initial loading date of the cycle. The nitrate accounted for 98% of the total nitrogen discharged from the system. A nitrogen balance across the infiltration-percolation system indicated that only a 9% reduction in nitrogen occurred during the study period (St. John, 1977). This reduction was likely associated with denitrification and volatilization since the basin surface was operated without vegetative cover.

Significant seasonal variations were observed in the concentrations of both the applied and discharged nitrogen.

The influent variations were indicated previously in Table I, and Figure 3 illustrated the seasonal variations in the ammonium and nitrate concentrations of the renovated water. The highest ammonium concentrations measured in the renovated water occurred during the two coldest loading cycles, and reached levels of as high as 1.89 mg/ℓ NH_4^+-N. The secondary effluent applied to the bed also contained the highest levels of ammonium during this same period, and averaged 4.1 mg/ℓ mg/ℓ-N above the annual average. The higher concentrations of ammonia detected in the renovated water during winter operation were likely related in some part to the higher concentrations in the applied secondary effluent. In addition, the typical winter operation cycle hindered the oxidation of the ammonium ions retained in the soil. The transport of oxygen into the soil was restricted by both the ice layer described earlier and shorter drying periods which occurred between loadings during the colder cycles. This reduced oxygen availability, together with the temperature related reduction in the nitrification rate limited the rate of recovery of the ammonium adsorption capacity within the soil matrix. As a result, the soil became more heavily loaded with ammonium ions and some leakage resulted.

Although the magnitude of the ammonium leakage was not large enough to affect the average nitrate concentrations during the cold weather period, a long term accumulation of fixed ammonium ions resulted in extremely high nitrate discharges in the spring, as is indicated in Figure 3. The ammonium accumulation is shown in the curves of Figure 4, which indicate a cumulative nitrogen balance for the first year of basin operation. The cumulative nitrogen storage, or loss, on any date is simply the difference between the application and discharge lines of this figure. Increased storage during the winter is illustrated by the larger difference between applied and discharged nitrogen during these months.

The summer and autumn fluctuations in the nitrogen levels of both the applied and renovated water were less pronounced, which indicated that the capacity of the soil to absorb and biologically oxidize the ammonium was not fully taxed during these periods.

Loss of nitrogen through denitrification or volatilization was very limited in the system. However, variations of the system management and operation to optimize nitrogen removal were not practiced during this period of investigation.

Phosphorus. Phosphorus removal during the study period was generally excellent, with renovated water phosphorus concentrations consistently less than 1.5 mg/ℓ PO_4-P. Figure 5 illustrates the trends observed in the total phosphorus levels of the applied effluent and renovated water throughout the study period. A distinct seasonal trend was observed as is indicated in this figure. The concentration of total phosphorus appearing in the renovated water increased significantly during the winter loading cycles when the wastewater

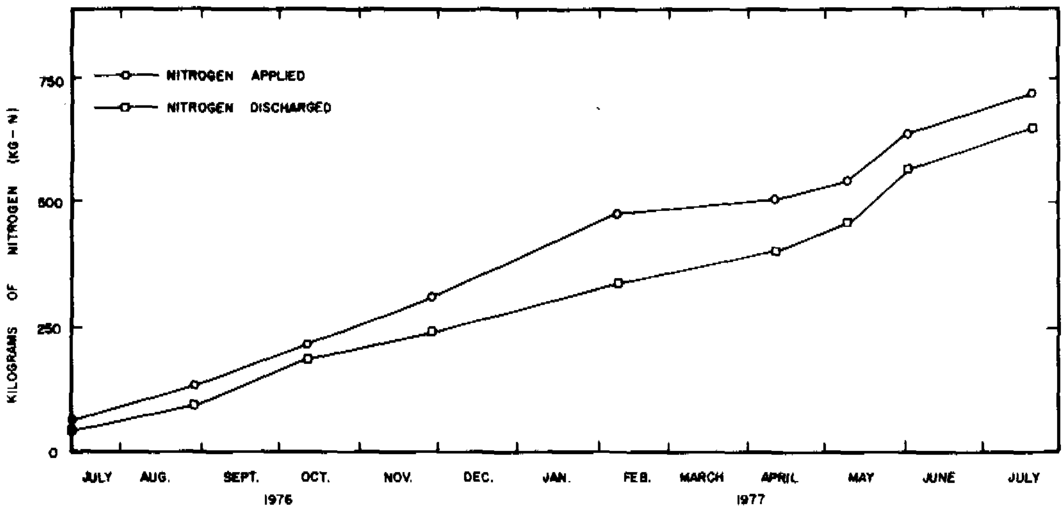

Figure 4 - Cumulative Nitrogen Applied and Discharged from Basin No. 1

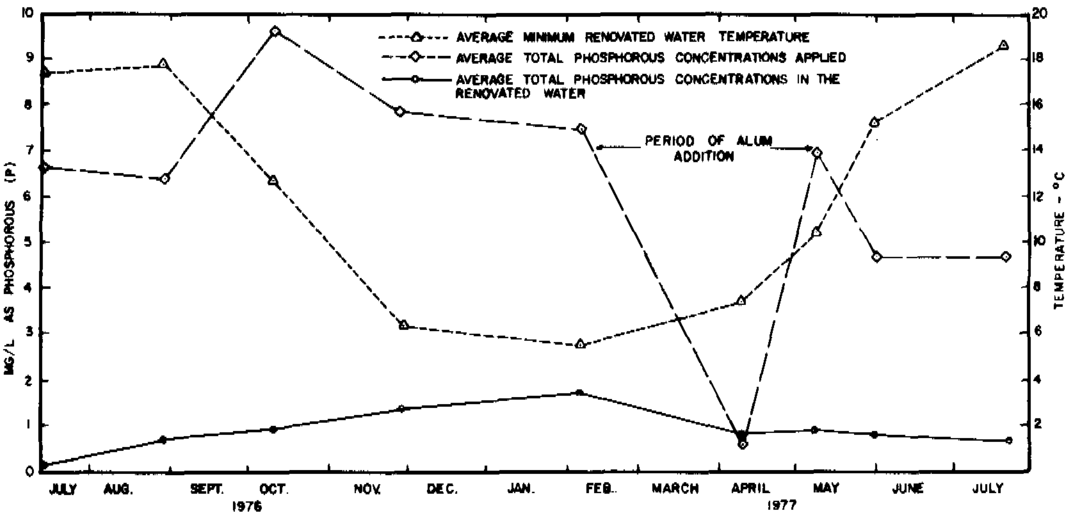

Figure 5 - Basin No. 1 Performance - Phosphorus

temperatures were the lowest. Phosphates introduced to the soil system are initially adsorbed in the mineral or organic fraction of the soil (Harter, 1969; Hsu, 1964). The adsorbed phosphate is subsequently precipitated or mineralized as inorganic compounds into the soil matrix by a slower reaction. This precipitation reaction serves to release the adsorption sites within the soil for additional phosphorus removal. The adsorption process will control the rate at which phosphorus can be removed from the soil until the soil matrix becomes saturated, at which point the slower precipitation reaction will limit the rate of phosphorus removal (Barrow, 1975).

The increase in phosphorus levels applied to the bed during the autumn and winter apparently saturated the phosphorus adsorption capacity within the soil system. At this point, the slower precipitation reaction effectively controlled the rate and effectiveness of the phosphorus removal process.

Organics. The removal of wastewater organics in the infiltration-percolation system was monitored by determining the chemical oxygen demand (COD) of the applied wastewater and that of the renovated water 24 hours after loading. The average COD of the applied effluent and bed effluent has been plotted for each six-week loading cycle in Figure 6. Included with these data are the average removal efficiencies for each cycle. As can be seen from these data, the infiltration-percolation system was generally effective in reducing the wastewater organic concentration by about 70-80%. The increase observed in the renovated water COD during periods of colder temperatures was accompanied by higher applied effluent levels. These high influent COD concentrations were attributed to highly concentrated industrial waste discharges which appeared in the Boulder sewage during that time.

The apparent decline in COD removal efficiency during the spring months was due to the unusually low COD levels in the applied effluent. As illustrated in Figure 6, the renovated water COD levels remained low during this period. Alum was used from March 5 to May 5, 1977 to facilitate settling in the secondary clarifier. This alum addition resulted in average COD values of 43 mg/ℓ as compared with a seasonal average of 58 mg/ℓ (St. John, 1977).

Suspended Solids. Excellent removal of suspended solids was experienced throughout the operation of this system. Although the suspended solids in the applied effluent increased during the winter months, the effect on the renovated water was minimal. Figure 7 illustrates this observation, and shows little or no suspended solids present in the renovated water. Again, the points in the figure were averages of a six-week loading cycle plotted as a function of the initial loading date.

Microorganisms. Total and fecal coliforms were monitored on a grab sample basis to determine the effectiveness of the infiltration-percolation system for bacterial removals. Although the removal efficiencies averaged 96.2% through the study period, the fecal

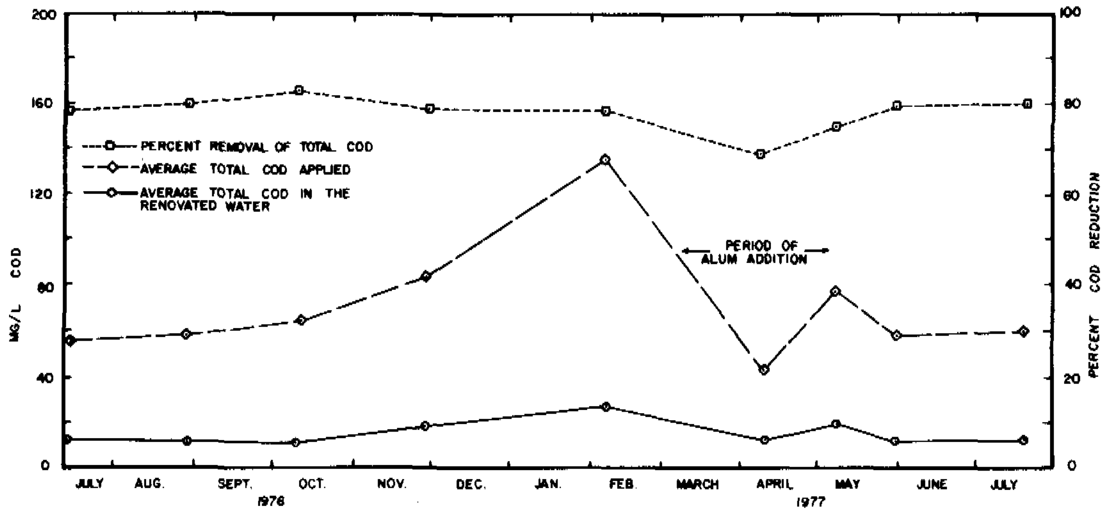

Figure 6 - Basin No. 1 Performance - COD

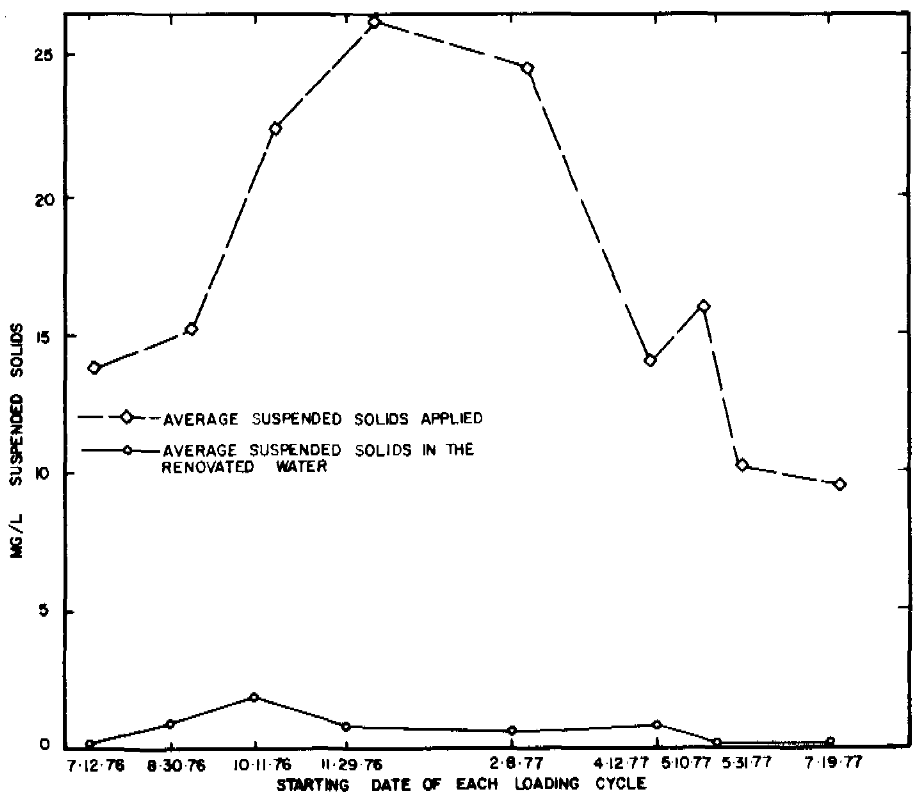

Figure 7 - Basin No. 1 Performance - Suspended Solids

coliform count in the renovated water varied considerably. Fecal coliform organism counts of as high as 10^3/100 ml appeared in the renovated water at times. From these data it is clear that significant bacterial leakage can occur in high rate soil treatment systems.

SUMMARY AND CONCLUSIONS

The infiltration-percolation system evaluated in this study was shown to be very effective in reducing the concentrations of many important wastewater constituents found in secondary effluents. The suspended solids and coliform organism concentrations were consistently reduced by over 96%.

Significant phosphorus and COD removals were also attained, although the leakage of these constituents was somewhat more pronounced. Organic and ammonium nitrogen were also efficiently removed, with subsequent oxidation and release of nitrate nitrogen. These removals and transformations were effected throughout the year with relatively minor seasonal fluctuations.

ACKNOWLEDGEMENTS

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CASE STUDIES

NITROGEN AND PHOSPHORUS REMOVAL AFTER 30 YEARS OF RAPID INFILTRATION

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ABSTRACT

Groundwater and soil nitrogen and phosphorus concentrations were measured beneath a rapid infiltration site that has received primary municipal effluent for 30 years. Similar measurements were taken at adjacent locations and used as controls. After percolation through 7 m of unsaturated gravelly and sandy loams, total nitrogen values in the underlying aquifer approached those recorded at offsite control locations. Influent values of 40.2 mg/L total-N decreased to less than 4 mg/L in the shallow aquifer. Nitrate-N did not pose a pollution hazard. Effective phosphorus removal required longer travel distances, but the sorption capacity of the soil had not been exhausted even after 30 years of continuous wastewater application.

INTRODUCTION

The success of a rapid infiltration facility is measured in part by the soil's ability to prevent the migration of wastewater applied nitrogen and phosphorus into the underlying groundwater. Specific questions regarding the fate of these nutrients still remain unanswered because most systems have been monitored for only short periods of time. Evaluation of long-term treatment efficiency is therefore necessary to sharpen design criteria for widespread use and estimate the environmental impact of land application of wastewater.

With the above objective in mind, an investigation was conducted at a rapid infiltration site that has received primary effluent for over 30 years. Soil chemistry and groundwater quality data obtained at this site were compared with equivalent data from a nearby control site which had never received wastewater. A total of 23 soil parameters and 40 groundwater parameters were monitored. This discussion, however, focuses primarily on the behavior of nitrogen and phosphorus.

SITE DESCRIPTION

The City of Hollister is located 35 km inland from Monterey Bay and 144 km south of San Francisco. Its climate is marked by warm dry summers and cool rainy winters. Mean annual precipitation is 33.4 cm, with greater than 19 cm occurring from December to February. The highest average temperature occurs in July (20°C). Mean annual temperature is 15°C.

A facilities plan for the Hollister rapid infiltration site is presented in Figure 1. Clarified effluent flows by gravity to a series of 20 individually controlled infiltration basins ranging in area from 0.3 to 0.7 ha. Total spreading basin area equalled 8.8 ha. Average daily flow was 43.8 litres per second (L/s). Wastewater was applied to a depth of 30 cm in each basin on a rotating basis; measured infiltration rates varied among basins and ranged from 4.6 to 31.7 cm/d with an average of 17.7 cm/d. The basins were allowed to dry for periods from 14 to 21 days prior

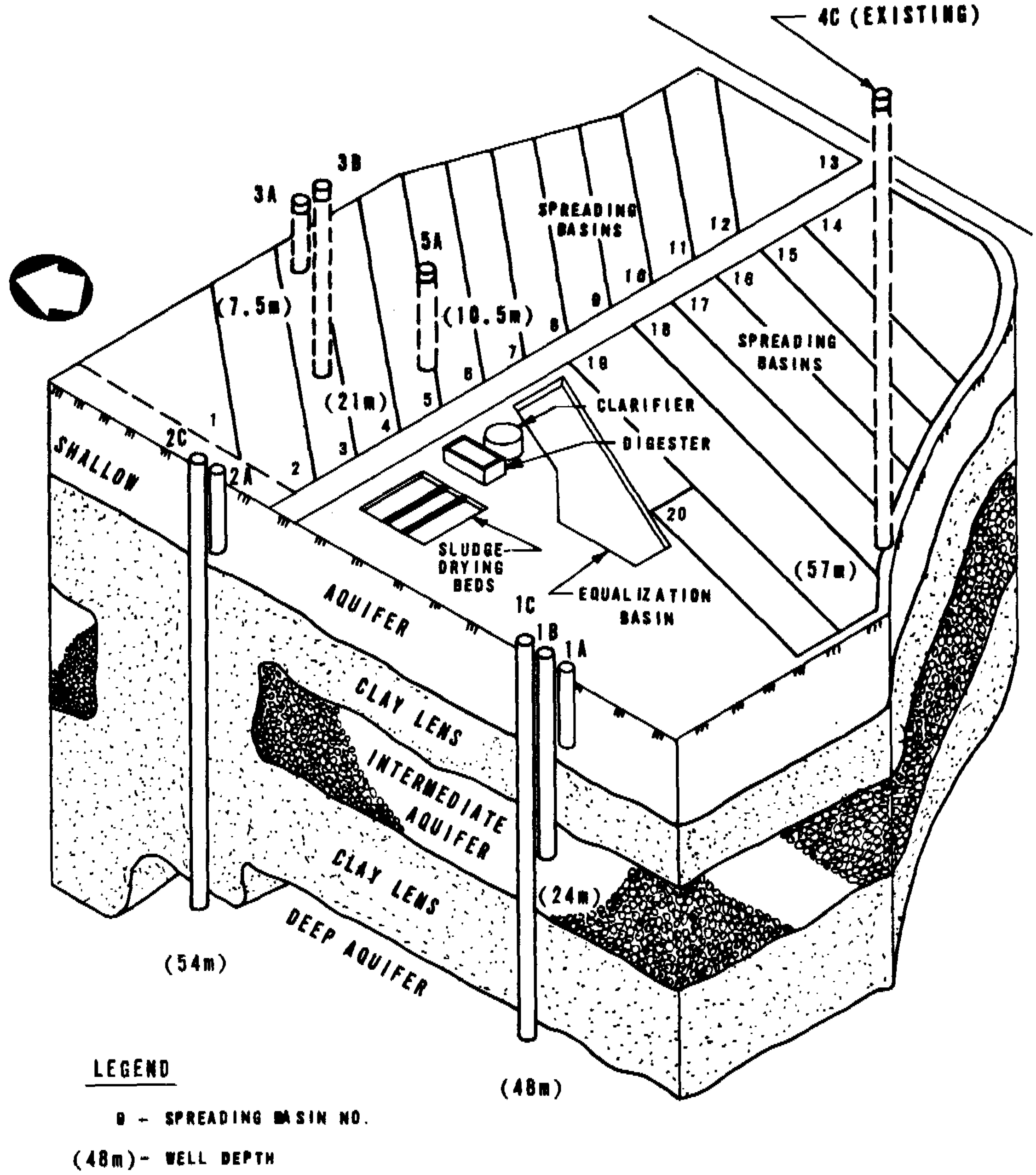

Figure 1. Hollister Rapid Infiltration Site Layout

to reapplication. Effluent application over the entire rapid infiltration site was 30 cm per week (or 15.6 metres per year). All basins were free of vegetation and regularly disked to restore surface infiltration capacity. Sludge from the clarifier was drawn off and applied to drying beds which are separate from the rapid infiltration area.

The soil at the Hollister site is mapped as a Metz sandy loam (Typic Xerorthent) [1], an alkaline soil (pH 8 to 9) underlain by stratified calcareous sands and gravels. The terms gravelly sand or gravelly sandy loam, though, often offer a more appropriate description of soil texture.

The principal water bearing units beneath the site are lenticular beds of sands and gravels interbedded with clay (Figure 1). Preliminary data [2] suggested that groundwater flowed in a predominantly westerly direction.

Table 1. Average Effluent and Groundwater Quality Results
mg/L Except as Noted

			Groundwater[a]											
							Deep[b]							
	Effluent		Shallow[c]		Intermediate[d]				Offsite control wells					
Constituent	Range	Average[e]	3A	5A	1B	3B	1C	2C	4C	6C	7C	8C	9C	Significant contrasts[i]
COD	546-1,029	706	46	50	21	24	18	127[f]	16	14	14	21	30	
BOD	134-414	220	6	13	3	4	5	16	3	<6	5	4	1	
TOC	240-264	248[g]	10[h]	11[h]	9[h]	<1[h]	13[h]	20[h]	<1[h]	--	6[h]	--	5[h]	
Total N	29.7-58.5	40.2	3.8	1.7	6.7	7.8	2.9	1.8	2.1	0.2	6.0	4.1	9.9	E-S, E-I, E-D, E-C, D-I, I-S
NH_4-N	19.7-44.0	25.3	<0.4	<0.1	<0.2	<0.1	0.4	0.3	<0.1	0.2	<0.2	<0.2	<0.05	E-S, E-I, E-D, E-C
Org-N	6.7-21.8	14.5	2.2	1.0	0.5	0.8	2.4	1.1	<0.5	0.2	<0.5	<0.2	0.4	E-S, E-I, E-D, E-C, C-S, I-S
NO_3^--N	0.16-0.8	0.43	1.2	0.6	6.1	7.0	0.1	0.5	1.9	0.05	5.6	3.9	9.5	E-I, E-C, C-S, C-D, D-I, I-S
Total P	10.0-21.5	12.4	8.0	9.6	0.1	0.2	0.1	0.1	0.1	0.04	0.03	0.1	0.1	E-S, E-I, E-D, E-C, C-S, C-I, D-S, I-S
PO_4-P	9.0-13.2	10.5	6.8	8.7	0.1	0.1	0.1	0.03	0.04	0.04	<0.02	0.06	<0.01	E-I, E-D, E-C, C-S, D-S, I-S
pH, units	7.0-8.1	7.3	7.5	7.7	8.2	7.6	8.0	7.9	7.8	7.2	7.7	7.6	7.9	

a. Average of four grab samples.
b. Depth of well screen = 48 m.
c. Depth of well screen = 7.5-10.5 m.
d. Depth of well screen = 19.5-24.0 m.
e. Average of twelve 24 h composite samples.
f. Contamination by COD exerting drilling mud.
g. Average of three 24 h composite samples.
h. Single grab sample.

i. Pairwise contrasts, t-statistic, $\alpha = 0.05$
E = Effluent
S = Shallow wells
I = Intermediate wells
D = Deep wells
C = Control wells.

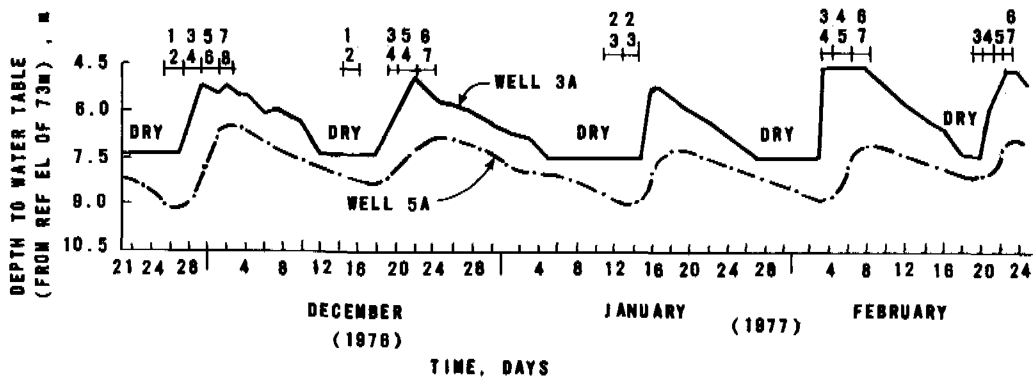

Figure 2. Water Level Response to Basin Flooding in Observation Wells 3A and 5A

EXPERIMENTAL PROCEDURES

Effluent was sampled quarterly from June 1976 through March 1977. A battery operated, refrigerated composite sampler was installed to sample from the effluent trough of the primary clarifier. Each sampling period produced seventy-two 1-hr samples that were composited according to flow. The samples were analyzed according to Standard Methods [3].

Groundwater quality was monitored at a series of observation wells (Figure 1). One quarterly sample was taken from each well at the time of effluent sampling. The shallow groundwater samples were taken to coincide with maximum groundwater mound height created by wastewater application. Thus, the results represent renovation at minimum travel distance and detention time of the percolate in the unsaturated soil profile. All observation wells were flushed thoroughly prior to sample collection. Samples were analyzed in accordance with Standard Methods [3]. Shallow depth wells were designated "A"; intermediate "B"; and deep "C." Wells numbered 1, 2, 3, and 5 were sited to intercept groundwater flow leaving the basins, while Well 4C was located upgradient and served as a control. Four existing offsite wells, 6C, 7C, 8C, and 9C also served as control wells downgradient of groundwater flow from the site. (Results from these wells were statistically equivalent to that recorded at Well 4C.) These wells were located approximately 0.8 km west of the rapid infiltration site. Wells 3A and 5A were equipped with continuous water level recorders. Shallow depth wells, located at Sites 1 and 2 (Figure 1), yielded no water during the monitoring program.

The soil profile was sampled randomly at three locations that received primary effluent and two offsite control locations. Composite samples were obtained at four depth increments (0-16, 25-35, 95-105, and 295-305 cm) at each sampling location. The soil was air dried, thoroughly mixed and screened through a 2 mm diameter sieve. Analytical procedures for nitrogen and phosphorus are described by Black [4]. All soil data are reported on a whole soil basis.

RESULTS

Wastewater and Groundwater

Characterization of Hollister's primary effluent and groundwater is presented in Table 1. The effluent $COD:BOD_5$ ratio of 3.2 is somewhat higher than expected for medium density residential areas and may reflect input from a paper recycling plant which accounts for 25% of the total flow. Other constituent concentrations are within a range typical of primary effluents [5]. Ammonia nitrogen accounts for greater than 60% of the total nitrogen, and inorganic orthophosphate makes up 85% of the total phosphorus.

Water level fluctuations observed in Wells 3A and 5A are presented in Figure 2. Depth to water table decreased when adjacent infiltration basins were flooded, but both wells remained dry or nearly dry when adjacent basins were allowed to dry. These results suggest that groundwater at this depth resulted exclusively from wastewater application. Further support for this conclusion is provided by noting that shallow Wells 1A and 2A, located at offsite locations in equivalent geological strata, remained dry throughout the study.

Total nitrogen values in the effluent were significantly higher than those reported in any of the groundwater aquifers. Total N in the shallow aquifer was only 11% of input levels. This result becomes even more important by noting that the shallow groundwater is derived solely from wastewater application, eliminating dilution as a treatment mechanism. Thus, despite groundwater sampling at times of minimum vertical travel, substantial nitrogen renovation was achieved. In contrast, total N in the intermediate aquifer was higher than that recorded in either the shallow or deep aquifers. However, total N levels in all three aquifers were statistically indistinguishable from control well values, suggesting that wastewater applied nitrogen has not adversely affected groundwater quality.

Ammonium-N showed statistically significant contrasts when wastewater concentrations were compared to observation or control well values. All aquifers were statistically indistinguishable from control wells and themselves. Input levels of ammonium-N approximating 25 mg/L were reduced to nearly zero after 7 m of vertical travel through unsaturated soil.

All well groups had significantly lower organic nitrogen than applied wastewater. Organic nitrogen was reduced from initial values of 14.5 mg/L to 1.6 mg/L after 7 m of travel. Organic-N in the shallow aquifer was significantly higher than intermediate or control well values. This result conflicts somewhat with total N data, and probably reflects sampling variability rather than nitrogen movement.

The behavior of nitrate-nitrogen was more difficult to evaluate because ambient control well levels were so high. The value of 9.5 mg/L at Well 9C was already close to the recommended maximum of 10 mg/L set for drinking water supplies [6]. The high concentrations of nitrate-N that were observed in the intermediate wells were comparable to offsite control wells, suggesting that the nitrogen in the intermediate aquifer was derived from a source other than surface applied wastewater. Had NO_3 been derived from the infiltration basin area, Wells 5A and 3A would have responded because they were within the groundwater mound created during wastewater application. The mobile nitrate anion [7] would probably be associated with the groundwater mound. In addition, there is evidence for significant denitrification as discussed in the following section.

Total input phosphorus levels were reduced significantly from 12.4 to 8.8 mg/L after percolation through 7 m of soil, a reduction of 30%. Total-P in the intermediate aquifer was also higher than in the control wells. However, the difference is small and may only reflect sampling variability.

Shallow aquifer orthophosphate-P was significantly higher than that recorded in either the intermediate, deep, or control wells, again reflecting the influence of wastewater addition. Although input levels were reduced from 10.5 to 6.8 and 8.7 mg/L at Wells 3A and 5A, respectively, the reduction was not statistically significant. The lack of significance probably indicates sampling variability rather than a lack of continued treatment, as is discussed later. In addition, the ratio PO_4-P/total-P is comparable in both the wastewater and shallow aquifer, suggesting that selective removal of inorganic P relative to organic P was not occurring.

Soils

The results of the soil sampling program are presented in Table 2. The vertical distribution of soil nitrogen and phosphorus is shown in Figure 3. A two factor analysis of variance was performed for both nitrogen and phosphorus species and indicated that depth and treatment represented significant ($\alpha = 0.05$) sources of variation in all cases. A depth-treatment interaction was observed for all parameters except total-P.

Both nitrogen components exhibited significant concentration increases as a result of wastewater treatment with selective enrichment occurring at the soil surface. Total-N and organic-N were closely related in both trend and magnitude. The highest concentrations of total-N occurred at the surface where values of nearly 1,500 and 700 ppm were observed for treatment and control sites, respectively. The respective values decreased by factors of one-fifth and one-tenth at 300 cm depth. Organic-N accounted for 80% or more of the total-N throughout the soil profile, although it accounted for only 40% of the total-N in the wastewater, suggesting that organic-N was being preferentially bound by the soil.

A comparison of total-N input with the mass of total soil nitrogen currently in excess of background levels revealed that the surface 300 cm of soil accounted for only 2% of the total-N applied over the 30 year period. This implies that much of the nitrogen had been converted to mobile forms and no longer existed within the sampled soil profile. Conversion to N_2 or N_2O would result in atmospheric loss, while conversion to NO_3 would result in loss to the underlying groundwater.

In the theoretical denitrification reaction, where glucose is used as a carbon source, 3.2 grams of carbon is required for each gram of nitrogen denitrified (C:N = 3.2:1). Lance and Whisler [8] observed that stabilized municipal wastewater (C:N = 1.2:3) did not contain enough unstablized organic matter to denitrify wastewater applied to soil columns. These laboratory results were verified at the Flushing Meadows rapid infiltration project [9]. On the other hand, it is probable that a high BOD wastewater does denitrify rapidly when applied to the soil. Law et al. [10] reported 83 to 90% removal

Table 2. Results of Soil Chemical Analyses

Depth, cm	pH	Organic matter, %	Nitrogen, ppm Total	Nitrogen, ppm Organic	Phosphorus, ppm Extractable	Phosphorus, ppm Total
Control No. 1						
0-16	8.5	0.76	560	550	13	710
25-35	9.2	0.08	76	62	0.67	610
95-105	9.4	0.02	64	45	0.56	390
295-305	9.4	0.01	45	34	3.7	330
Control No. 2						
0-16	7.8	1.35	840	818	8.4	790
25-35	8.4	0.15	110	98	0.15	460
95-105	8.9	0.02	34	30	0.09	360
295-305	9.1	0.02	180	167	0.17	470
Treatment Site No.1						
0-16	6.3	1.20	1,200	1,100	57	1,600
25-35	6.8	0.20	370	330	61	1,200
95-105	7.9	0.06	120	99	42	720
295-305	8.8	0.11	180	140	54	960
Treatment Site No.2						
0-16	6.4	2.19	1,720	1,600	109	2,000
25-35	6.5	0.12	350	290	73	1,400
95-105	7.8	0.03	93	74	39	790
295-305	8.8	0.08	190	150	32	840
Treatment Site No.3						
0-16	6.7	2.02	1,600	1,500	110	2,800
25-35	7.6	0.28	200	174	63	1,400
95-105	8.0	0.05	104	86	33	920
295-305	8.6	0.03	77	65	21	850

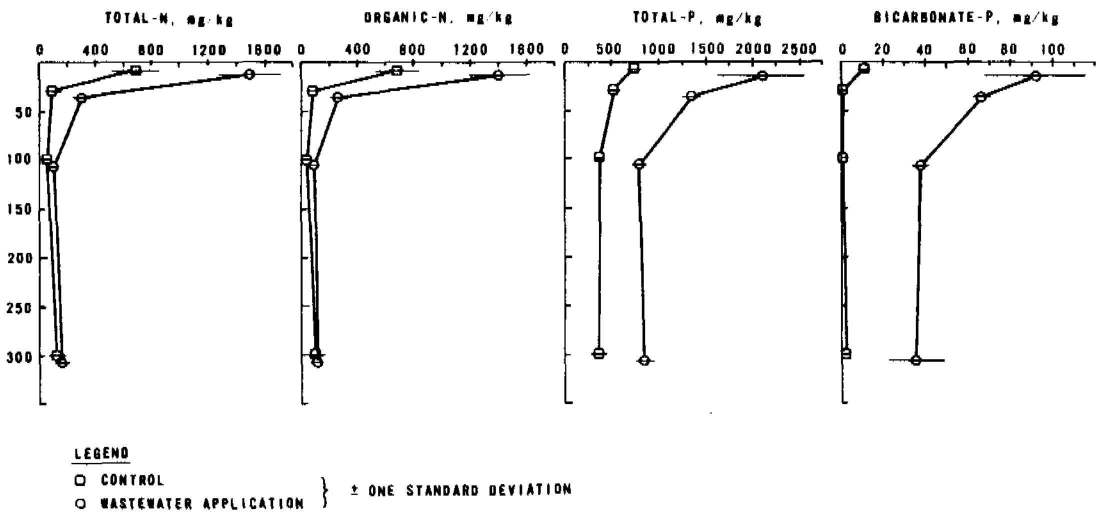

Figure 3. Vertical Distribution of Total Nitrogen, Organic Nitrogen, Total Phosphorus, and Extractable Phosphorus

of total nitrogen from overland flow treatment of high BOD cannery wastes. Lance et al. [11] also demonstrated that with the addition of 150 mg/L glucose (C:N = 5:1), soil columns intermittently flooded with secondary sewage water realized a 90% nitrogen reduction. Nitrogen removal decreased to 60% when the carbon concentration was 80 ppm (C:N = 2.7:1]. At Hollister, the C:N ratio was approximately 6 to 1, a condition favoring denitrification.

Gilmour et al. [12] showed that a flooded surface soil containing 0.9% total carbon denitrified applied nitrate readily without organic amendments. At Hollister, surface organic carbon approximated 1%. Therefore, the zone of most active denitrification is likely to be near the soil surface in spite of its proximity to the atmosphere. This has been demonstrated in field experiments by Rolston et al. [13] who also observed that denitrification took place near the soil surface.

Although the soil pH is slightly lower than optimum for denitrification reactions, the pH is well above limiting values that would inhibit the denitrification process [14].

Groundwater data also present strong evidence for denitrification. Organic-N is generally converted (through NH_4) to NO_3-N without much change in absolute concentration. If nitrification and subsequent loss to the underlying groundwater were occurring, total-N in the shallow aquifer would be expected to approach input total-N. Examination of the groundwater and effluent data (Table 1) reveals that input levels of total-N are more than ten times greater than total-N levels in the shallow groundwater, suggesting that denitrification is the primary mechanism responsible for soil nitrogen loss.

Significant differences between control and treatment sites were also observed for both total and bicarbonate extractable phosphorus. Both phosphorus species accumulated in the surface 100 cm of the soil profile. At the surface, treatment site total-P increased by a factor of about 3 while extractable-P increased by a factor of 10 over control sites. Only 33% of the total applied P was retained in the upper 300 cm of the soil profile, indicating significant transport of wastewater applied phosphorus at an annual wastewater application rate of 15.6 m. Poor phosphorus removal efficiencies were also observed at the Flushing Meadows rapid infiltration site [9, 15].

Rapid infiltration systems require sandy soils that can sustain high water intake rates and high transmissivity in the subsurface environment. Therefore, layers with high phosphorus sorption capacity are not likely to be encountered. In an attempt to determine if the soil at the Hollister rapid infiltration site still retained its ability to sorb phosphorus, solution phosphorus was equilibrated with soil obtained from both control and treatment site locations. Experimental details were modified after Enfield [16]. Results of this experiment are presented in Figures 4 and 5.

Phosphorus sorption in both soils was a function of time and equilibrium concentration. After 288 hours of equilibration, an equilibrium concentration of 12.4 mg/L (the average total-P concentration in Hollister's wastewater) yields 69 and 108 ppm of sorbed phosphorus for the treatment and control site soils, respectively. Thus, even after 30 years of wastewater application, the soil retains 64% of its original sorption capacity.

Sawhney and Hill [17] found equivalent behavior for Connecticut soils. They suggested that alternating periods of wetting and drying may bring fresh mineral surfaces into equilibrium with the soil solution, creating new sites for phosphorus sorption. Similar conclusions were reached by Kao and Blanchar [18]. Enfield [16] noted that there is a balance between sorption and conversion of adsorbed P to more insoluble forms. He suggested that reactions occur which utilize adsorbed orthophosphate to form phosphate minerals that have solubilities that are somewhat less than the adsorbed forms. This regenerates some sites for adsorption.

Despite the fact that the soil at the Hollister facility retains its ability to adsorb phosphorus, a longer travel distance would be required for effective phosphorus removal.

SUMMARY

Nitrogen and phosphorus removal efficiencies were investigated after 30 years of rapid infiltration. Results suggested that 7 m of vertical travel distance provided effective nitrogen removal. Input total-N levels of 40.2 mg/L were reduced to less than 4 mg/L within the shallow aquifer underlying

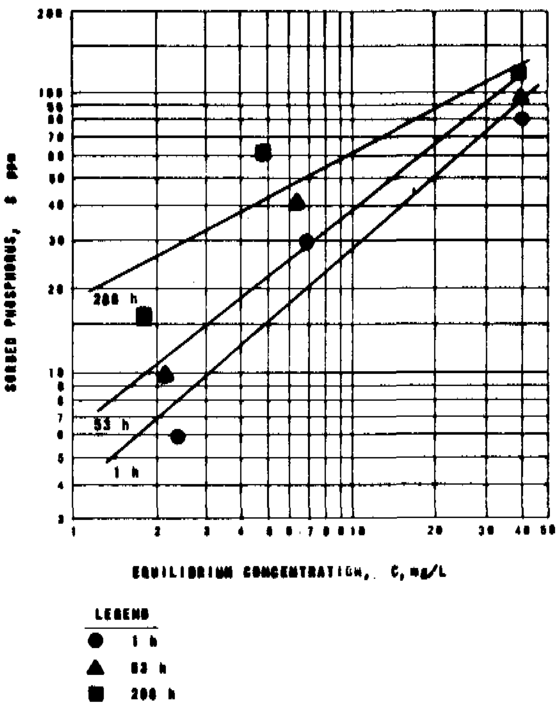

Figure 4. Logarithmic Plot of Freundlich Regression Equation to the Experimental Data For Sorbed Phosphorus as a Function of Time and Equilibrium Concentration. Control Site Composite, 0-16 cm depth.

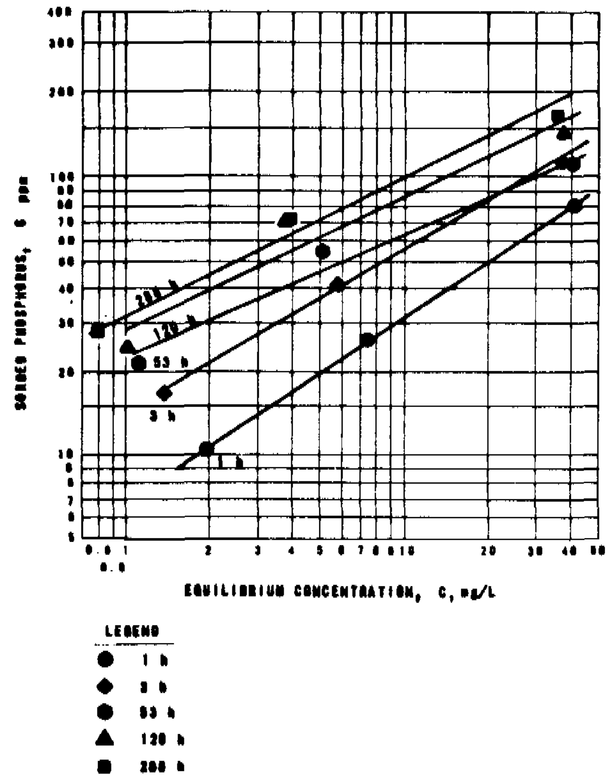

Figure 5. Logarithmic Plot of Freundlich Regression Equation to the Experimental Data For Sorbed Phosphorus as a Function of Time and Equilibrium Concentration. Treatment Site Site Composite, 0-16 cm Depth.

the site. Nitrate-N did not pose a pollution hazard, in fact, nitrate-N values beneath the infiltration site were lower than those recorded in offsite control wells. A comparison of total nitrogen input to that gained by the soil revealed that only 2% of the wastewater nitrogen could be accounted for in the upper 300 cm of soil profile. Denitrification was believed to be the primary nitrogen removal mechanism.

Phosphorus removal was found to be less effective, despite the soil's continued ability to adsorb solution phosphorus after 30 years of wastewater application. Input concentrations of 12.5 mg/L were reduced to 8.8 mg/L after 7 m of vertical travel at an annual wastewater application rate of 15.6 m. Longer travel distances would be required to increase removal efficiency.

ACKNOWLEDGMENTS

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CASE STUDIES

THE LAKE GEORGE VILLAGE (NY) LAND APPLICATION SYSTEM

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The Lake George Village Sewage Treatment Plant has been applying unchlorinated secondary treated effluent onto natural delta sand beds by rapid infiltration since 1939. The sand system has been shown to remove nonconservative materials such as BOD, COD, alkybenzenesulfonates, coliforms and fecal coli and ammonia and organic nitrogen. Soluble inorganic materials such as sodium, chloride, potassium and nitrate generally pass through the sand system. Calcium and magenesium showed no significant changes. Alkalinity increased due to carbonate and bicarbonate reactions within the soil. Orthophosphates were completely removed in the top 3 m of the sand beds and nitrates were removed in beds which were 18 m in depth or more.

Sewage flows vary from approximately 1,900 m^3/day in winter to 4,700 m^3/day during the summer tourist season. Dosing of the sand beds is intermittent with average overall loading rates of 0.12 m/day and actual infiltration rates of 0.08 to 0.3 m/day depending on the depth of water on the bed and the time since the last cleaning. The operation has continued successfully throughout the cold winter months experienced in this area.

The greatest removal of constituents occurred in the top 10 m of the sand beds. Continued quality improvement was observed in the further vertical flow and the approximately 600 m of horizontal flow before the applied effluent reemerges as seepage along the south bank of West Brook, a tributary to Lake George. There were no indications that the soil's capacity to treat the applied sewage effluent was approaching exhaustion.

INTRODUCTION

In 1936 there was concern for the potential pollution of beautiful Lake George by the increase in population at the southern end of the lake around the Village of Lake George. In order to prevent any contamination of the lake, a sewage treatment plant was designed. The inclusion of a land application system in this design was to comply with the regulation that there shall be no discharge of sewage or sewage effluent into any waters of Lake George or into any streams flowing into the Lake (1).

DESCRIPTION OF THE TREATMENT SYSTEM

The treatment plant as it exists today, is shown in Figure 1. The original plant put into operation in 1939 was built in triplicate to accommodate the summer tourist flows which were approximately 3 times the winter permanent population flows. The sewage from the Village is collected in a central sump where it is pumped by force main a distance of approximately 1.6 km to the treatment plant. After metering, the sewage flows through one of the three two-stage settling tanks with separate sludge digestion compartments, through one of three dosing chambers serving the siphons which are used to provide sufficient head to spread the water onto one of three trickling filters. Two of the trickling filters are of the high-rate rotary distributor type and are used in the summer. The third is a low-rate fixed nozzle filter which is covered by boards on sawhorses and used exclusively in the winter. After secondary sedimentation the unchlorinated final effluent is applied to the rapid infiltration sand beds. The original fixed beds (north beds 1-6) are in continual use to this day. The sand is a naturally occurring delta sand deposit which was deposited by the melting glacier approximately 10,000 years ago.

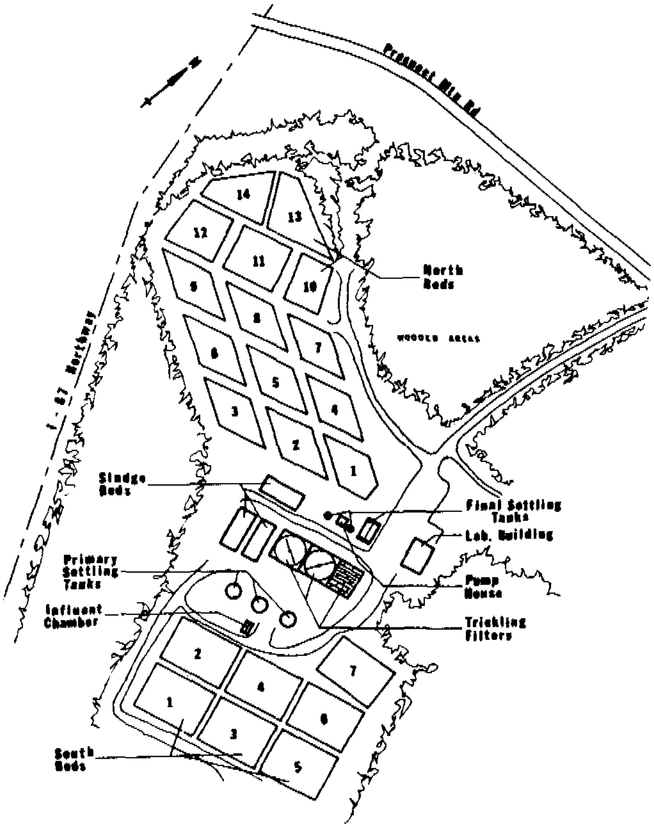

Figure 1. Plan of the Lake George Sewage Treatment Plant

The treatment plant has had 5 additions to the initial 6 sand beds since it was built. The major addition of south beds 1-6 and beds N-13 and -14 was made in 1965 when a new sewer system was built to serve the adjacent Town of Lake George. The present total infiltration area of the sand beds is 2.15 ha. The combined digested sludges are dried on adjacent sand beds and the dried sludge is removed to a landfill west of the treatment plant. The sand beds have operated satisfactorily through the rather severe winters of the area. Winter temperatures may reach as low as -35°C and periods of at least 33 days of continuous below freezing temperatures have been recorded. No chlorination is applied anywhere within the sewage treatment plant.

The lower (north) sand beds are all dosed by gravity, whereas the upper (south) beds are serviced by means of a pump. Normal operation of the infiltration beds is to dose one north and one south bed from 8 a.m. to 4 p.m. and another similar pair of beds from 4 p.m. to 8 a.m. On week-ends or holidays, two north and two south beds are dosed simultaneously for a 24-hour period. The beds normally drain in 1 to 3 days depending upon the amount of sewage applied and the time since the last cleaning. The design is to allow the beds to dry for at least one day between dosing. Occasionally, during extreme high summer flows, this rest period may become shortened or nonexistent. Periodically, at least twice a year, the beds are drained and any clogging material is removed along with the top few cm of sand which is disposed of in the areas adjacent to the sand beds. The bed is then plowed and releveled prior to being returned to service. Weed growths within the sand beds are considered undesirable and are normally removed as soon as possible.

BACKGROUND

In his original description of the treatment plant, Vrooman (2) stated that "the final effluent becomes ground water which in all probability seeps eventually, to some water course as a highly purified liquid which cannot be identified as a sewage effluent". However, this was purely speculation based on reasonably understood principles and it remained until the present authors made additional studies in the area to prove the validity of this statement. Studies made by Fink (3) measuring ground water resistivity indicated a band of low resistivity ground water following in a generally northerly direction along Gage Road toward West Brook (See Figure 2). With this information available, a survey was made of the south shore of West Brook in the vicinity of Gage Road. Considerable seepage was observed coming out of the ground along the south bank of West Brook at the edge of the flood plain. Conductivity measurements taken of this seepage showed the highest concentration of total dissolved solids to be in the closest proximity to Gage Road with the exception of some runoff which was later traced to an area where the highway department had stored highway de-icing salt uncovered on the surface of the ground for several years (4).

EXPERIMENTAL PROCEDURES

In order to study the quality of the ground water between the infiltration beds and the seepage along the south bank of West Brook, a series of observation wells was placed in this area as shown in Figure 2. The well points were located at different depths within the aquifer. In addition to the sampling wells, sampling stations were set up at the location of the two seepages prior to their flow into West Brook, and in West Brook upstream and downstream from the seepage areas. Seepage above (west of) Gage Road collects naturally in a small stream which discharges into West Brook immediately upstream from the culvert under Gage Road. The seepage in the area downstream (below Gage Road) was more diffuse, so a ditch was dug to consolidate these seepages into one stream. The seepages were monitored for flow and samples secured before discharge into West Brook.

Specific data for the depths of the individual wells, the approximate ground water levels within the wells and the bed rock or rejection point for the wells is shown in Table 1. In all cases, for each well site the depth of the points was lower with the notation proceeding through the alphabet from A (AA) to D. The steel wells consisting of a steel point and screen and iron pipe were manually driven. The wells marked P were augered with a 10 cm auger and then a plastic screen and pipe were placed in the hole. Use of the plastic pipe lessened contamination of the sample due to metals, primarily iron which may interfere with phosphorus measurements. As

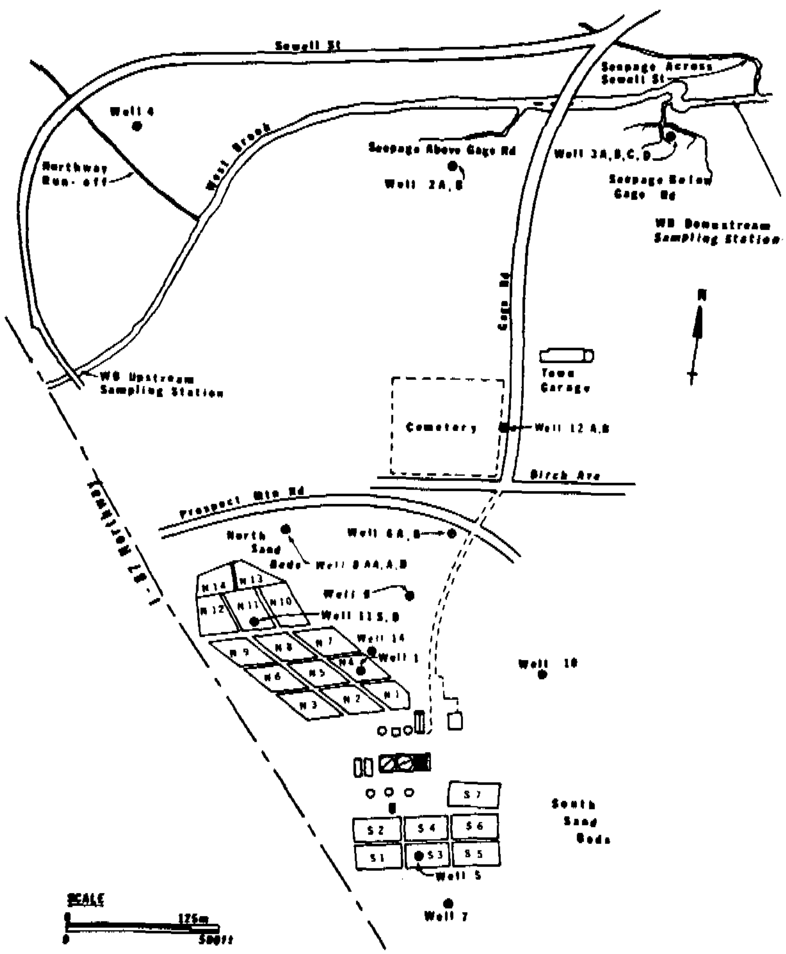

Figure 2. Map of the General Area of Study Showing the Observation Wells and Other Sampling Points.

TABLE I. WELL DATA
(Elevations in m. Above Mean Sea Level)

Location	Steel or Plastic	Top of Well	Ground Surface	Approx. Ground Water	Bottom of Point	Bedrock
1	P	145.57	144.77	126.47	124.44	123.48
2A	P	115.25	114.45	109.28	107.36	93.29
2B	S	115.44	114.45	109.31	100.72	93.73
3A	P	104.05	103.63	103.47	102.57	95.73
3B	P	104.19	103.63	103.52	100.32	95.73
3C	P	104.35	103.63	103.54	97.94	95.73
3D	S	103.73	103.63	103.44	96.23	95.73
4	S	115.68	114.52	114.50	112.65	112.65
5	P	152.50	151.03	147.22	146.16	145.55
6A	P	140.03	139.73	120.92	118.68	109.76
6B	S	140.25	139.85	120.59	109.76	109.76
7	S	153.85	152.53	151.55	150.47	150.47
8AA	P	143.37	143.30	123.48	121.96	118.38
8A	P	143.66	143.21	122.03	121.21	118.38
8B	P	143.68	143.29	122.02	118.38	118.38
9	S	143.11	142.28	123.10	120.58	120.58
10	S	141.80	141.08	134.66	133.81	133.81
11S	S	144.98	143.95	124.06	121.09	115.91
11D	S	145.17	143.95	123.81	117.40	115.91
12A	P	136.04	135.69	117.54	116.55	109.05
12B	P	136.06	135.71	117.04	109.07	109.07
14	S	144.27	143.80	125.81	124.61	124.00

the wells were put into service, samples were removed from them by pumping (in the shallower wells) or by means of a bailer. The first samples were secured in 1973; the major portion of the study was completed by the end of 1976. Some sampling and analysis are still being performed. In general, samples were secured on a bi-weekly basis.

In addition to the observation wells to determine changes in water quality with horizontal distance, studies were undertaken to determine the change in quality with depth in bed N-11. A series of driven well points, suction lysimeters, and pumping wells were installed in this sand bed as shown in Figure 3. Here again, the normal sampling procedure was bi-weekly, although during the flow tracer study to determine the velocity of flow through the sand bed, the pumped wells were monitored continuously.

In general samples were secured in rinsed plastic containers, returned to the laboratories, and analyzed as soon as possible. Preservation with mecuric chloride was performed in the field for all portions of the samples to be analyzed for nitrogen and phosphorus. All analyses were conducted according to Standard Methods (5).

INFILTRATION RATES

Infiltration rates were estimated based on the amount of sewage applied to each sand bed, the time it took for the sewage to drain through a bed, and the frequency of dosing. Since precise flow data to each sand bed are not available, it was assumed that half of the flow reaches the treatment plant from 8 a.m. to 4 p.m. and the other half of the daily flow occurs during the 16 hour night time period. Since two beds are dosed simultaneously during each period, it was assumed that each bed dosed received approximately one quarter of the daily flow. This value may not be entirely accurate since the lower north beds are dosed continuously by gravity and the upper south beds are dosed intermittently by means of a pump which is actuated by the depth of the water in a wet well filled by the flow over an adjustable weir in the pipe to the north beds. Time and facilities were not available for the actual measurements of the flow to the upper beds or the time of operation of the pump to the upper beds. It is felt that dividing the flow equally between the north and south beds provides a reasonable estimate for

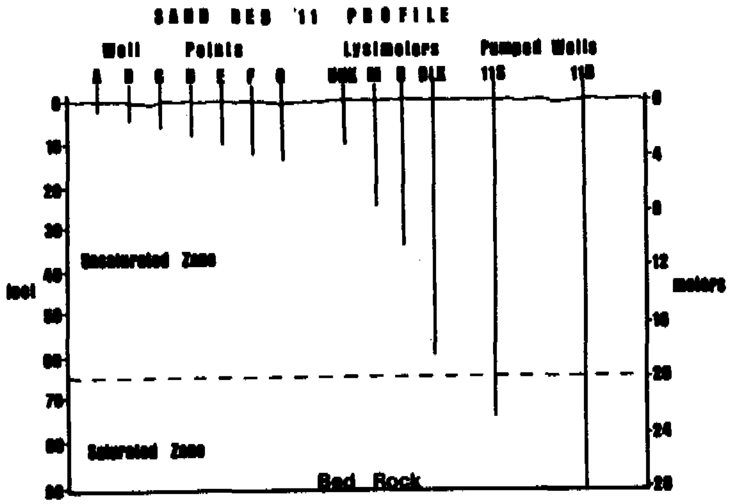

Figure 3. Profile of Bed N-11 Showing the Depths of Driven Well Points, The Operational Lysimeters and the Shallow and Deep Pumped Wells, The Latter Two of which Penetrate into the Saturated Aquifer.

calculation of the loading rates.

The estimated monthly loading rates for a 12 month period are shown in Table 2 (6). The maximum loading rate occurred during the month of August with a loading of 1.37 m^3/ha-min or 0.2 m/day. This represents the amount of liquid that can be safely applied to the sand beds without exceeding the total infiltration capacity. It must be mentioned, however, that during the latter part of August 1975 and 1976, the sand beds were all completely loaded and the normal drying time between dosing was either very short or non-existent. After the tourists departed after Labor Day, the flows diminished markedly and the sand beds were allowed to dry and were scraped, thereby increasing the infiltration capacity.

The actual infiltration rate was measured in several of the sand beds by installing a water level recorder in those beds. The rate of infiltration increased with the head of liquid on the sand beds as shown in Figure 4 (6). The lowest rates recorded with less than 0.3 m of liquid on the sand bed were in the range of 0.8-0.18 m/day under normal operating conditions. It may be seen that different beds have different infiltration rates, with bed S7 having a rate exceeding 0.3 m/day with a water depth on the bed of 0.6 m. An infiltration rate exceeding 0.6 m/day was measured on a freshly scraped bed with a depth of water of 0.3 m. With continued intermittent operation the flow rate decreased gradually to the values previously stated.

PURIFICATION WITH DEPTH IN THE UNSATURATED ZONE

All of the studies relating to the purification of the sewage effluent applied to the sand bed as a function of depth within the unsaturated zone in the sand bed were conducted in North Bed 11. The driven well points in this bed (Figure 3) were essentially ineffective from the standpoint of securing samples. The samples secured from the 4 operating lysimeters at depths of 3, 7, 11, and 18 m provided most of the information for the changes with depth in the unsaturated zone. Samples were also secured from the 2 pumped wells within the aquifer at 23 and 28 m depths, to evaluate the quality of the water in the saturated zone immediately beneath the sand beds.

Temperature measurements showed that the applied sewage effluent temperature varied with the ambient air temperature, whereas the temperature near the bottom of the unsaturated zone showed a lesser degree of fluctuation. Within the saturated zone the temperature range throughout the entire year was between 8 and 13°C.

There was little significant difference in pH at various depths. The lowest value observed was 6.5 at the 11 m depth in fall and the highest was 7.4 in well 11S during the summer.

TABLE 2
WASTEWATER LOADING RATES, LAKE GEORGE,NY

Month	Flow	Loading Rate	
	10^6 l/d	l/m^2-d	m/d
1974			
Sep	2.80	1.11	0.13
Oct	2.23	0.89	0.10
Nov	1.57	0.62	0.07
Dec	1.86	0.74	0.08
1975			
Jan	1.89	0.75	0.09
Feb	1.94	0.77	0.09
Mar	2.15	0.85	0.10
Apr	2.66	1.05	0.12
May	2.50	1.00	0.12
Jun	2.95	1.17	0.14
Jul	3.67	1.46	0.17
Aug	4.24	1.70	0.20
Average	2.54	1.01	0.12

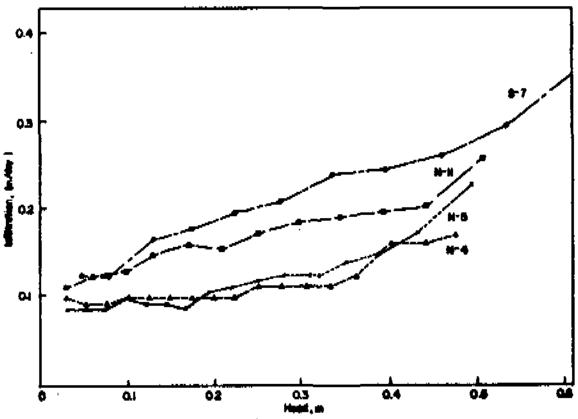

Figure 4. Comparison of the Infiltration Rates of Several Sand Beds in Normal Use

There was a consistent slight increase in dissolved solids concentration with depth during all seasons. The highest value of 300 mg/l occurred during the summer and the lowest (175 mg/l) during the winter. In the saturated zone immediately beneath the bed values never exceeded 125 mg/l, indicating the influence of the ground water in this area.

The principle of intermittent dosing of the sand beds is to allow the beds to become aerated between dosing, thus promoting aerobic treatment of the applied waste effluent. Measurements of the DO with depth showed the lowest values to be approximately 2.5 mg/l at the 18 m depth and in well 11S. These lowest values occurred during the fall and winter. The redox potential was determined only during the spring of 1976 and was positive at all times with the one exception being the 18 m depth at which point it was only slightly negative. The redox potential variation coincides fairly closely with the DO level variations measured in Bed N-11.

The highest chloride concentration of 100 mg/l occurred during the winter which is also the time of the greatest use of highway deicing salt. There was no significant change in the chloride content with depth in the sand beds.

The calcium content of the applied sewage effluent was approximately 20 mg/l with no significant changes in concentrations with depth within the unsaturated zone of Bed N-11. There was even less variation in magnesium content with depth, maintaining an average value of approximately 6 mg/l.

There was a consistent trend of an increase in alkalinity from about 100 mg/l in the applied sewage effluent to approximately 250 mg/l at the 18 m depth. There seems to be some relationship between alkalinity and pH and DO. As the alkalinity increased, the pH and DO decreased indicating the possible presence of microbial activity which utilizes oxygen, producing CO_2 which ultimately converts carbonates to bicarbonates which are measured as alkalinity.

There appears to be an inter-relationship between the organic, ammonia and nitrate nitrogen in the soil (8). During all seasons, there was a decrease in ammonia and organic nitrogen within the top 3 m of the sand bed with a consequent increase in the nitrate content at this depth. However, for summer and fall there appeared to be a subsequent decrease in nitrate with a significant increase in the organic and ammonia nitrogen at the 7 and 11 m depths. By the 18 m depth both the organic and ammonia nitrogen contents were less than 8 mg/l during the summer and fall compared to 16-18 mg/l of nitrogen applied to the sand bed. During spring there was no recurrence of the high ammonia and organic content at greater depths. Instead, values of organic, ammonia and nitrate nitrogen were all less than 0.5 mg/l. The changes in nitrogen compounds with depth during the spring in Bed N-11 are shown in Figure 5, with the depth of bed S-3 indicated on this figure for comparison. The upper dotted line

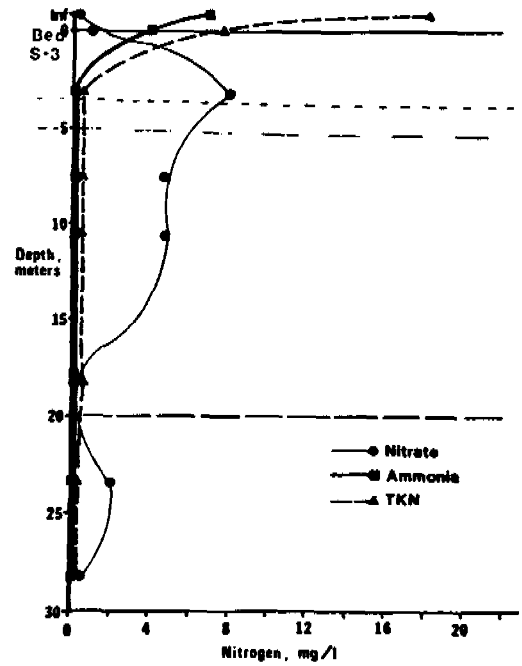

Figure 5. Variation of the Various Forms of Nitrogen Measured with Depth in Bed N-11 During Spring 1976. The Depth of Bed S-3 is Shown for Purposes of Comparison.

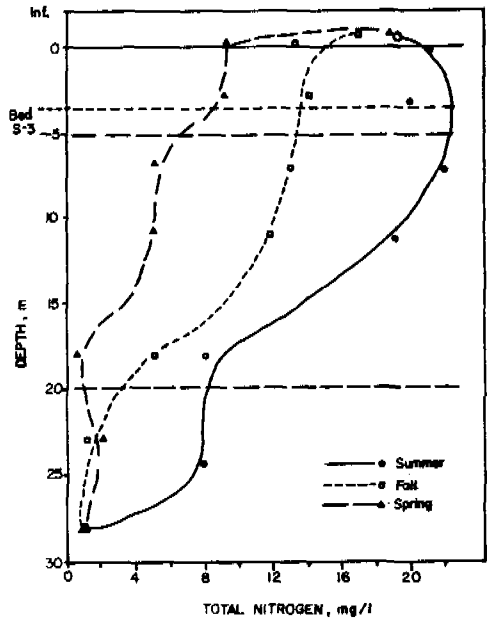

Figure 6. Variations in Total Nitrogen During Summer, Fall and Spring with Depth in Bed N-11, with the Depth of Bed S-3 Shown for Comparison.

indicates depth to water in bed S-3 and the upper dashed line the bedrock in bed S-3. The lower dashed line indicates the depth to water in Bed N-11; below this level of 20 to 22 m the sand is saturated with water. The high nitrate content at the 3 m depth also has significance in terms of other less deep infiltration beds. Some of the beds have only about 3 m of unsaturated zone followed by another 2 m of saturated zone above the bedrock. In these sand beds the oxidation of organic and ammonia nitrogen to the nitrate would result in nitrate entering the saturated zone and being carried through the soil. This could account for the elevated nitrate concentration measured in the ground water as described in the next section.

The loss of nitrogen from the aqueous system probably to elemental nitrogen is shown in Figure 6 in which the various forms of nitrogen are added for each season. The greatest loss of nitrogen occurred during spring. The loss of nitrogen corresponds with a low DO and a slightly negative redox potential at the 18 m depth in spring, and is assumed to represent reduction of nitrate.

In general the orthophosphate content was reduced from approximately 1.4 mg/l in the applied wastewater to 0.1 mg/l by the time the sewage effluent reached the 7 m depth. The total phosphorus indicated some initial reduction after 3 m vertical movement followed by higher values at greater depths. The highest values observed in the shallow pumping well were approximately 0.4 mgP/l of total phosphorus (8).

There were significant fluctuations during various seasons in the iron content at various depths. In general there was an increase in iron content with depth, with values varying between 0.5 and 8 mg/l. No direct relationship could be made between the iron content, pH, DO, redox potential or phosphorus content.

There were no significant changes in the sodium or potassium content of the applied sewage effluent as related to depth. The average sodium content was approximately 14 mg/l and the average potassium content was approximately 6 mg/l.

Other parameters measured included BOD, COD, alkylbenzenesulfonates, coliform and fecal coli all of which were essentially removed in the top 3 m of the unsaturated zone of the sand bed. Preliminary measurements were made for copper; however, all of the concentrations were less than 0.5 mg/l which is the lowest detectable limit using the atomic absorption spectrometer system used. Aluminum was not found within the detectable limit of about 1 mg/l.

PURIFICATION WITH DISTANCE IN THE SATURATED ZONE

After the applied sewage effluent flows vertically through the unsaturated zone of the sand beds it flows approximately another 600 m northward through the saturated zone before it emerges as seepage along the south banks of West Brook. This distance provides some additional purification of the liquid. In order to monitor the quality of the water in this saturated zone, a series of observation wells was installed. The location of the observation wells is shown in Figure 2, and specific data for the depths of the wells is given in Table 1.

There was little change in pH in the saturated zone during the spring and winter. During the summer the pH reached a high value of 8.8 in well 6A. Thereafter, there was a gradual decrease in the pH in all the wells, but it remained above pH 7.0 which was the average value of the sewage treatment plant effluent. During the fall the pattern was the same as during the summer but with less high pH values.

During all seasons the shallower wells indicated higher values of dissolved solids than the corresponding deeper wells. This indicates that the sewage containing higher dissolved solids than the ground water remained nearer the surface of the aquifer with the deeper sampling points more representative of the normal ground water. High dissolved solids were observed at wells 3 as influenced by the storage of the highway deicing salt at the town garage.

The shallower wells consistently had higher DO values than the deeper wells, with the lowest values occurring during the summer at 0.5 mg/l in wells 9 and 3D. There was a slight trend toward increasing DO with increased distance from the sand infiltration bed.

Redox potential measurements were only made during the spring of 1976 with values ranging between +100 and +150 mV. The lowest values were observed in the control wells which received no effluent from the sewage treatment plant.

There was a slight trend of decreasing chloride concentration with distance from the sand infiltration bed. Again, wells 3 showed the influence of the nearby highway deicing salt storage area.

There was little change in the organic nitrogen content of 2 mgN/l through the saturated portion of the soil system. However, the ammonia-nitrogen content was reduced from an average value of 4 mg/l to less than 0.1 mgN/l. During the summer there was a marked reduction in the nitrate content with values in well 2B of approximately 0.6 mgN/l. During the fall, winter and spring the nitrate values at well 2A ranged between 5 and 7 mgN/l. There appears to have been some microbially mediated conversion of the nitrate to gaseous forms of nitrogen during the summer.

In general the total phosphorus content was reduced to values less than 200 ug/l prior to emergence in the seepage at West Brook. The orthophosphate was in general less than 10 ugP/l with the lowest values of less than 0.2 ugP/l (the minimal detectable limit of the analytical method used) in well 2B.

DISCUSSION

The combined vertical and horizontal transport of the unchlorinated effluent from the Lake George Village Sewage Treatment Plant through the sand achieves the production of a highly purified effluent. There are no significant adverse effects upon ground water as indicated by the parameters of temperature, pH, alkalinity, coliforms, BOD, COD or soluble phosphorus. Whereas there are some increases in the total dissolved sölids, the alkalinity and the chloride content of the ground water, these are within acceptable limits.

The only parameter of possible concern is the nitrate content which is in the range of 7 mgN/l in the seepage. This is close to the recommended drinking water standards of 10 mgN/l (7). It appears that the sand system is capable of further lowering this nitrate level primarily during the warmer seasons and at greater depths within the unsaturated zone. This is attributed to conversion of the nitrate ion to gaseous nitrogen which escapes from the aqueous system. Studies are presently being conducted to develop a method of creating conditions which would convert more nitrates to gaseous nitrogen without interfering with the phosphorus removal.

The Lake George Village Sewage Treatment Plant land application system has been successfully achieving the equivalent of tertiary treatment of domestic sewage since 1939. There are no indications that the system will not continue achieving this high degree of treatment for a long period of time. Thus, a land application system using sand can be considered to be a satisfactory method for providing tertiary treatment of wastewater.

ACKNOWLEDGEMENTS

The cooperation of Harold Gordon, Operator of the Lake George Village Tratement Plant, and his staff is greatly appreciated. This study was initiated with a small grant from the U.S. Army Corps of Engineers Cold Regions Research and Engineering Laboratory under Purchase Order BACA89-75-1265. The project was completed with support by EPA Grant R-803452-01-0 and a significant contribution by Rensselaer Polytechnic Institute.

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CASE STUDIES

YEAR ROUND APPLICATION OF WASTEWATERS
TO LAND IN A HUMID CONTINENTAL CLIMATE
WITH UTILIZATION OF NATURAL VEGETATION

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The project was carried out on a 2 section 6 hectare overland flow treatment system in Morris County, New Jersey. The two main objectives of the study were to ascertain whether tertiary treated wastewaters could be effectively applied to an over land flow treatment system on a year round basis despite sub freezing temperatures and whether the use of natural vegetation would be an acceptable substitute to a harvestable crop system. The results of the two year study (1976 through 1977) indicate that year round application with certain precautions is possible in climates with a 2 to 3 month sustained winter season. Ice buildup was in some respects beneficial to the system. Concurrently the manipulation of the natural vegetation as a crop cover rather than a harvestable crop proved to be not only more cost effective but of direct benefit to wintertime operations.

INTRODUCTION

The study site is located in the northwestern portion of Morris County, New Jersey approximately 64 kilometers west of New York City. Geographically this portion of Morris County is identified as being within the Highland Province of the State. An area characteristically identified by its broad rounded flat-topped ridges, rising 121 to 183 meters above deep and generally narrow valleys. The climate falls within the general classification of "humid continental" and is characterized by a great annual range of temperature. Mean monthly temperatures for June, July and August are near or above 21.1°C while in January the averages are typically below 0°C. Microclimatological data of the site is presented further on. The treatment facility which provided the effluent to be discharged was 250,000 GPD capacity contact aeration plant with rapid sand filters, phosphate removal and chlorination as tertiary treatment. The organic design load was 270 mg/l $B.O.D._5$ and the plant did meet the following average quality effluent paramters throughout the study.

Suspended Solids 6 mg/l or less

$B.O.D._5$ 6 mg/l or less

The plant during the study processed a maximum of 80,000 GPD which was well below design. The high degree of treatment and requirement that it be applied to the land rather than discharged to a stream was a result of a request of the New Jersey Department of Environmental Protection that extra precautions be taken to protect a small tributary of the headwaters of the South Branch of the Raritan River. The original design called for the effluent to be applied to the land for 9 months of the year and recharged directly into the soil through four 30.48m x 30.48m x 1.82m groundwater recharge beds for the remaining 3 months during cold weather. Subsequent to construction of the four recharge beds however it was determined that they were non-functional due to an unanticipated artesian groundwater table.

This abandonment left the owner of the sewage treatment plant in a precarious position. The D.E.P. was especially concerned because no downhill collection and recirculation system had been provided for this overland flow system. However after serious deliberation a determination was made to allow year round land application on a trial basis.

DISCUSSION

When the treatment plant was completed in the Spring of 1975 a single area of 2.83 hectares had been set aside for land application of the treated wastewaters. The vegetative cover at that time was of mesic upland forest with an Quercus sp.-Acer sp. canopy and a Lindera benzoin-Viburnum sp.understory. The soil on the site was identified according to the National Cooperative Soil Survey as Cokesbury extremely stony loam with a fairly uniform 4% slope. The site was cleared of all natural vegetation and rough graded but because of the extreme rocky nature of the soil imported fill (clay loam) was brought in and graded 30 to 45 centimeters deep over the entire site. An earthern berm was also constructed along the southern perimeter of the field since this was adjacent to a small stream. The effluent piping in the field consisted of buried 15.24cm and 10.15cm diameter mains with 5.08cm and 3.81cm diameter PVC laterals with Model 41 Rainbird sprinkler heads set 20 to 25 centimeters above the ground surface and provided with adequate backdrains to prevent freezing. Five 7.6cm diameter PVC monitoring wells were established around the field to monitor groundwater quality. Depths average 7.6 meters and static water level was 1.8 meters below ground surface. One well of the 5 (see exhibit 2) was established upstream as a background reference and the remaining 4 were established at various locations at the downstream end of the field. Samples were collected from these wells every 2 weeks and analyzed for B.O,D.5 orthophosphate and NO_3 Nitrogen. The intent of the owner was to establish a crop cover of Phalaris arundinacea (reed canary grass) however due to other associated problems, namely improper soil fertilization and erosion a nurse crop of Lolium perenne (perennial rye grass) had to be established first. The reed canary grass subsequently seeded through late Spring and early Summer.

In later summer of 1975, the field was receiving approximately 4.2cm/week (45,000 GPD) and runoff was becoming a severe problem due to the saturated condition of the soil. The rye grass was cloning thus reducing the effective basal coverage on the land surface and at the downhill end of the site flooding and ponding were increasing in frequency. Here the rye grass cover had been completely destroyed. The saturated conditions of the soil however were conducive to the natural propagation of indigenous freshwater marsh species such as Galamagrostis inexpansa, Typha latifolia, Scirpus validus, Sagittaris brevirostra, Eleocharis flaccida, Sparganiu minimum, and Impatiens biflora. By early fall reed canary grass seedlings 10 to 15cm high were also appearing sporadically around the field. In September of 1975, because of the failure of the 4 recharge beds and severe runoff problem occurring on the spray field the owner was advised to seek additional areas to spray his effluent. He selected a more elevated and exposed area of some 3.2 hectares approximately 30 meters due south in what had once been an active farmfield but which had been dormant for several years and was now in a vegetational transition known as "successional field." The dominant vegetation consisted of annual and perennial herbs such as Solidago canadensis, Potentilla pumila, Fragraria americana, Chicorium intybus, Verbascum thapsus, Reseda lutea, Rubus alleghaniensis and woody herbaceous shrubs such as Cornus amomum and Cornus stolonifera and tree seedlings parented by the surrounding dominant canopy forest usually Fraxinus americana, Betula populifolia, Juniperis virginiana various species of Quercus and Acer Rubrum. The D.E.P. recommended that the existing vegetation be preserved as much as possible to avoid the runoff problems which had occurred on field #1. An uninformed contractor however subsequently clearcut the entire area with a brush hog prior to installation of the piping system. The soils of the site were determined to be Annandale gravelly loam with an average slope of 5% and a maximum of 24% in one small area at the downhill end of the site. Construction and layout were similar to field #1 however only 44 spray heads were installed. An earthern berm similar to the one installed at field #1 was also installed along the entire downhill side of the field as a precautionary measure to protect the adjacent stream and the South Branch of the Raritan River. Again reed canary grass was broadcast generally over the clearcut existing vegetation and wherever construction had taken place. In November 1975 the field was put into operation at a minimal rate of 1.2cm/ week(15,000GPD). The winter of 1975 was particularly severe with the State experiencing some of the coldest weather in a decade during the months of December 1975 and January 1976. (See summary of temperatures and precipitation in EXHIBIT 1.) Ice buildup was substantial on both fields however it was particularly severe on field #2 due to the higher elevation and more exposed conditions. In most instances the ice on field #2 accumulated in mounds 1.2 to 1.8 meters above the surface of the ground immediately near the spray head and gradually tapered to several centimeters in thickness some 6 to 9 meters radially from the spray head. On field #1 the ice built up to a maximum of 0.6 to 0.9 meters of depth. All spray heads remained, with very few exceptions, free of ice buildup and were able to maintain an ice free zone around themselves some 0.9 to 1.2 meters in diameter. Very little runoff occurred during the subfreezing conditions of December and January despite the frozen ground. However during daytime hours when the incoming solar radiation was particularly intense, melting did occur but very little runoff was produced. Because of the sparse vegetative cover the ground surface was solidly frozen. It was assumed that very little treatment was occurring through the soil. Samples of the ice mounds were taken in late February and early March 1976 to determine the PO_4 and NO_3 concentrations since a rapid melt could cause a short term severe runoff problem. The results indicated 0 mg/l for orthophosphate and 2 mg/l of nitrate nitrogen. Fecal coliform analysis showed 20 MPN/100 ml. During March and April of 1976 the vegetative growth on both fields had increased significantly. Field #1 contained a **mixture** of successional mesic upland marsh plant species and reed canary grass seedlings with an average height of 45.7 to 60.9 centimeters. Field #2 saw a return of the original successional field vegetation with some reed canary grass seedlings occurring only in the wettest portions of the site particularly near the spray heads. Vegetative height average 30.4 to 35.5 centimeters. In late July 1976 the D.E.P. made two recommendations to the owner. One that he immediately divert 20,000GPD from field #1 to field #2 and secondly that the vegetation on both fields not be cut or harvested except for keeping the spray heads clear. The effects of the first recommendation were immediate. Field #1 within a month had dried out substantially, ponding had ceased and runoff had been sharply curtailed. It was hoped that recommendation two would allow the maturing reed canary grass time to reseed itself thus enabling the Department to assess its competiveness with indigenous species. For the owner this also meant a substantial savings in the capital cost associated with the cutting, collecting and disposing of such vegetation. In those areas where the reed canary grass grew around the spray head it became necessary to cut these areas bi-weekly but where the successional field vegetation predominated cutting was necessary only every 6-8 weeks. In August 1976 the DEP allowed the owner to raise the gallonage allotted to field #2 to 4.5cm/week (55,000GPD). By the fall of 1976 the vegetative cover on both tracts consisted of predominantly indigenous species as previously outlined ranging in height from 60.9 to 9.4 centimeters. The reed canary grass occupied 30% and 5% of the total vegetative cover on fields #1 and #2 respectively. During the early part of December 1976 heavy snowfalls caused the collapse of the tall vegetation and subsequent freezing conditions resulted in a frozen canopy of ice and snow over the matted vegetation. This frozen canopy was broken through at several locations on both fields in January 1977. The soil was found to be in an unfrozen state and despite ambient subfreezing temperatures assumed to be biologically active. No soil samples were taken. Ice buildup was similar in all respects to the previous winter, and again with a few exceptions the spray heads remained free of ice buildup. The only problem spray heads were those that had been installed lower than 7.6cm above the ground surface and those at the most downhill portions of the field. These were subject to ice aggradation as a result of intermittent melt and freeze. During March and April 1977 the spring thaw results in very little runoff from either field except when associated with a heavy downpour. From May 1977 through September 1977 the vegetation on both fields grew prolifically. The reed canary grass increased its surface area coverage on field #1 from 30% of the previous summer to 40%. Mature height ranged from 1.2 to 1.4 meters. On field #2 the woody and herbaceous shrubs and annual and perennial weeds which had previously been clearcut grew back with an intense resurgence. The reed canary grass however did not increase significantly preferring to remain in the moist soil conditions immediately adjacent to the spray heads. Both fields were examined in early November 1977 and it was observed that a mat of dead and decaying vegetation had begun to buildup on the soil of field #1 which although beneficial for wintertime operation could hinder the natural reseeding of the reed canary grass and retard new vegetative growth in the spring of 1978. D.E.P. recommended that the owner harvest the dead vegetation on field #1 in early April 1978 or just prior to the growing season and that said harvesting be done in the spring every two years hence. Field #2 on the other hand exhibited no symptoms which could adversely affect its ability to function as a land treatment system. It was determined that it could be as long as four years between cutting operations due to the characteristics of this successional field cover. This study was terminated in December 1977 however this land application system is still being followed closely.

CONCLUSIONS

Utilization of the naturally occurring vegetation in land disposal systems appears to be less costly in two major respects:

(1) less capital expenditure is required for establishment as compared to imported or exotic species of vegetation and;

(2) maintenance costs are lower because less frequent cutting and cultivation are required on both a short term and long term basis.

By using naturally occurring vegetation it would also appear that start up of such land application facilities may be quicker.

This study would also seem to indicate that subfreezing temperatures and ice buildup have little major effect on the hydraulics of these application systems except during thawing periods when accompanied by heavy rainfall, or when application rates were excessive for the soil conditions and vegetative cover of the site. The buildup of ice on the sprayfields also appeared to be of some benefit when overlain on a heavy vegetative cover. Even though ambient air temperatures were below freezing the soil remained unfrozen in such instances and appeared to be still biologically active. While this last point was not especially critical in this particular case study because of the advanced treatment of the effluent there may be other instances when this may become critical when the effluent is not so "polished".

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EXHIBIT I
TEMPERATURE AND PRECIPITATION
LONG VALLEY, N.J. STATION
NATIONAL OCEANIC AND ATMOSPHERIC ADMINISTRATION

Long. 74°-47'
Lat. 40°-47' Elev. 550 feet

DATE	°C MONTHLY AVG.MAX.TEMP.	(°C) MONTHLY AVG.MIN.TEMP.	(°C) MONTHLY AVG.TEMP.	(°C) DEPARTURE FROM NORMAL	SNOW & SLEET (CENTIMETERS)
Jan. 1976	.9	-10.4m	-4.7m	-2.1	20.3
Feb.	8.5	-5.0	1.8	+3.5	17.8
March	12.0	-1.4	5.3	+2.7	20.3
April	17.8	3.5	10.7	+1.7	Trace
May	20.6	5.8	13.2	-1.0	0
June	26.8	12.8	19.8	+.6	0
July	26.9	12.0	19.5	-2.3	0
August	27.0m	11.5m	19.3m	-1.4	0
September	22.4m	8.3m	15.4m	-1.5	0
October	14.4	3.6	9.0	-2.4	0
November	8.1	-3.5	2.3	-3.0	2.54
December	1.4	-9.2	-3.9	-2.8	27.9
Jan. 1977	-2.2	-13.7	-7.9	-5.3	29.2
February	4.3	-7.0	-1.3	+.4	17.8
March	11.9m	-.3	5.8m	+3.2	12.7
April	17.4	2.3	9.9	+9	0
May	23.8	7.3	15.5	+1.3	0
June	24.4	11.3	17.8	-1.4	0
July	28.0	15.0	21.5	-.3	0
August	24.9	14.2	19.5	-1.1	0
September	20.7	11.7	16.2	-.7	0
October	14.5	3.0	8.8	-2.6	0
November	9.0	1.5	5.3	-.05	7.6
December	1.4	-7.1	2.8	-1.7	10.1

m* Some Daily Temperatures not collected during the month
Hence not complete monthly average.

*All
Analyses in Mg/l

EXHIBIT 2
Field 1
MONITORING WELL DATA

WELL #1 (BACKGROUND	Monthly Avg. 1976	Monthly Avg. 1977
Nitrate N	1.3	.35
Orthophosphate	3.1	.20
B.O.D.$_5$	.8	1.3
WELL #2 (DOWN STREAM)		
Nitrate N.	.2	.25
Orthophosphate	.006	.50
B.O.D.$_5$	2.7	6.5
WELL #3 (DOWN STREAM)		
Nitrate N	.11	.06
Orthophosphate	.8	0
B.O.D.$_5$	1.3	1.5
WELL #4 (DOWN STREAM)		
Nitrate N	.2	.45
Orthophosphate	.04	.20
B.O.D.$_5$	1.4	7.1
WELL #5 (DOWN STREAM)		
Nitrate N	.25	.19
Orthophosphate	.8	0
B.O.D.$_5$	1.4	1.6

*All
Analyses in Mg/l

EXHIBIT 3
Field 2
MONITORING WELL DATA

WELL #8 (DOWN STREAM)	Monthly Avg. 1976	Monthly Avg. 1977
Nitrate N	.16	.36
Orthophosphate	0	0
$B.O.D._5$	1.5	6.4
WELL #9 (DOWN STREAM)		
Nitrate N	.72	5.3
Orthophosphate	1.2	0
$B.O.D._5$	1.8	1.8
WELL #10 (DOWN STREAM)		
Nitrate N	.32	.67
Orthophosphate	.40	0
$B.O.D._5$	.63	4.8
WELL #11 (DOWN STREAM)		
Nitrate N	.39	.36
Orthophosphate	0	0
$B.O.D._5$	1.2	1.7

CASE STUDIES

FIELD INVESTIGATIONS OF ADVANCED TREATMENT OF MUNICIPAL WASTEWATER BY OVERLAND FLOW

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ABSTRACT

Overland flow treatment of municipal facultative lagoon effluent was studied on 24 research plots measuring 4.6 x 46 m at Utica, Mississippi. Factors evaluated included the amount of wastewater applied, length of application period, slope of treatment area, crop management and reduction in biochemical oxygen demand (BOD), suspended solids (SS), nitrogen, phosphorus, heavy metals and coliforms. Runoff BOD and SS average 10 and 14 mg/ℓ, respectively, after overland flow treatment. Nitrogen reductions of 80 percent or more can be achieved at application rates of 2.54 cm in 18 hours for the warmer months of the year and of 1.27 cm in 18 hours for the cooler winter months. Overseeding the treatment area with a winter ryegrass cover helps to maintain nitrogen reduction. Phosphorus reduction ranges from 40 to 60 percent at an application rate of 1.27 cm in 6 hours. Addition of aluminum sulfate (alum) to the wastewater prior to land application enhances phosphorus reduction to over 80 percent. Fecal coliforms in undisinfected wastewater are not reduced to 200 MPN/100 ml with 5-day-per-week wastewater applications. Concentrations of fecal coliforms in runoff were higher than those in the applied wastewater during the summer months. Continuous application of wastewater for 7 days per week has produced coliform concentrations in runoff waters less than 200 MPN/100 ml. Overland flow effectively reduces heavy metals in applied wastewater by more than 90 percent. Overland flow treatment of municipal wastewaters can produce grass yields on marginal agricultural soils that approach the yields obtained on better agricultural soils.

INTRODUCTION

In 1971, the U. S. Army Corps of Engineers began an investigation of comprehensive wastewater management on a regional basis. This work was carried out within a cooperative agreement between the Department of the Army and the Environmental Protection Agency. In addition, the Civil Works Directorate of the Office of the Chief of Engineers requested that a technical assessment of land treatment methodologies be provided for use in the Corps of Engineers Wastewater Management Program. This assessment revealed a number of areas of insufficient information for proper implementation of land treatment systems. The U. S. Army Cold Regions Research and Engineering Laboratory (CRREL), Hanover, New Hampshire, was given the responsibility to develop a research program entitled "Soil as a Purification Medium for Wastewater." This program was designed to provide the necessary information for the proper design and operation of land treatment systems. The overland flow research project at the Waterways Experiment Station (WES) is part of this Wastewater Research Program.

In 1972, the WES initiated a greenhouse modeling effort to study overland flow for treatment of municipal wastewater. This initial greenhouse research indicated that overland flow was very effective in wastewater treatment for nitrogen and heavy metals and moderately effective for phosphorus reduction (1-3). A need existed for the evaluation of a prototype overland flow system in a natural environment. Such a research facility was established in 1975 near Utica, Mississippi, about 24 miles southeast of Vicksburg.

Climate at Utica averages 137 cm of precipitation per year with 5 cm as snow and sleet. Mean annual maximum and minimum temperatures are 24° and 12°C, respectively. Temperature extremes have been recorded at a maximum of 41°C and a minimum of -20°C.

DESCRIPTION OF THE STUDY

The research facility consists of 24 plots, each measuring 4.6 x 46 m constructed on slopes of 2, 4 and 8 percent, with eight plots on each slope. This experimental design enables duplication of four different modes of operation on each slope for a comprehensive evaluation of overland flow treatment of wastewater. Each plot was seeded with a 5:2:2:1 mixture of four grass species: reed canary (*Phalaris arundinacea* L.), Kentucky 31 tall fescue (*Festuca arundinacea*), perennial ryegrass (*Lolium dactylon* L.), respectively. The rate of seeding was three times that recommended by agricultural extension specialists to ensure a dense grass cover. The soil is a Grenada silty clay loam with a pH of 7.0, a bulk density of 1.2 gm/cm^3, a permeability of 0.042 cm/hr, and a moisture content at 1/3 and 15 bar of 23.0 and 9.9 percent, respectively.

Wastewater for irrigation is obtained from one of the two 2.4-ha facultative lagoons that serve most of the 1000 people of Utica. Lagoon effluent at Utica is lower in nitrogen, phosphorus and heavy metals than representative wastewaters across the Nation. Therefore, additional nutrients as NH_4Cl, $NH_4H_2PO_4$ and heavy metals as chlorides are injected into the lagoon effluent before irrigation to raise the total nitrogen to 20 mg/ℓ, total phosphorus to 10 mg/ℓ, zinc to 0.3 mg/ℓ, copper to 0.1 mg/ℓ, nickel to 0.1 mg/ℓ and cadmium to 0.05 mg/ℓ. Biochemical oxygen demand (BOD) and suspended solids (SS) average 22 and 35 mg/ℓ, respectively. The pH of the lagoon effluent ranges from 7 to 11, with an average at pH 9.0. Effluent application to each plot is automatically controlled to give predetermined amounts of wastewater over predetermined time periods. Wastewater is applied into a trough made from a rain gutter at the top of the slope. Holes in the bottom of the trough distribute wastewater evenly across the plot. A flow regulator is adjusted to deliver from 1.27 to 5.08 cm of wastewater per day. The period of application is set by the opening and closing of an electrically timed solenoid valve. Application periods of 6, 18, or 24 hours per day, 5 or 7 days per week, are being evaluated. Runoff water is collected in a sump and pumped off the plot through a meter to measure the volume of runoff. Wastewater and runoff are collected periodically to determine water quality.

Aspects of overland flow treatment of wastewater that are being evaluated include: rate of wastewater application, length of application period, slope of treatment area, crop management, reduction of BOD, SS, nitrogen, phosphorus, heavy metals and fecal coliforms.

Two crop management systems are being evaluated. One system consists of the four grasses previously mentioned and is harvested three times per year: spring, early summer and early fall. The second system is similar to the first with the exception that after the fall harvest, the plot is overseeded with a winter cover crop of ryegrass and harvested one additional time during early spring.

Information generated from the Utica research project will be used as a basis for the development of improved design criteria and operational guidance for overland flow wastewater treatment systems. This paper describes some of the highlights of the research to date.

RESULTS AND DISCUSSION

BOD and SS Treatment

The ability of an overland flow treatment system to reduce BOD and SS is of considerable importance since these parameters are limited by virtually every discharge permit issued in the United States. BOD concentrations in the applied effluent ranged from 6 to 37 mg/ℓ with an average concentration of 22 mg/ℓ. After overland flow, BOD concentrations in the runoff from each slope had been reduced to approximately

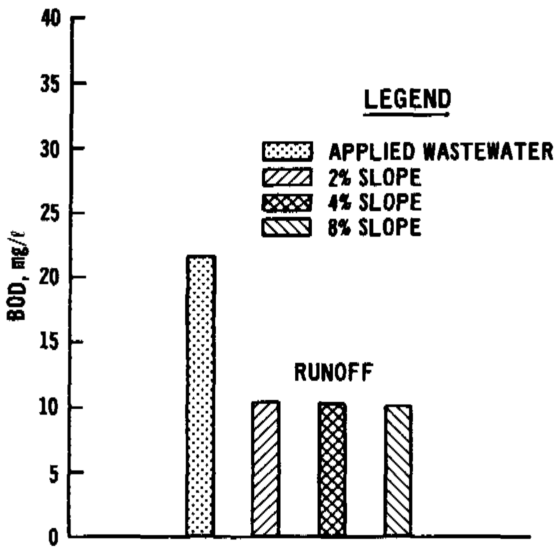

Figure 1. Reduction of wastewater BOD by overland flow at indicated slope for an application rate of 1.27 cm applied in 6 hours, 5 days per week.

10 mg/ℓ (Figure 1). Similar results were observed for SS, which ranged from 8 to 75 mg/ℓ, with an average throughout the year of 35 mg/ℓ in applied wastewater and approximately 15 mg/ℓ SS in runoff waters (Figure 2). There were no significant differences in BOD and SS treatment between slopes. Studies have shown the potential of satisfactory removal of BOD and SS at elevated hydraulic loadings (4).

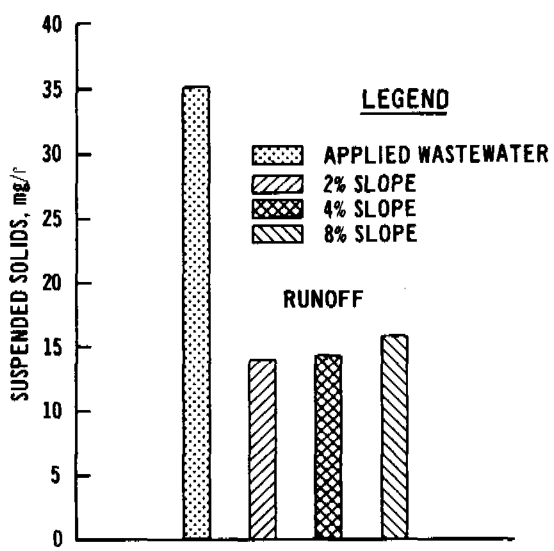

Figure 2. Reduction of wastewater SS by overland flow at indicated slope for an application rate of 1.27 cm applied in 6 hours, 5 days per week.

Nitrogen Treatment

Both greenhouse and field research indicate that nitrogen is effectively reduced from applied wastewater after treatment by overland flow. During the summer and spring of the year, nitrogen reduction approximates 90 percent when wastewater is applied at a rate of 1.27 cm in 6 hours for 5 days per week (Figure 3). As temperatures become cooler, nitrogen reduction decreases to approximately 80 percent in the fall and below 80 percent in the winter. While there are no differences in nitrogen reduction among slopes during most of the year, the 8-percent slope has a significantly higher nitrogen reduction than either of the other slopes during the winter. This difference could be related to the orientation of the 8-percent slopes toward the sun. Snow and ice appear to melt more quickly on the 8-percent slope during the winter months than on the other slopes.

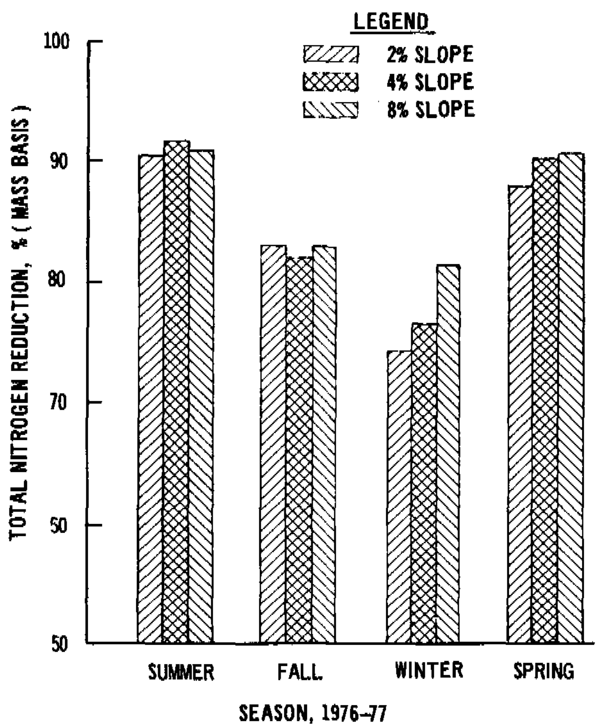

Figure 3. Influence of slope and season on total nitrogen reduction at an application rate of 1.27 cm in 6 hours.

Increasing the amount of applied wastewater to 2.54 cm in 6 hours during November 1976 resulted in comparable runoff Total Kjeldahl Nitrogen (TKN)

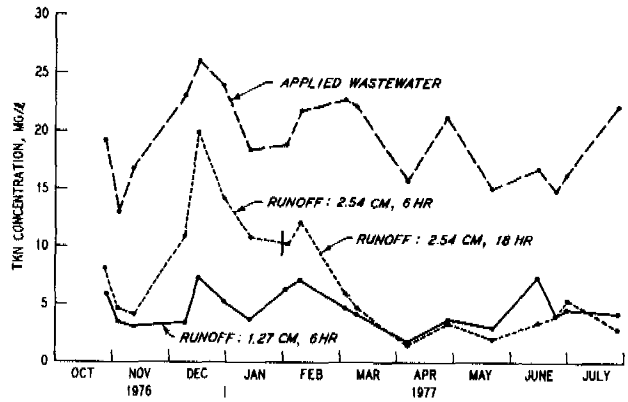

Figure 4. Influence of wastewater application rate on TKN concentrations in runoff waters from a 2-percent slope.

concentrations to that of a 1.27-cm-in-6-hour application rate (Figure 4). However, during December and January, significantly higher TKN was found in runoff waters at the higher application rate. This nitrogen was predominantly organic nitrogen. Nitrogen reduction at the 2.54-cm-in-6-hour application rate resulted in 70 percent or more reduction in applied nitrogen. On 31 January 1978, the application period for the 2.54-cm amount of wastewater was increased to 18 hours. During February, the runoff TKN concentrations for both rates began to approach each other and were similar through the spring months. Recent results from the winter of 1977 indicate that while runoff TKN concentrations remained approximately 5 mg/ℓ, runoff nitrate concentrations were observed at 10 mg/ℓ or more. Therefore, a total nitrogen concentration of 15 mg/ℓ or more in runoff waters was observed during the coldest winter months of 1977. Even the 1.27-cm-in-6-hour application rate resulted in nitrate concentrations of 5 mg/ℓ or more. Only when 1.27 cm of wastewater was applied in an 18-hour period during this winter did total nitrogen concentrations become approximately 5 mg/ℓ or less. These results indicate that nitrogen treatment by overland flow is significantly hindered at colder winter temperatures. The contributing factors for this are slower growth of grasses and decreased microorganism activity at cooler temperatures. In an effort to increase grass growth, the treatment area was overseeded with ryegrass after the fall harvest in 1976. Nitrogen reduction was maintained at approximately 80 percent while the dormant reed canary mixture dropped to approximately 50-percent reduction (Figure 5). This difference was not as pronounced in the winter of 1977. While the existence of an actively growing grass cover during the winter months is believed to be beneficial in maintaining nitrogen treatment, additional data are needed to substantiate this.

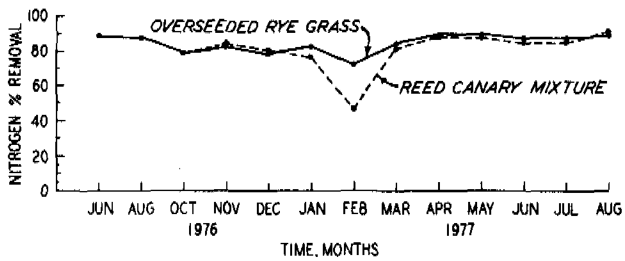

Figure 5. Influence of crop management practices on nitrogen removal from wastewater applied at a rate of 1.27 cm in 6 hours.

Phosphorus Treatment

Overland flow can provide moderate reduction of applied wastewater phosphorus. Phosphorus concentrations in runoff waters for 2.54 cm of wastewater applied in 6 hours, 5 days per week, were significantly higher than the 1.27-cm amount (Figure 6). However, when the

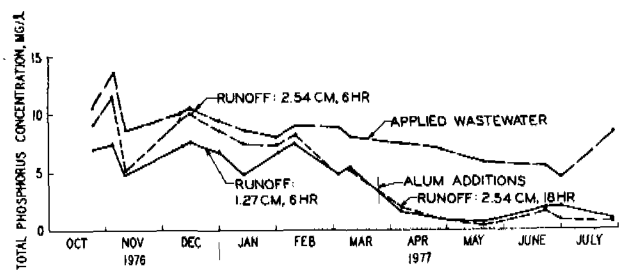

Figure 6. Influence of wastewater application rate and alum additions on total phosphorus concentrations in runoff waters from a 2-percent slope.

application period was increased to 18 hours on 31 January, the 2.54-cm application produced similar phosphorus concentrations in runoff waters as the 1.27-cm application. Addition of aluminum sulfate (alum) to the wastewater in amounts equal to 1:1, Al:P, prior to irrigation, significantly reduced runoff phosphorus concentrations for both application rates to as low as 1 mg/ℓ in May. Phosphorus reduction

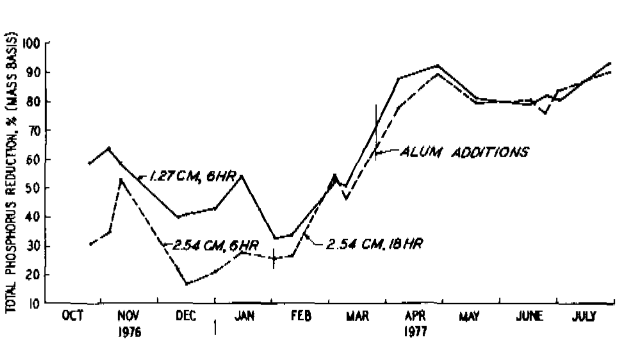

Figure 7. Influence of wastewater application rate and alum additions on total phosphorus reduction on a 2-percent slope.

was significantly lower when wastewater was applied at 2.54 cm in 6 hours compared to 1.27 cm in 6 hours (Figure 7) Increasing the application period to 18 hours on 31 January resulted in the phosphorus reduction becoming similar for both rates of wastewater application. Alum additions to the wastewater in March resulted in phosphorus reductions of 80 percent or greater for both application rates. When more stringent phosphorus removal is required, pre- or post-treatment will be necessary.

Coliform Treatment

The lagoon effluent at Utica is not disinfected prior to application since the relatively high pH of 7 to 11 makes effective chlorination costly. While overland flow has been observed to reduce coliform numbers in runoff waters during October through March, significantly larger numbers of coliforms than those in applied wastewater have been found in runoff waters during the summer months (Figure 8). The exact cause of this increased coliform count in the overland flow runoff is not clear. However, the occurrence of this event in two consecutive summers would suggest that fecal coliforms are not a reliable indicator of sanitary conditions during the summer months. Very seldom during the year has the number of coliforms in runoff water approached 200 MPN/100 ml. However, since August 1977, the continuous application of 2.54 cm of wastewater in 24 hours for 7 days per week has maintained runoff water coliform counts below this level.

Heavy Metal Treatment

Both the greenhouse and the field research indicate that heavy metals are effectively reduced from applied wastewater treated by overland flow. Over 90-percent reduction in zinc, copper, nickel and cadmium has been observed from applied wastewaters to overland flow treatment areas. Accumulations of heavy metals in the organic mat on the soil surface near the point of wastewater application has been shown previously in greenhouse research (3). Elevated levels of heavy metals were observed in the grass closest to the point of wastewater application. Similar results are expected at Utica and may present potential problems if the grass is used as forage. Long-term effects of heavy metal accumulation on the soil surface need to be determined.

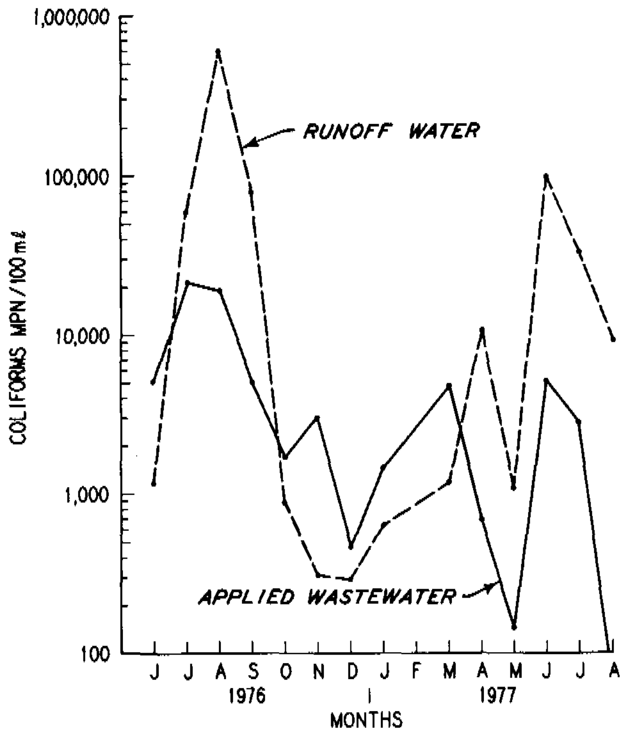

Figure 8. Influence of season on fecal coliform response at wastewater applications of 1.27 cm in 6 hours, 5 days per week.

Grass Yield

Grass yield from the Utica overland flow plots approached yields commonly obtained on better agricultural soils. The reed canary plots receiving 1.27 cm of wastewater in 6 hours, 5 days per week, produced 11,700 kg/ha of forage after three harvests. This yield

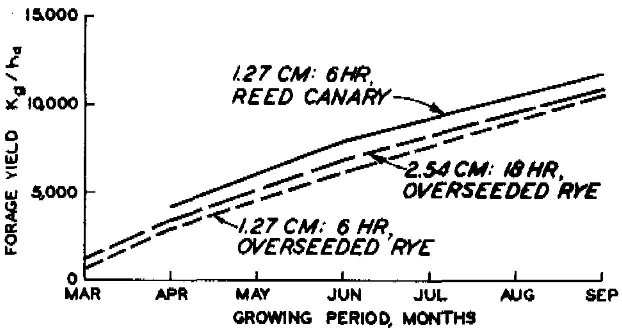

Figure 9. Forage yield from overland flow plots at Utica, Mississippi, for 1977.

compares well with reed canary yields in Minnesota by Larson et al. The overseeded ryegrass plots produced approximately 10,000 kg/ha of forage after four harvests for both 1.27-cm and 2.54-cm application amounts of wastewater (Figure 9). These data illustrate the potential productivity of marginal agricultural soil for forage production through overland flow treatment of municipal wastewaters.

SUMMARY

For municipal lagoon effluent similar to that at Utica, effective treatment for reduction of nitrogen in the southeast U.S.A. can be achieved at application rates of 2.54 cm in 18 hours for the spring, summer and fall seasons of the year. A lower application rate of 1.27 cm in 18 hours should be used during the winter season. Overseeding the treatment area with a winter cover grass may help to maintain nitrogen treatment during the winter. Phosphorus treatment by overland flow is moderate, but can be enhanced to over 80-percent reduction by addition of alum to the wastewater before application. Fecal coliforms are not a good indicator of the sanitary conditions on an overland flow system during the summer months. Forage production on an overland flow system can compare well with production on better agricultural soils.

ACKNOWLEDGEMENTS

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CASE STUDIES

NITROGEN REMOVAL PROCESSES IN AN OVERLAND FLOW TREATMENT OF WASTEWATER

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Biochemical transformations of labelled NH_4^+-N resulting from the overland flow treatment of simulated wastewater were studied in a small-scale Olivier soil-Bermuda grass test model on a 2% slope in a sealed growth chamber. The results indicate that the fraction of applied ^{15}N recovered at the end of 90-day overland flow treatments in the runoff and subflow, soil fraction, plant tissue, and gaseous phase was 7.0%, 34.6%, 22.9%, and 35.5%, respectively. A maximum concentration of 210 mg $^{15}N_2 \cdot m^{-2}$ was detected in the closed atmosphere at the end of the 90-day experiment. Significant recovery of volatilized NH_3 in the closed atmosphere confirmed the occurrence of this process in the overland flow system.

The effect of N forms on the kinetics of ^{15}N loss was also investigated in small-scale soil-plant system simulating overland flow conditions with controlled alternate aerobic-anaerobic cycles. The rate of gaseous ^{15}N loss followed a zero order reaction and was 3.93 μg $^{15}N \cdot g^{-1}$ day^{-1} for the $^{15}NO_3^-$-N treatments and 2.09 μg $^{15}N \cdot g^{-1}$ day^{-1} for the $^{15}NH_4^+$-N treatments. The absence of NO_3^- accumulation in both the experiments indicates that the denitrification reaction was not a limiting process in the N removal. The results of our study show that plant uptake, denitrification reactions, and NH_3 volatilization were all important processes that should be maximized for an efficient removal of wastewater N in overland flow treatment systems.

INTRODUCTION

Implementation of Public Law 92-500 has been a driving force in the development of land application as an alternative for the effective treatment of wastewater. This has generated interest in studying the processes involved in pollutant removal. Utilization of the entire biosystem, soil and vegetation, as a 'living filter' has been acclaimed the most ecologically appropriate wastewater treatment technique because it allows recycling of the major nutrient elements, N and P (1). The reported high renovation efficiency of N, a major pollutant in municipal wastewater, ranging from 75 to 90% in the overland runoff treatments, indicates the effectiveness of such a land treatment system (2,3,4).

The major physical, chemical, and biochemical processes involved in the removal of wastewater N include physical adsorption of NH_4^+-N by the soil exchange complex, NH_3 volatilization, immobilization of NH_4^+- and NO_3^--N by microbes, biological denitrification reactions, and plant uptake. Excellent review articles describing the mechanisms of these processes in terrestrial soils, wetlands, and lakes have been published in recent years (5,6,7). However, quantitative data on the relative importance of various N removal mechanisms in an overland flow system such as plant uptake, denitrification, and NH_3 volatilization are lacking. The use of unlabelled N source in the overland flow treatment experiments may result in discrepancies in the removal efficiencies measured for various processes due to a much larger pool of indigenous soil N compared to wastewater N (3). It is therefore necessary that labelled N should be used to compute a more reliable estimate of N removal efficiencies. This investigation was designed to evaluate the role of nitrification-denitrification reactions, NH_3 volatilization, plant uptake, and immobilization of applied inorganic ^{15}N in N removal in an overland flow treatment of wastewater.

MATERIALS AND METHODS

Olivier silt loam soil (Aquic Fragiudalfs) collected from Burden Research Plantation, Baton Rouge, Louisiana was used for this study. This soil is poorly drained, slowly permeable, and is formed from loesslike material. The surface soil (0 to 15 cm depth) has a CEC of 6.2 meq/100 g soil, a pH of 5.8, and contains 8.0% clay and 0.75% organic matter. The bulk soil samples collected were partially air-dried, passed through a 0.63-cm mesh screen, limed to pH 6.5, and used for the overland flow experiment.

Experiment 1. The Distribution of Applied $^{15}NH_4^+$-N in Various Components of Overland Flow Treatment Model in a Sealed Growth Chamber in Olivier Soil-Bermuda Grass System

Description of overland flow model. A sealed growth chamber was designed to study the removal of applied labelled NH_4^+-N in various effluent, soil, plant, and gaseous components (Fig. 1). The container for the overland flow test model was constructed of 1.27-cm thick plywood with inside dimensions of 30 cm W, 152 cm L, and 13 cm D. Two holes were drilled in the lower end of the container on one side to collect subflow. A plexiglass port was attached on the same end to collect runoff. The interior of the constructed box was coated with an epoxy paint and lined with polyethylene sheeting to prevent water seepage.

In the test container described above, 68 kg of Olivier soil calculated on oven-dry weight basis, was uniformly packed to 12.5-cm depth and planted to Bermuda grass (*Cynodon dactylon*, L.) sod. Bermuda grass was grown for several months to establish a good grass cover. An additional application of $CaCO_3$, equivalent to 10 meq/100 g soil was made on the soil surface 3 weeks prior to enclosing the soil-plant system. The experimental model was placed on a 2% slope in a room fitted with fluorescent and incandescent lights for normal plant growth. Platinum electrodes were installed in triplicate at 1-cm and 7-cm soil depths at two points across the length of the test container (20 cm and 120 cm downslope) to monitor redox potentials. Two weeks after the installation of the platinum electrodes the test container with the soil-plant system was enclosed in a 0.63-cm thick plexiglass box and sealed airtight with silicon rubber. Two sampling ports for gas analysis, a Hg manometer for monitoring the pressure of the chamber's atmosphere, a thermometer, and outlets for platinum electrode wires were installed. Provisions were made for influent addition, effluent, and condensed water collection. The temperature of the closed system was maintained at 30 $\pm$ 1°C by constantly circulating a cold water ethanol mixture (-10°C) through copper tubing running inside the closed system. A schematic diagram of the overland flow treatment model and the enclosed system is shown in Fig. 1.

Treatment application. The sealed soil-plant system was purged with a 4:1 mixture of Ar and O_2 gases for 24 hours to bring the level of N_2 gas in the system to less than 1%. Twelve liters of solution containing 23 μg $^{15}NH_4^+$-N/ml as $(NH_4)_2SO_4$ (10.372 atom % ^{15}N) and 27 μg soluble C/ml as glucose was applied to the closed overland flow system at a flow rate of 11·1 ml/minute. The runoff and subflow collected were recirculated for 4 to 5 days to create reduced soil conditions. No solution was then applied for 1 to 2 days, and the soil-plant system was allowed to reoxidize to facilitate nitrification. Seventeen influent applications, each containing ^{15}N and C, were made during the 90-day experiment period. Runoff and subflow collected at the end of each application were combined and analyzed for NH_4^+- and NO_3^--^{15}N. Gas samples were collected from the closed system at regular intervals and analyzed for labelled N_2 and N_2O gases using a Dupont Model 21-614 mass spectrometer. The condensed water collected from the

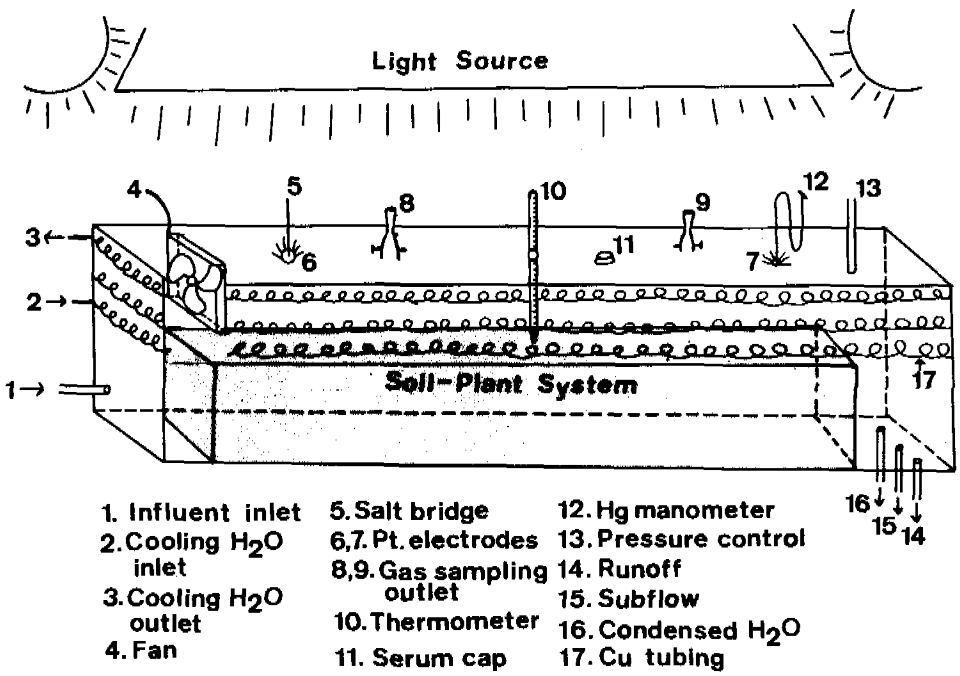

Figure 1. A schematic diagram of the overland flow wastewater treatment model in a sealed growth chamber.

closed system was analyzed for NH_4^+-N to determine the occurrence of NH_3 volatilization during overflow. Redox potential measurements were made everyday to monitor the occurrence of oxidized and reduced conditions conducive to nitrification-denitrification reactions. At the end of overflow treatments plant samples were analyzed for ^{15}N. Soil mass was sectioned with depth and downslope length and analyzed for NH_4^+-N, NO_3^--N, and organic N according to the methods described by Reddy (8). The proportion of applied $^{15}NH_4^+$ recovered in various components was calculated, as was an ^{15}N mass balance. The pH of the sectioned soil samples was also determined.

Experiment 2. Effect of NH_4^+- and NO_3^--N forms on the Removal of Applied ^{15}N in a Simulated Wastewater Treatment

The kinetics of N loss when N was applied as $^{15}NH_4^+$- and $^{15}NO_3^-$-N during an alternate anaerobic-aerobic cycle, as occurs in overland flow wastewater treatment, were studied in this experiment. Two hundred grams of Olivier soil limed with $CaCO_3$ to pH 6.5 was placed in 10-cm tall plastic containers and adjusted to 35% moisture for 1 week to allow $CaCO_3$ reaction with soil. Rye grass seeds (Lolium perenne, L.) were evenly spread on the soil surface at the rate of 0.5 g to each container and were covered until the seeds germinated. A blanket application of 100 µg P and 50 µg soluble C/g soil was added to each container. Rye grass seedlings were grown for 10 days at controlled moisture content of 20 to 35%.

Treatment Application. Labelled NH_4^+-N or NO_3^--N treatments (10.372 atom % ^{15}N as $(NH_4)_2SO_4$ and 10.059 atom % ^{15}N as KNO_3) were applied at the rate of 50 µg N/g soil on days 1, 7, 13, 18, 22, 28, and 33. Twenty-five µg C/g soil as glucose was also applied with N treatments. After each N treatment addition, moisture level in the soil-plant system was adjusted to 55% and then let drop to 20% and maintained at this level for 24 hours. This cycle of alternate flooding and drying was repeated one more time. At the end of every second cycle, 4 to 6 days after N treatment, duplicate $^{15}NH_4^+$- and $^{15}NO_3^-$-N treatments samples were taken out from the experiment. Plant samples were washed, dried at 60°C, and analyzed for ^{15}N. Soil samples were analyzed for labelled NH_4^+-N, NO_3^--N, and organic N as described in Experiment 1. Soil pH was also determined at the end of two cycles. Nitrogen treatments were applied to the remaining soil-plant containers and subjected to two alternate flooding and drying cycles. Various soil-plant components were then analyzed for ^{15}N. Soil and plant samples were taken out at the end of days 6, 12, 17, 21, 27, 32, and 38 after the start of the experiment and analyzed for ^{15}N. Loss of applied ^{15}N due to NH_3 volatilization and denitrification processes was determined by subtracting ^{15}N recovered in the soil and plant components from that applied to the soil-plant system. Kinetics of N loss in the $^{15}NH_4^+$- and $^{15}NO_3^-$-N treatments were computed using a linear zero order kinetics model.

RESULTS AND DISCUSSION

Experiment 1. The Distribution of Applied $^{15}NH_4^+$-N in Various Components of Overland Flow Treatment Model in a Sealed Growth Chamber

Redox potential measurements during overland flow treatment. The redox potentials were measured at 1-cm and 7-cm depths to determine the oxidation-reduction status of the surface and subsurface soil layers. The redox potential values of a surface layer of the soil-plant system ranged between +450 mv and +700 mv and remained well oxidized during the 90-day overland flow treatment. The redox potential values determined at the 7-cm depth fluctuated between +150 mv and +710 mv during the first 60 days of the experiment and between -180 mv and +690 mv during the latter part of the experiment (Fig. 2). It is evident from the data that well oxidized conditions near the soil surface provided a favorable environment for the nitrification of added $^{15}NH_4^+$-N retained by the soil mass. The redox potential values of the subsurface layer oscillated from reduced to oxidized to reduced conditions during the course of overland flow treatments which indicate that conditions for the denitrification of NO_3^--N that may have diffused down from the surface layer were occurring in the test model. Redox potential values of approximately 350 mv or less would cause the NO_3^--N to be unstable and subject to denitrification reactions (9).

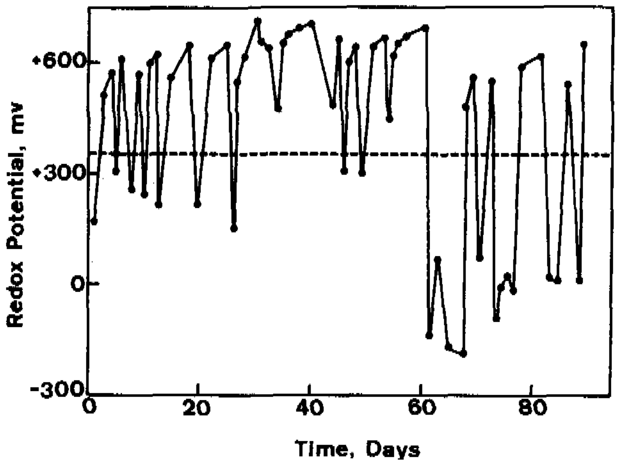

Figure 2. Redox potential values at the 7-cm soil depth as a function of time in an overland flow treatment of wastewater.

The aerobic-anaerobic zones identified in the soil-plant system would facilitate simultaneous nitrification-denitrification processes and enhance N losses to the atmosphere (8,10).

Mass Balance of Influent $^{15}NH_4^+$-N Applied to Olivier Soil-Bermuda Grass Sealed Overland Flow

The runoff and subflow collected after influent applications were made were analyzed for $^{15}NH_4^+$- and $^{15}NO_3^-$-N to determine the proportion of added $^{15}NH_4^+$-N not retained by the soil-plant system. At the end of 90-day overland flow treatments the plant root and shoot mass were analyzed for ^{15}N taken up by the plant. The soil mass was sectioned with depth and model length and analyzed for $^{15}NH_4^+$-N, $^{15}NO_3^-$-N, and organic ^{15}N fractions. A mass balance of added $^{15}NH_4^+$-N recovered in various system components is presented in Table 1. The data indicate only 7% of added $^{15}NH_4^+$ was determined in the effluent runoff and subflow. Most of the added ^{15}N recovered in the effluent was present in the NH_4^+ form. The fraction of applied ^{15}N present in NO_3^- form in the runoff and subflow was insignificant (0.25%).

Approximately 93% of added ^{15}N not found in the effluent was either taken up by the plants, adsorbed by the soil complex, incorporated into the organic fraction, or lost to the atmosphere through denitrification or NH_3 volatilization reactions.

Table 1. Mass balance of $^{15}NH_4^+$-N (4742.3 mg N) added to Olivier soil-Bermuda grass sealed overland flow system

System Component	^{15}N Recovered	
	mg N	% of added N
Effluent (Runoff & subflow)		
NH_4^+-N	321.10	6.77
NO_3^--N	11.87	0.25
Plant Uptake		
Shoots	782.10	16.49
Roots	305.60	6.44
Soil Fraction		
NH_4^+-N	584.79	12.33
NO_3^--N	15.11	0.32
Organic N	1039.65	21.92
Total N Recovered	3060.23	64.57
N Loss	1682.07	35.43
(N_2 gas)	Incomplete recovery	(2.80)
(NH_3 volatilization)	Incomplete recovery	(4.30)

The uptake of ^{15}N by Bermuda grass plants accounted for 23% of added $^{15}NH_4^+$-N. The aboveground plant shoots and primary roots accounted for 16.49% and 6.44% of total ^{15}N added, respectively. The proportion of added ^{15}N retained by the soil fraction amounted to 12.33% in the NH_4^+ form, 0.32% in the NO_3^- form, and 21.92% in the organic form. Similar to the effluent data, very little added $^{15}NH_4^+$-N was accumulated in the NO_3^- form in the soil at the end of the experiment. This indicates that NO_3^--N was not stable in the system and could have transformed immediately to other forms which will be discussed later in the paper.

A balance sheet calculated for the added ^{15}N shows that 64.57% was recovered in the effluent plant and soil fractions (Table 1). This means that 35.43% of added ^{15}N, a significant fraction, was unaccounted for. The ^{15}N may have been lost to the atmosphere through denitrification reactions or NH_3 volatilization. The sealed soil-plant system was set up to monitor the evolution of N_2 and N_2O gases due to denitrification reactions occurring in the system. The data plotted in Fig. 3 indicate that N_2 production

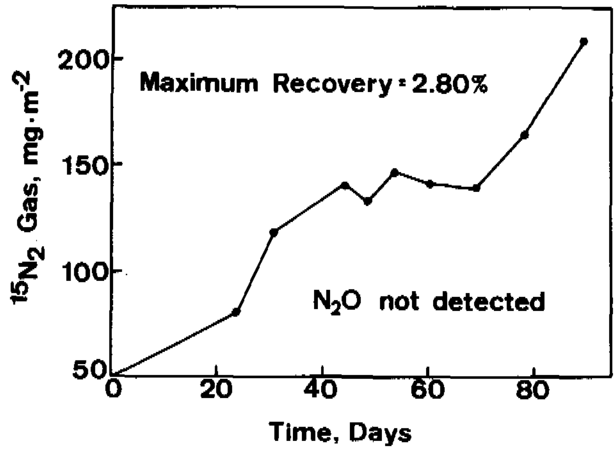

Figure 3. Gaseous $^{15}N_2$ concentration as a function of time in a sealed overland flow Olivier soil-Bermuda grass system.

steadily increased with time, although not at the same rate, and a maximum concentration of over 200 mg $N_2 \cdot m^{-2}$ was detected at the end of 90-day overland flow treatments. The proportionately greater N_2 production in the last 20 days may be attributed to more reduced soil conditions in the subsurface (Fig. 2) favorable to the biological denitrification of NO_3^--N present in the soil. No N_2O gas was detected in the closed atmosphere. The absence of N_2O in the system may be due to the high soil pH 6.6 to 7.6, which could favor N_2 formation. Studies on denitrification products show that acid conditions are more favorable for the formation of N_2O and NO than neutral and alkaline conditions which enhance N_2 (11,12). Also, due to high solubility of N_2O, traces of this gas produced in the closed system may have dissolved in the condensing H_2O and escaped the system. The recovery of N_2 gas in the enclosed atmosphere was not quantitative since a steady increase in N_2 and Ar gases in the mass spectrometric analysis was detected. This indicates that the enclosed atmosphere was slowly contaminated by the outer atmosphere, possibly through the influent water application and may have displaced $^{15}N_2$ gas through runoff, subflow, or condensing H_2O outlets. In spite of the incomplete recovery of N_2 gas, the data plotted in Fig. 2 indicate the occurrence of denitrification reactions in the overland flow system at the low redox potential levels.

The loss of applied $^{15}NH_4^+$-N through NH_3 volatilization was detected by collecting the condensed H_2O from the closed system in 0.1 N H_2SO_4 and analyzing for NH_4^+-N. The recovery of this NH_3 volatilized, although qualitative and incomplete, accounted for 4.3% of total ^{15}N added and confirmed the occurrence of NH_3 volatilization process in the overland flow system. Surface soil pH that ranged from 7.2 to 7.6 provided favorable conditions for the volatilization of applied $^{15}NH_4^+$-N. Appreciable losses of applied NH_4^+-N under alkaline soil conditions have been detected under laboratory and field conditions (13,14) and may be a significant process in N removal from the wastewater.

Based on the qualitative detection of NH_3 volatilization and denitrification products it can be concluded that 35.43% of applied ^{15}N not accounted for in the mass balance sheet was lost to the atmosphere through biological denitrification and NH_3 volatilization reactions.

Distribution of Soil ^{15}N Fraction as a Function of Soil Depth and Downslope Length

The efficiency of N removal by the soil mass was investigated by determining the concentration gradients of various N forms at the end of the experiment. The soil was divided into 0- to 37-cm, 37- to 74-cm, 74- to 111-cm, and 111- to 148-cm sections downslope from the point of influent application. Each section was then sliced to 0- to 2.5-cm, 2.5- to 5.5-cm, and 5.5- to 12.5-cm depths and analyzed for soluble and exchangeable $^{15}NH_4^+$-N, $^{15}NO_3^-$-N, and organic ^{15}N. The data summarized in Table 2 indicate that $^{15}NH_4^+$-N decreased sharply with slope, and this trend was obvious at all soil depths studied. A consistent decrease in $^{15}NH_4^+$-N concentration with depth was also observed in the 0- to 37-cm and 37- to 74-cm sections. No accumulation of $^{15}NO_3^-$-N occurred at the end of the experiment, and the concentration of $^{15}NO_3^-$-N was insignificant (0.16 to 0.37 μg $^{15}NO_3^-$-N/g soil, Table 2). The concentration gradient of

Table 2. Distribution of influent $^{15}NH_4^+$-N in various soil fractions determined at the end of overland flow treatments in sealed soil-plant system

Soil depth cm	Distance downslope, cm			
	0-37	37-74	74-111	111-148
	$^{15}NH_4^+$-N, μg/g soil			
0-2.5	33.18	14.38	4.08	1.12
2.5-5.5	28.74	12.44	1.82	0.81
5.5-12.5	14.73	4.27	2.48	1.48
	$^{15}NO_3^-$-N, μg/g soil			
0-2.5	0.32	0.36	0.37	0.17
2.5-5.5	0.21	0.26	0.24	0.17
5.5-12.5	0.20	0.24	0.21	0.16
	Organic ^{15}N, μg/g soil			
0-2.5	87.44	48.34	18.21	3.90
2.5-5.5	31.12	10.94	3.57	1.31
5.5-12.5	12.22	6.88	5.90	3.05

organic ^{15}N with depth and with distance downslope was similar to that of $^{15}NH_4^+$-N.

The accumulation of NH_4^+-N and organic N in the upper half of the test model suggests that the capacity of the soil complex to adsorb NH_4^+-N was not saturated at any point. Once retained by the soil complex the NH_4^+-N may have been immobilized by the organic C applied with the influent, utilized by the soil microbes present in the soil, or taken up by the plants. Since the surface soil was normally oxidized, it facilitated nitrification reaction resulting in NO_3^--N which diffused downward and underwent biological denitrification and loss to the atmosphere. This sequential immobilization and nitrification-denitrification processes most probably resulted in adsorption sites constantly being available for NH_4^+ adsorption, and the applied $^{15}NH_4^+$ did not move appreciably beyond half of the test model length. Similar results were reported by Hoeppel et al. (2) for the removal of cannery wastewater N in a 3-m long overland flow model. They found that about 50% of the NH_4^+- and NO_3^--N losses from the amended wastewater were incurred within the initial 30 cm. Law et al. (15) also suggested a nonlinear pattern in N removal rate with slope in a grassland with cannery wastewater spray irrigation. They reported 15.0, 0.70, and 1.0 ppm total N after 0, 12.19, and 30.48 m of overland flow, respectively. The relative distribution of these N forms may be different in the long-term overland flow models where the system may reach an equilibrium stage resulting in the mineralization of organic N. Plant removal, NH_3 volatilization, denitrification processes, NO_3^- leaching, and runoff will be the limiting factors in the removal of wastewater N under such conditions.

Experiment 2. Effect of N Forms on the Kinetics of ^{15}N Loss in Simulated Wastewater Treatment in Olivier Soil-Rye Grass System

Each application of $^{15}NH_4^+$- or $^{15}NO_3^-$-N to the Olivier soil-rye grass system established in small containers was followed by two alternate flooding and drying cycles (55% moisture to 20% moisture). Soil and plant samples were then analyzed for ^{15}N, and the distribution in various system components was studied. The amount of ^{15}N lost to the atmosphere through denitrification reactions was calculated by subtracting the amount of ^{15}N recovered in the soil and plant components from that added to the system. Data summarized in Table 3 show that the accumulation of added $^{15}NH_4^+$-N in the soluble and exchangeable fractions increased with time as additions of $^{15}NH_4^+$-N were made. About 6.6% of added $^{15}NH_4^+$-N was still present in the NH_4^+ form at the end of 38 days. Essentially no $^{15}NO_3^-$ was found in the soil during the length of the experiment. As the maximum concentration determined at the end of the last ^{15}N applications was only 0.42 μg $^{15}NO_3^-$-N/g soil. It is evident that unlike NH_4^+-N, the NO_3^- formed due to biological NH_4^+ oxidation under oxidized soil conditions was very unstable and immediately transformed into other N forms.

The incorporation of added $^{15}NH_4^+$-N into the organic fraction increased with time in a linear mode and was several orders of magnitude greater than the residual $^{15}NH_4^+$-N fraction. This organic fraction plays an important role in the overland flow treatment system as a N sink thus reducing NH_4^+ levels in the runoff waters. However, in the permanently established overland flow systems, the organically fixed N would mineralize with time and may be subject to recycling.

Labelled N applied as NO_3^- was not converted to $^{15}NH_4^+$ or accumulated as NO_3^- to any significant levels as their concentrations remained below 1.0 μg N/g soil. The applied $^{15}NO_3^-$-N may have converted to gaseous forms (N_2, N_2O) upon flooding, been taken up by the growing plants, or immobilized to organic N. The analysis of organic N shows that 2.75 μg to 23.85 μg ^{15}N/g soil was incorporated with the organic fraction. Several mechanisms are proposed in the literature for the conversion of NO_3^--N to organic N such as anoxic NO_3^- assimilation (16,17) and respiratory reduction to NH_4^+ and organic N (18). The soil conditions conducive to substantial respiratory reduction of NO_3^--N include excessive amounts of easily reducible C source and intense soil reduction. In simulated wastewater treatment systems described in Experiments 1 and 2 in our study and in others (2,3) the organic C source in the applied wastewater is generally low (<100 mg C/ℓ), and the redox potential values fluctuate between reduced and oxidized with alternate wetting and drying cycles. The dissimilatory reduction of NO_3^- to NH_4^+ under such conditions would be limited. Very low levels of $^{15}NH_4^+$-N (0.1 to 0.66 μg $^{15}NH_4^+$-N/g, Table 3) converted from $^{15}NO_3^-$-N treatment in our study suggest that respiratory NO_3^- reduction to NH_4^+-N was not a significant process in our system.

The immobilization of $^{15}NO_3^-$-N to organic N, although smaller in magnitude than the denitrification losses (discussed later in the section) is important in overland treatments in that it would reduce NO_3^--N losses through runoff, subflow, or leaching through the soil profile.

The data presented in Table 3 also indicate that the fraction of added ^{15}N retained in the soil (NH_4^+-N + NO_3^--N + organic N) was much greater in the $^{15}NH_4^+$-N amended treatments than in the $^{15}NO_3^-$-N amended treatments. Soil ^{15}N in the $^{15}NH_4^+$ treatments ranged from 11.84% to 22.19% of total ^{15}N added, whereas only 5.40% to 7.16% of added ^{15}N was retained in the soil in the $^{15}NO_3^-$ treatments. Rye grass plants (shoots and roots) accumulated up to 25% more ^{15}N in the NH_4^+ treatments compared to NO_3^- treatments. The mass balance of ^{15}N added to the soil-plant system during the 38-day experiment shows that 18.47% to 30.65% of added $^{15}NH_4^+$-N and 33.17% to 55.80% of added $^{15}NO_3^-$-N were lost from the system. Denitrification reactions resulting in N_2

Table 3. Effect of N forms on the recovery of applied ^{15}N in various soil and plant components

System components analyzed	Time, days						
	6	12	17	21	27	32	38
	N applied; μg ^{15}N/g soil						
	50	100	150	200	250	300	350
	Applied $^{15}NH_4^+$-N recovered, μg ^{15}N/g soil						
NH_4^+-N	0.39	1.11	1.91	6.85	9.64	20.39	23.04
NO_3^--N	0.08	0.09	0.08	0.19	0.23	0.26	0.42
Organic N	7.85	12.75	15.75	20.65	25.05	39.30	54.20
	Applied $^{15}NO_3^-$-N recovered, μg ^{15}N/g soil						
NH_4^+-N	0.10	0.20	0.53	0.65	0.66	0.51	0.62
NO_3^--N	0.18	0.12	0.13	0.12	0.88	0.60	0.60
Organic N	2.75	5.10	8.80	11.35	14.50	19.15	23.85
	Applied $^{15}NH_4^+$-N recovered, % of ^{15}N added						
Soil N	16.70	13.90	11.84	13.85	13.98	19.98	22.19
Plant N	63.10	60.35	59.63	55.50	60.54	61.55	57.47
N Loss	20.20	25.75	28.53	30.65	25.48	18.47	20.34
	Applied $^{15}NO_3^-$-N recovered, % of ^{15}N added						
Soil N	6.10	5.40	6.33	7.43	6.42	6.75	7.16
Plant N	38.10	46.65	60.50	52.15	48.18	56.03	47.31
N Loss	55.80	47.95	33.17	40.42	44.40	37.22	45.53

and N_2O gas production and NH_3 volatilization may account for these ^{15}N losses. However, soil pH values in the NH_4^+ treatments ranged from 5.4 to 6.4, and in the NO_3^- treatments from 6.6 to 6.8. The acidic soil conditions in the NH_4^+ treatments preclude the possibility of any significant NH_3 volatilization occurring in the system (19).

Loss of applied $^{15}NH_4^+$- and $^{15}NO_3^-$-N due to denitrification reactions increased as N additions were made with time. The data plotted in Fig. 4 suggest the rate of ^{15}N disappearance followed a zero order reaction and was independent of NH_4^+- or NO_3^--N concentrations. Similar results have been reported by Broadbent and Clark (20) and Patrick (21) who suggested that the rate of denitrification was independent of NO_3^- concentration and followed a zero order kinetics. The slope of the N loss calculated from the linear regression analysis in our experiment demonstrates that the rate of ^{15}N loss in the $^{15}NO_3^-$-N treatments was about two orders of magnitude greater than $^{15}NH_4^+$-N treatments (3.93 μg $^{15}N \cdot g^{-1}$ day^{-1} and 2.09 μg $^{15}N \cdot g^{-1}$, respectively). The complete disappearance of $^{15}NO_3^-$ in the soil fraction in both $^{15}NH_4^+$-N and $^{15}NO_3^-$-N treatments in Experiment 2 (Table 3) and in $^{15}NH_4^+$-N treatment in Experiment 1 (Table 2) strongly suggests that denitrification process was not a limiting factor in the gaseous loss of applied N.

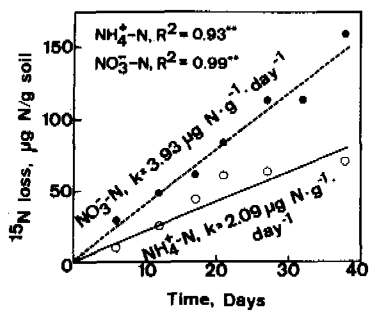

Figure 4. Kinetics of ^{15}N loss in $^{15}NH_4^+$- and $^{15}NO_3^-$-N treatments in Olivier soil-rye grass system.

NITROGEN REMOVAL PROCESSES - A GENERAL DISCUSSION

The predominant form of N in the wastewater is usually NH_4^+-N, although some NO_3^--N in the preapplication treated secondary effluent and small amounts of organic N are also present (19). When wastewater is applied to a sloped soil a variety of physical, chemical, and biochemical reactions are initiated. The effects of physical factors such as hydraulic characteristics of the soil, the friction slope, and the slope angle which can influence the direction and intensity of NH_4^+ ion movement in the overland flow system were discussed in a recent article (22). A schematic diagram of the major chemical and biological processes significant in the renovation of wastewater are presented in Fig. 5. Volatilization of applied NH_4^+-N as free NH_3 from the soil surface is an important chemical reaction resulting in losses of applied NH_4^+ to the atmosphere (14). The magnitude of NH_3 volatilization is largely controlled by pH of the wastewater, loading rates, clay content of the soil, and temperature (13,19).

Of the biological reactions nitrification and denitrification are very important. Nitrification is important because it converts NH_4^+-N, adsorbed on the exchange complex and not easily diffusible, to NO_3^--N that moves readily with the percolating water to the lower anaerobic soil layers. Denitrification is important because it is the principal process by means of which NO_3^--N is lost from the soil system through conversion to N_2O or N_2 gases (6,7). Under the conditions of our experiments 20 to 56% loss of applied ^{15}N was attributed to denitrification reactions. Patrick and Reddy (23) attributed about 25% loss of applied $^{15}NH_4^+$-N from the soil-plant system under field conditions to denitrification reactions. The results of our study also show that 23 to 62% of applied ^{15}N was removed by the plant tissue. This significantly large fraction of added ^{15}N removed by the plants confirms that plant growth may be an efficient process in wastewater treatment in overland flow systems. However, very little research has been conducted on the role of plant species in the establishment of land treatment models.

Our findings indicate that plant uptake, denitrification, and NH_3 volatilization are important N removal mechanisms in overland flow treatment of wastewater. However, quantitative data showing the relative importance of these processes are lacking. Also, more research is needed to optimize the soil-plant conditions for the maximum removal of wastewater N that will result in effective low-cost overland flow treatment with minimum groundwater and stream contamination.

ACKNOWLEDGEMENTS

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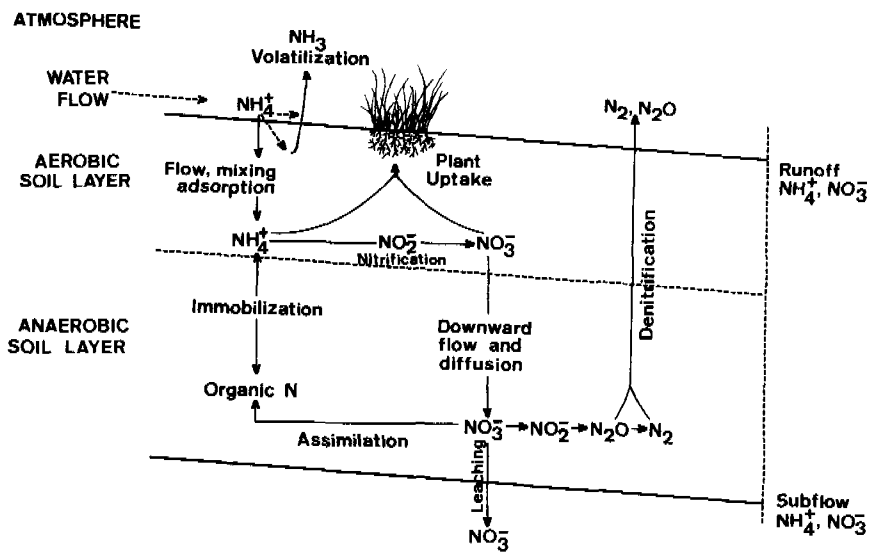

Figure 5. A schematic diagram of various N removal processes occurring in an overland flow treatment of wastewater.

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CASE STUDIES

PERFORMANCE OF OVERLAND FLOW LAND TREATMENT IN COLD CLIMATES

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ABSTRACT

In a study conducted at the Cold Regions Research and Engineering Laboratory (CRREL), Hanover, NH, primary wastewater, secondary wastewater, and tapwater were applied to separate sections of a prototype overland flow site. Each section was 2.9 m wide, 30.5 m long and graded to a 5% slope. The site was underlain by a rubber liner to ensure that water did not percolate below a depth of 15 cm. Water and wastewater were applied at the top of the slope from perforated plastic pipe. The surface and subsurface flows were collected at the base of the slope in large galvanized steel tanks. Flowmeters were used to monitor the volume of water applied and collected. Composite water samples were analyzed for pH, BOD, TOC, SS, NO_3, TKN, NH_4, total P, major cations, fecal coliform, conductivity and chloride.

The objective of this study was to evaluate the performance of overland flow systems, especially during the winter months. Operation of the CRREL overland flow facility began in May 1977 and continued through the winter of 1977-78. The results of this study indicated that satisfactory BOD removal did not occur at soil temperatures below 4°C. Based on this criterion, 105 days of storage would be needed at the CRREL site. This is 30 days less than the storage needs predicted by the EPA-1 computer program.

In addition, results indicated that secondary treatment before overland flow can be detrimental to maximum nitrogen removal efficiency due to the poor treatment of nitrate applied. Suspended solids removal was found to be excellent throughout the winter application season.

Warm weather treatment efficiences of the system using primary effluent were 94% for BOD, 97% for suspended solids, 94% for nitrogen and 89% for phosphorus. Fecal coliform was found to be a poor indicator of disinfection efficiency from an overland flow system.

INTRODUCTION

Land application of municipal wastewater has been practiced in the United States for over a hundred years. The use of this technique was primarily oriented toward disposal, analogous to discharging to a surface water body. Land application systems, with treatment as the primary goal, were relatively few in number. However, this trend has changed within the last ten years.

Wastewater systems in which effluent is applied to the land can be classified as operating in one or more of the following modes: slow infiltration, rapid infiltration or overland flow (Reed et al., 1972). The overland flow mode, called grass filtration in Australia (Seabrook, 1975), involves applying wastewater to a vegetated slope at a flow rate exceeding the infiltration capacity of the soil. The water moves by sheet flow over the soil surface and is discharged to a surface water course or given additional treatment.

The overland flow process was first studied in detail in the late 1960's by a group of investigators interested in the treatment of cannery wastes at Paris, Texas (Law et al., 1969). Their results indicated a high level of treatment for nitrogen, solids and oxygen demanding substances but only partial removal of phosphorus. In the early 1970's, the potential use of overland flow to treat municipal wastewater was investigated (Carlson et al., 1974; Thomas et al., 1974). These results indicated that overland flow could be used as a tertiary process to reduce the nitrogen content of secondarily treated wastewaters or as an initial treatment step for raw comminuted wastewater. When raw wastewater was applied, reductions of oxygen demanding substances, equivalent to secondary treatment, were achieved (Thomas et al., 1974). Phosphorus, however, was not treated to a level acceptable for discharge to phosphorus limiting surface waters. As a result, investigations were initiated to study alternate methods of removing phosphorus (Thomas et al., 1976, Lee et al., 1976).

The mechanisms of nitrogen treatment in overland flow systems have also been studied (Hoeppel et al., 1974; Khalid et al., 1978; Lee et al., 1976). Plant uptake is a major removal mechanism particularly when the crop is harvested and removed from the site. A second mechanism, denitrification, converts nitrate nitrogen to gaseous products (N_2 or N_2O) which are lost to the atmosphere. Denitrification is a microbiological process and is dependent on localized environmental constraints, namely EH (oxidation-reduction potential) and availability of degradable organic carbon. A third process, ammonia volatilization, is also operative under certain conditions (Khalid et al., 1978, Avnimelech and Laher, 1977). The ability of overland flow systems to treat applied nitrate nitrogen has not been studied in detail, but speculation by Hoeppel et al., 1974 has indicated that excellent performance was to be expected.

The use of overland flow systems to date has been limited to locations with year-round warm climates. This is due to a lack of information regarding cold weather performance and speculation of poor treatment efficiencies as temperatures approach freezing. Large storage facilities may be required to handle wastewater generated during periods when application would be curtailed. In the absence of cold weather performance criteria, state agencies have been overly conservative in regulating the amount of storage capacity required. Some states have required as much as 8 months storage for all land application systems.

Actual data relating overland flow system performance to climatic conditions have not been available to date. This study was conducted to generate criteria for predicting performance of overland flow systems as a function of temperature, allowing a more rational establishment of storage requirements.

METHODS

The overland flow facility is located at the U.S. Army Cold Regions Research and Engineering Laboratory (CRREL) in Hanover, N.H. This area has a mean annual temperature of 7°C and approximately 160 days are below freezing each year. Annual precipitation and snowfall average 95 and 185 cm/year respectively. The site faces south-southwest with prevailing winds from the west and northwest.

The overland flow site is 8.8 m wide by 30.5 m long and graded to a 5 percent slope. It was divided into three equal sections (2.9 m x 30.5 m) each of which received either primary, secondary or tapwater exclusively. The tapwater section served as a control. Water and wastewater were applied at the top of the slope through perforated plastic pipe which could be backdrained when not in use. Surface and subsurface flows were collected at the base of the slope in large galvanized steel tanks. Flowmeters were used to monitor volumes of water applied and collected. A diagram of the overland flow site is shown in Figure 1.

The soil used on the overland flow slope was classified as Hartland silt loam. The percentages of sand (≥ 50 μ), silt (50 - 2μ) and clay ($\leq$ 2μ) were 5, 72 and 23 respectively. The cation exchange capacity of the soil is about 5 mequiv/100g and the pH is about 7.1. Bulk densities averaged 1.4 g/cc and the specific gravity was 2.7. Underlying the soil at a depth of 15 cm (6.0 in) was a 1.0 mm rubber membrane preventing downward percolation of water and simulating impermeable soils required for overland flow sites. Crushed stone was placed at the top of the slope to pre-

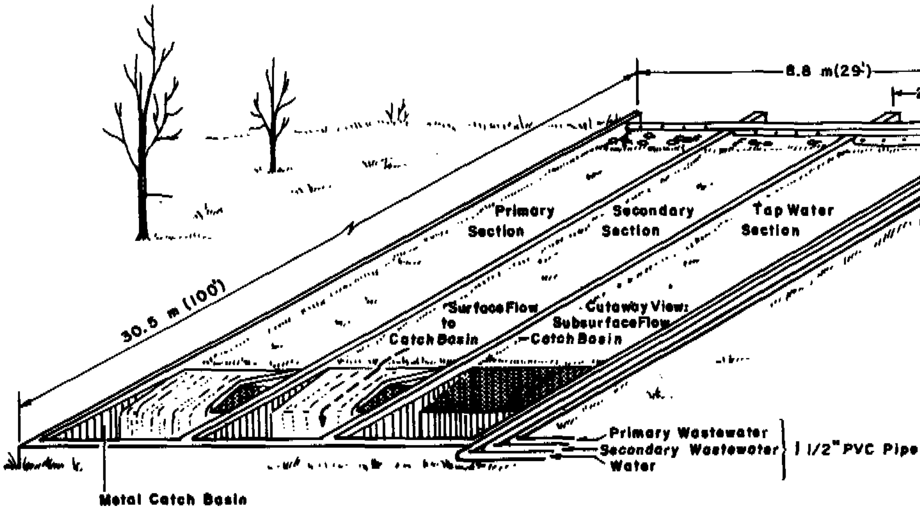

Figure 1. Diagram of Prototype Overland Flow System.

vent erosion and to allow an even flow distribution.

All three sections of the overland flow slope were seeded with orchard-grass, tall fescue, reed canary and perennial ryegrass. Orchardgrass and tall fescue became the predominant species on the wastewater sections while tall fescue predominated on the control section along with a considerable amount of red clover. The absence of red clover on the wastewater sections was probably due to a more dense and prolific grass growth which crowded out the clover. A picture of the relative growth characteristics of each section is shown in Figure 2.

The primary and secondary wastewaters used in this study were obtained from the CRREL pilot plant facility (Iskandar et al., 1976). Both wastewaters were ozonated and stored in 5300 l (1400 gal.) concrete tanks prior to application. Tapwater was obtained from the Hanover water supply system and applied directly to the site. The tapwater, primary effluent and secondary effluent were applied at a rate of approximately 5 cm (2 in.) per week. The daily application rate was 0.25 cm (0.1 in.) per hour for a period of five hours. Application ceased whenever rainfall exceeded 1.3 cm (0.5 in.) per day.

Both primary and secondary wastewater were applied throughout the winter of 1977-78. Tapwater application was discontinued in late December due to freezing problems with the pipeline system. The site was covered with snow from December 6, 1977 to March 29, 1978, which, along with the heat content of the wastewater, prevented the soil from freezing. A picture of the overland flow site during mid-winter is shown in Figure 3.

SAMPLING AND ANALYSIS

Flow proportioned composite samples were taken of the applied wastewater and runoff from each overland flow section. Runoff was also sampled after substantial rainfall or snowmelt to provide information on the "flushing" effect. Sampling of subsurface flows was discontinued after it become apparent that it was insignificant (less than 2% of applied wastewater) with respect to volume and concentration of components. All samples were refrigerated at 4°C until analyzed.

Each sample was analyzed for chloride, nitrate ($NO_3^- - N$), and ammonium (NH_4^+-N). Approximately twice a week, samples were analyzed for BOD_5. Suspended solids (total and volatile), fecal coliform, pH, conductivity, total organic carbon (TOC), total and Kjeldahl nitrogen were analyzed once a week. Periodic measurements were obtained for phosphate, potassium, sodium, calcium, magnesium and nitrite. All analyses were conducted in accordance with Standard Methods for the Examination of Water and Wastewater, 14th Ed. (1976) and Methods for Chemical Analysis of Water and Wastes (1974).

Soil temperatures were measured with thermocouples located 1.27 cm (0.5 in.) below the soil surface. Thermocouples were placed upslope, midslope and downslope on each section of the

Figure 2. View of Overland Flow Site During Summer Operation.

overland flow site. Air temperatures near the site were also measured. Every four hours these data were recorded by an automatic data logger and stored on tape.

RESULTS AND DISCUSSION

The average quality of the applied primary, secondary and tapwater is shown in Table 1. Both primary and secondary wastewaters contained somewhat higher concentrations of nitrogen than that used by other investigators (Law, et al., 1969; Hoeppel, et al., 1974). Ammonium was the most common form of nitrogen in both wastewaters. Primary wastewater contained very little nitrate whereas secondary wastewater contained up to 27 mg/l during the summer months. This was due to the nitrification process occurring in the extended aeration treatment plant. Suspended solids and BOD_5 concentrations were typical of effluents produced by primary and secondary processes. Tapwater contained no measurable pollutants.

Wastewater and tapwater application began on 13 May 1977. A two week acclimation period followed during which the system seemed to come to equilibrium. From 30 May to 16 October 1977, excellent removals of BOD, suspended solids, phosphorus and nitrogen were obtained from both primary and secondary sections. As shown in Table 2, BOD_5 and suspended solids removal were greater than 90 percent and runoff concentrations were well below secondary effluent standards. Phosphorus removal was greater than 80% for both primary and secondary sections.

Total nitrogen removal was greater from the primary section than the secondary. Higher total nitrogen concentration in the runoff from the secondary section was mainly due to a higher applied nitrate concentration (Table 2). It was found that periods of high nitrate in the runoff corresponded to days when high nitrate levels were present in the applied secondary wastewater. This relationship, which is shown in Figure 4, indicates that nitrate is not treated effectively by overland flow. Apparently, nitrate is not immobilized in the soil and remains in the mobile liquid phase. On the other hand, ammonium was effectively removed on both slopes. These results suggest that secondary treatment prior to application could decrease the nitrogen removal efficiency of overland flow systems.

The concentration of suspended

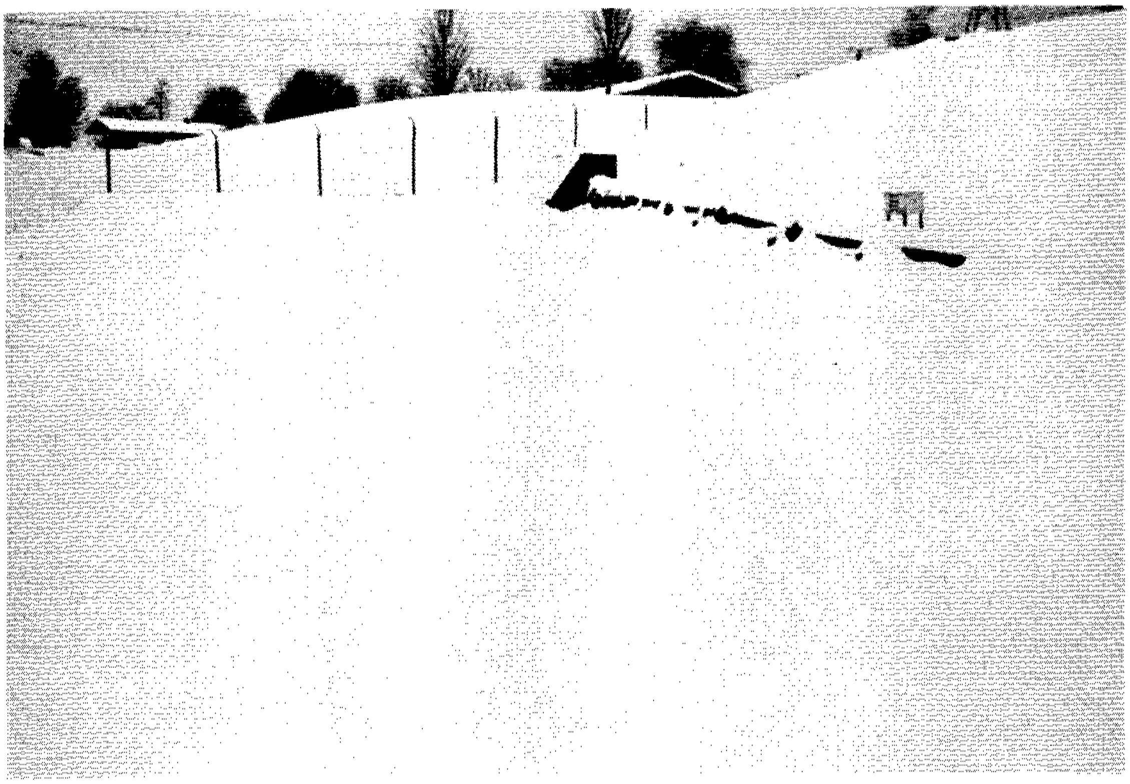

Figure 3. View of Overland Flow Site During Mid-Winter Operation.

Table 1
Average Wastewater Quality Applied
to Overland Flow Slopes
30 May 1977 - 1 April 1978

	Applicant Concentrations		
Parameter	Primary	Secondary	Tap
N(T) - N(mg/l)	36.6	33.5	0.3
NH_4 - N(mg/l)	33.1	27.3	0.1
NO_3 - N(mg/l)	0.5	5.1	0.0
P(T) - P(mg/l)	6.3	5.9	0.6
BOD_5 (mg/l)	85.3	53.2	0.4
SS(T) (mg/l)	74.6	30.2	1.4
SS(V) (mg/l)	60.7	21.7	0.7
C(O) - C(mg/l)	89.2	57.0	9.4
Cond (μmhos/cm)	524	519	91
pH (pH units)	7.4	7.5	7.1
Fecal Coliform (#/100 ml)	7.9×10^4	1.8×10^4	0
K (mg/l)	12.4	11.9	1.4

solids in the runoff from both wastewater sections was approximately equal to that from the tapwater section. Organic carbon levels were also quite similar (Table 2). This indicated that near maximum removal of those parameters had been achieved.

As shown in Table 2, the average fecal coliform concentration from the tapwater section was approximately four times higher than the secondary section. Similar results obtained by Hunt et al. (in press) suggest that fecal coliform concentration may not be a useful indicator of sanitary quality from runoff produced by overland flow sites.

After 16 October 1977 treatment efficiency began to decrease. By the end of December, system performance seemed to stabilize at the level shown in Table 3. Suspended solids were the only pollutants effectively removed

Table 2
Average Warm Weather Performance
from Overland Flow Slopes
30 May - 16 October 1977

Parameter	Runoff Concentrations Primary	Secondary	Tap
N(T) - N(mg/l)	5.4 (94%)*	8.0 (87%)	0.7
NH_4 - N(mg/l)	3.2	2.6	0.1
NO_3 - N(mg/l)	1.6	5.2	0.1
P(T) - P(mg/l)	1.9 (89%)	2.2 (80%)	0.2
BOD_5 (mg/l)	11.2 (91%)	4.6 (95%)	1.4
SS(T) (mg/l)	6.7 (97%)	3.8 (96%)	2.8
SS(V) (mg/l)	5.2	3.2	1.4
C(O) - C(mg/l)	29.0	26.0	22.0
Cond (μmhos/cm)	395	324	211
pH (pH units)	7.7	7.6	7.9
Fecal Coliform (#/100 ml)	$6.3x10^2$	18	72

*Numbers in parentheses refer to mass percent removal.

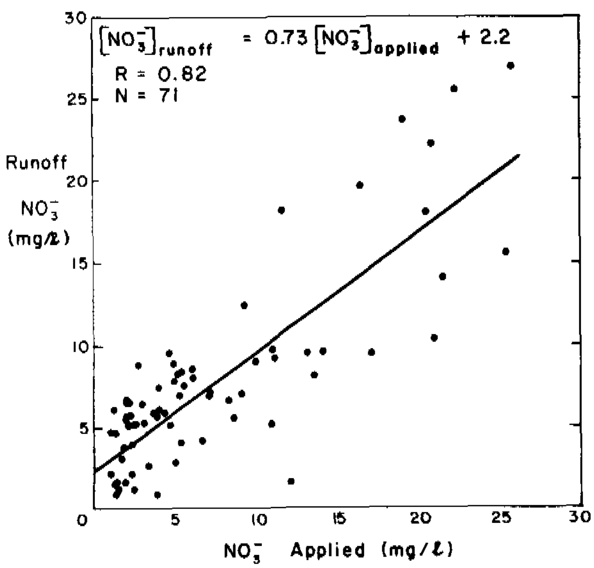

Figure 4. Runoff NO_3^- Concentration vs. Applied NO_3^- From Secondary Slope.

during this period. BOD_5 removal from the secondary section remained relatively high (80%) but removal from the primary section was only 58 percent. Both nitrogen and phosphorus removal decreased to the 25 to 32 percent range. As indicated previously, operation of the tapwater section was discontinued due to freezing problems with the pipeline system.

To determine the temperature at which the renovation capacity of the overland flow system effectively ceased, average weekly BOD_5 concentrations in the runoff from the primary section were plotted against soil temperature. As shown in Figure 5, a reasonable correlation (R = 0.78) was obtained with the second order polynomial

$$[BOD_5] = 0.226\ [\text{Soil Temp.}]^2 - 6.53\ [\text{Soil Temp}] + 53.0 \qquad (1)$$

The curve shown in Figure 5 indicates that runoff BOD_5 concentrations begin to increase as the soil temperature decreases below 14°C. However, satisfactory removal was maintained until soil temperature reached approximately 4°C where runoff BOD_5 concentrations exceeded 30 mg/l. At temperatures below 4°C, BOD_5 removal was minimal and approached the steady state condition described in Table 3.

The effect of soil temperature on ammonium concentrations in the runoff is shown in Figure 6. Runoff ammonium concentrations from both primary and secondary sections were used in this plot. An excellent correlation (R = 0.936) of this data was obtained and the curve of best fit can be described by the second order polynomial

$$[NH_4] = 0.114\ [\text{Soil Temp}]^2 - 3.94\ [\text{Soil Temp.}] + 35.1 \qquad (2)$$

Table 3
Average Cold Weather Performance
from Overland Flow Slopes
12 December 1977 - 19 March 1978

Parameter	Runoff Concentrations Primary	Secondary
N(T) - N(mg/1)	37.2 (25%)*	26.2 (32%)
NH_4 - N(mg/1)	24.3	21.5
NO_3 - N(mg/1)	2.0	3.8
P(T) - P(mg/1)	5.9 (31%)	4.4 (30%)
BOD_5 (mg/1)	65.3 (58%)	13.9 (80%)
SS(T)(mg/1)	13.6 (84%)	4.1 (88%)
SS(V)(mg/1)	11.4	3.5
C(O) - C(mg/1)	69.1	44.2
Cond (μmhos/cm)	606	616
pH (pH units)	7.2	7.3
Fecal Coliform (#/100 ml)	$8.1x10^4$	$6.3x10^3$

*Numbers in parentheses refer to mass percent removal.

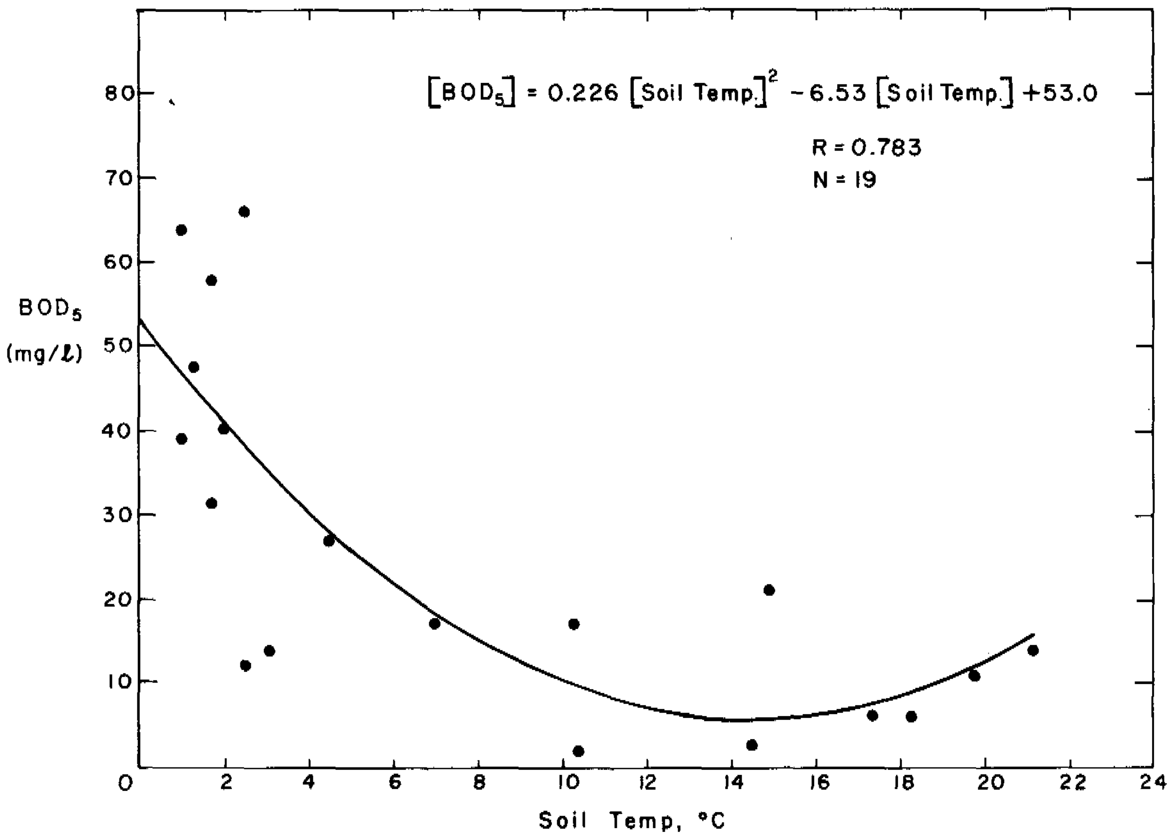

Figure 5. Average Weekly Runoff BOD Concentration vs. Soil Temperature (Primary Section).

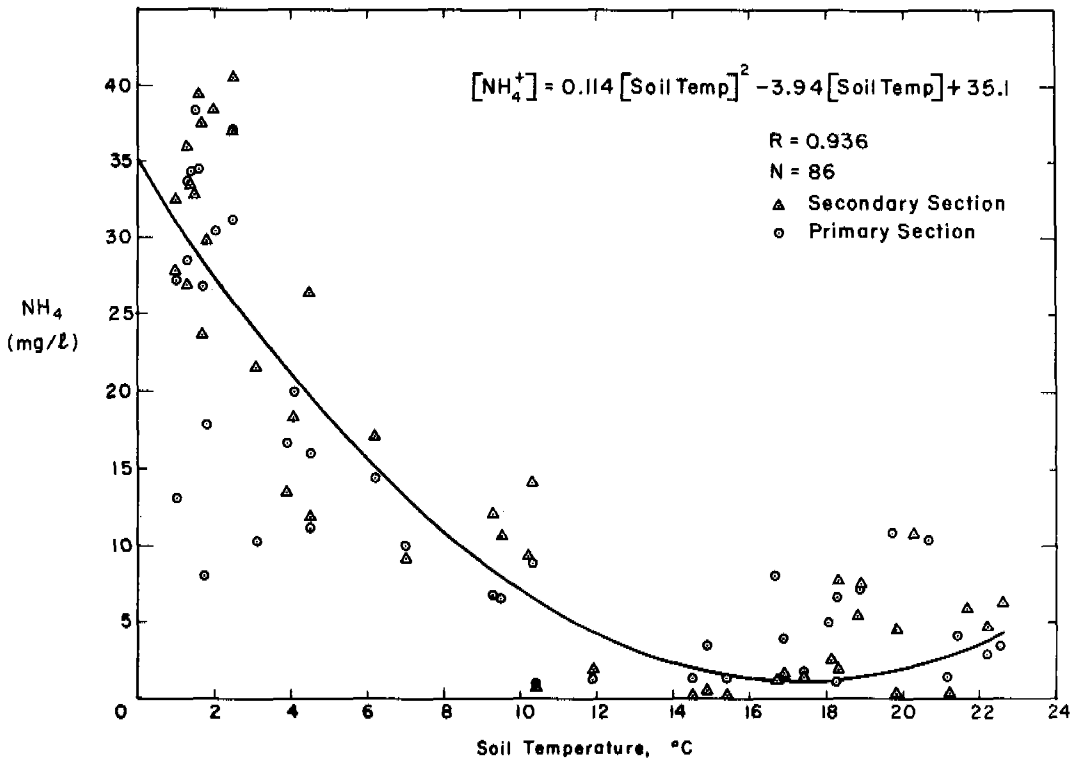

Figure 6. Average Weekly Runoff NH_4^+ Concentration vs. Soil Temperature for Primary and Secondary Sections.

From the curve shown in Figure 6, treatment of ammonium began to decrease at approximately 14°C. These results agree quite well with data presented by Metcalf and Eddy (1970) on the effect of temperature on nitrification in conventional biological treatment systems.

The number of storage days recommended for overland flow systems operating in cold climates was determined on the basis of runoff BOD_5 concentrations vs soil temperatures. As shown in Figure 5, runoff BOD_5 concentrations were greater than 30 mg/l at soil temperatures below 4°C. This indicates that wastewater applications should cease whenever the average weekly soil temperatures decrease to 4°C and resume when temperatures return to 4°C and above. If this criterion was applied to the overland flow facility at CRREL, 105 days of storage would be required. For the winter of 77-78, wastewater application would have ceased in late December and resumed in mid April.

It should be noted that even when soil temperature is restored to 4°C in the spring, treatment efficiency may not be satisfactory until the microbial population has had a chance to reestablish itself. Also, some reconditioning of the slope after a severe winter may be required. Consequently, additional storage capacity may be needed and it would be advantageous to have the ability to recycle whenever runoff quality is unsatisfactory.

Using threshold values of 0°C air temperature, 2.54 cm snowcover and 1.27 cm precipitation, the EPA-1 Program (Process Design Manual for Land Treatment of Municipal Wastewater, 1977) predicted a storage requirement of 135 days. This is 30 days longer than the storage time based on soil temperatures below 4°C. However, it should be noted that the EPA-1 Program predicts storage based on a design life of 20 to 30 years. The data presented in this paper represents only one year. Also, the difference in storage times could be reduced by modifying the EPA-1 Program and inserting new threshold values.

CONCLUSIONS

1. Wastewater application should cease whenever the soil temperature on the overland flow slope decreases to 4°C. The system should not be restarted until soil temperature is restored to 4°C.

2. The effect of temperature on ammonium removal from overland flow systems is similar to that of conventional biological systems.

3. Ammonium is more effectively removed by overland flow systems than nitrate. Evidence indicates that nitrate is not immobilized and is carried into the runoff.

4. Warm weather performance of the overland flow system was excellent. BOD_5 and suspended solids removals were greater than 90 percent.

5. Fecal coliform concentrations in the runoff were found to be a poor measure of the sanitary quality of overland flow runoff.

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SITE SELECTION

A METHOD FOR PRELIMINARY EVALUATION OF SOIL SERIES CHARACTERISTICS TO DETERMINE THE POTENTIAL FOR LAND TREATMENT PROCESSES

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A simple method for evaluating the suitability of soil series for land treatment processes is presented. Values of significant soil properties found in most soil surveys are used to achieve a preliminary assessment of the feasibility of the irrigation, overland flow and rapid infiltration land treatment processes. Depth to bedrock, slope and soil permeability are separated into ranges and assigned numerical values in a rating matrix. Matrix values reflect the relative importance of the soil property to a particular land treatment process. By combining rating values of each soil property, a rating of the suitability of a soil series results.

INTRODUCTION

This paper presents a method for preliminary evaluation of soils for land treatment processes. The land treatment process considered includes two types of slow rate irrigation, overland flow and rapid infiltration. The method presented is simple and may be most used by planners not intimately familiar with soils and land treatment processes. Existing literature on land treatment lists acceptable soil characteristics but seldom discusses the relative importance of the various soil characteristics. The approach presented in this paper allows consideration of several significant soil characteristics at the same time.

A rating matrix format can assist in the early stages of evaluating the technical feasibility of land treatment. It allows consideration of slow rate agricultural and woodland irrigation, overland flow and rapid infiltration. However, it is emphasized that the soil rating is only a portion of the early stages of land treatment planning. Other factors must be included and refinements in data made as planning progresses.

The method, based on a suitability rating matrix, was developed in part during a study (1) evaluating technical feasibility at the regional and local levels. In the study, soil suitability ratings were combined with other evaluation factors to establish technical suitability for land treatment processes.

The soil rating matrix was developed to relate the relative importance of major soil properties to land treatment process requirements. The matrix allowed evaluation of more than 90 soil series in 4 physiographic regions in the Nashville, Tennessee urban study area, covering 5,000 square miles (1). Although the matrix was developed for the Tennessee study area, it also has been found usable in the Piedmont and Coastal Plain physiographic regions and may be usable in most areas of the United States that have completed soil surveys.

Importance of Soil Properties

The literature on land treatment site evaluation includes listings of soil properties judged to be most important. Table 1 lists the characteristics emphasized by different studies. In general, except for References 1 and 2, previous work established limitation ratings for slow rate agricultural irrigation based on the most limiting soil property. No discussion of the relative importance of soil properties was presented. No consideration was given to woodland irrigation, overland flow or rapid infiltration.

Evaluation of all of the soil characteristics identified in Table 1 will aid in optimizing land treatment process design. However, it is not necessary to include all of these characteristics together to locate potential sites or evaluate potential for land treatment processes. The "Process Design Manual for Land Treatment of Municipal Wastewaters" (3) presents ranges of significant soil characteristics that distinguish the soil suitability for land treatment processes (Table 2). Soil depth, slope and permeability evaluated together should allow preliminary assessment of the suitability of a soil for irrigation, overland flow and rapid infiltration. This may seem to be an oversimplification but these properties represent and infer other soil characteristics.

Table 1. Soil properties to be Consired in Evaluating the Feasibility of Land Treatment.

U.S.E.P.A. (2); USEPA, USACOE, USDA (3):
Topography (slope, relief, flood potential), texture, structure, pH, salinity, nutrient levels (N, P, K, Mg, Ca, Na), adsorption (cation exchange) and fixation capacities, permeability, depth to water table.

Schneider and Erikson (4):
Soil texture, natural drainage, water table depth range, permeability, water holding capacity, phosphorus adsorption, slope.

Witty and Flach (5):
Limiting permeability, soil drainage class, runoff potential, flooding potential, available water capacity.

Olson (6), (for New York State):
All of Witty and Flach above, rooting depth, trafficability, erosion, stoniness, pH of B horizon, elevation.

Table 3 relates the importance of the significant soil properties to land treatment process evaluation.

Development of Soil Rating Matrix

The most appropriate classification unit for evaluating the significant properties as discussed is the soil series of a soil survey. Soil series are groupings of soils with similar properties. Except for different textures in the surface layer, all of the soils within one series have major horizons (subsurface layers) that are similar in thickness, structure and other characteristics (7). Soil series are mapped as soil phases on aerial photographs in the soil survey. A soil phase defines a unit of the soil series with a specific surface horizon thickness, surface texture, slope, erosion status or other characteristics that affect its management.

Based upon the significant soil characteristics presented in Table 2, relevant considerations presented in Table 3 and field experience, a numberical rating matrix for land treatment process evaluation was developed and is presented as Table 4. The partitioning of each soil characteristic into ranges was developed around the ranges of these characteristics commonly reported in USDA Soil Conservation Service Soil Surveys.

The numerical values assigned to ranges of each soil property are empirical. The values were assigned to emphasize the relative importance among possible ranges of a soil property in Table 4. As the suitability of a soil property increases for a land treatment processes, the value increases. An "E" value has been included where the range of a soil property would exclude a land treatment process regardless of other properties.

For example, a soil depth of 1-2 feet would in most cases preclude grading a slope for overland flow and was assigned an "E" rating value. The E results in an automatic poor rating, disregarding the rating values of other soil properties. In the soil depth range of 2-5 feet, adequate depth of soil for overland flow slope formation exists so a higher rating value is allowed. However, 2-5 feet of soil would not be as favorable as a deeper soil, therefore, an intermediate value of 5 was assigned. With soil depths greater than 5 feet no restriction to slope formation would be expected so a value of 7 was assigned for ranges 5-10 feet and 10+ feet.

The maximum value for a soil property was assigned to emphasize the relative importance between properties, e.g., low permeability is more significant for

Table 2. Ranges of Soil Characteristics Suitable for Land Treatment Processes[a]

Land Treatment Process	Soil Depth (ft)	Soil Slope (%)	Soil Permeability (in/hr)
Slow Rate Irrigation	2-3+	0-20	0.2-6
Woodland Irrigation	2-3+	0-40	0.2-6
Overland Flow	1-2+	2-15	0.006-0.6
Rapid Infiltration	10+	0-10	0.6-6

a) Modified from Reference 3

overland flow than is deep soil or shallow slopes. Thus a value of 10 was assigned to the lowest permeability whereas ratings of 7 and 8, respectively were assigned to the maximum soil depth and shallowest slopes shown in the table.

Use of Rating Matrix

How Soil Survey Information is Used in Rating Soil Series. The significant soil properties for soil series must be extracted correctly from the soil survey to get reliable results using Table 4.

Soil permeability is rated for each horizon in the soil profile. The lowest range of permeability for the horizons of a soil series in the table of soil properties in a soil survey should be used in establishing a numerical value from Table 4.

A soil series may include a very wide range of slopes, however series occur most often within a restricted range of slope. The predominant range of slope should be used to rate the soil series. The predominant slope can be estimated from the proportionate extent of phases of a soil series. A table containing the proportionate extent of soil phases can be found in the first few pages of the modern soil surveys.

The depth to bedrock listed in the table of soil properties in a soil survey is used in conjunction with Table 4. The depth of soil would be a more correct evaluation. However, soil surveys address the top 5 feet of soil and often do not describe soil, unconsolidated material or residium below this depth. Therefore, the depth to bedrock should be used.

Table 3. Impact of Soil Properties on the Evaluation of Land Treatment Process

Land Treatment Process	Soil Depth	Soil Slope	Soil Permeability
Slow Rate Irrigation	Adequate for Treatment	Tillage, Runoff, Erosion, Application System	Land Area Requirement
Woodland Irrigation	Adequate for Treatment	Erosion, Application System	Land Area Requirement
Overland Flow	Adequate for Slope Formation	Favorable for Slope Formation	Minimize Infiltration
Rapid Infiltration	Adequate for Treatment	Favorable for Basin formation	Maximize Infiltration

Table 4. Rating Factors For Soil Series[a]

Soil characteristic	Type of System: Agricultural irrigation	Woodland irrigation	Overland flow	Rapid infiltration
Soil depth, m(ft)[b]				
0.3-0.6 (1-2)	E[c]	E	E	E
0.6-1.5 (2-5)	3	3	5	E
1.5-3 (5-10)	8	8	7	4
>3 (>10)	9	9	7	8
Permeability, cm/hr (in./hr)[d]				
0.5 (<0.2)	3	3	10	E
0.5-1.5 (0.2-0.6)	5	5	7	1
1.5-5 (0.6-2.0)	8	8	2	6
>5 (>2.0)	8	8	E	9
Slope, %				
0-5	8	8	8	8
5-10	6	8	5	4
10-15	4	6	2	1
15-20	0	5	E	E
20-30	0	4	E	E
30-35	E	2	E	E
>35	E	0	E	E
Overall suitability rating[e]				
Good	20-25	20-25	20-25	20-25
Fair	14-19	14-19	15-19	15-19
Poor	<14	<14	<15	<15

[a]The rating factors are usable for establishing a preliminary rating of the suitability of a soil series; they will not replace field testing in the design of a land treatment system.
[b]Depth of the profile to bedrock.
[c]E=Excluded, automatically rated as poor.
[d]Permeability of most restrictive layer in soil profile.
[e]Sum of values from the three soil characteristics.

Arriving at a Rating. Once located, predominant slope, limiting permeability and depth to bedrock for a soil series or soil phase are compared to Table 4 and a numerical value assigned to each property. The values are summed and the total used to establish a soil suitability rating based on the ranges of numbers at the bottom of the table.

A soil suitability rating of good, fair or poor for each land treatment process results. A good rating implies suitability for the land treatment process. A fair rating means that the soil series is suitable for the land treatment process, but special considerations would apply. A poor rating infers that the soil would not be suitable for the land treatment process.

Although the rating matrix was developed around the ranges of properties commonly reported in USDA-SCS soil surveys, reported values for soil properties do not always match these ranges. In using Table 4 a flexible approach is allowed. The numbers in the table are not absolutes but may be varied within the limits between ranges in the matrix. For example, a soil series with a reported depth to bedrock of 4-7 feet should be rated with a number that compromises the values for soil with depths 2-5 or 5-10 feet. The number assigned would be based on a subjective judgement by the person compiling the rating. An example rating of an area for land treatment is presented.

Figure 1 shows approximately 40 acres closely resembling many areas of the Piedmont region of the eastern seaboard. This area is gently to moderately rolling with well developed drainage

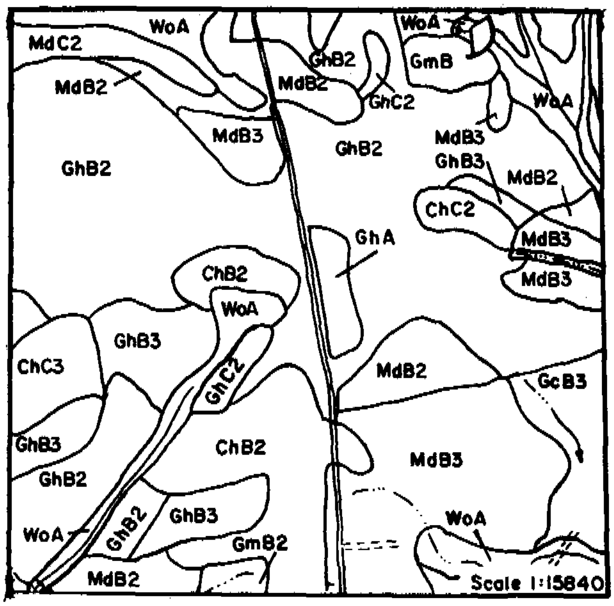

Figure 1. Example Area of Soil Map to be Evaluated (adapted from 8)

patterns. The bounded mapping units shown in the figure are soil phases. The soil phase label includes a two letter abbreviation of the soil series name, followed by a third letter denoting the slope range and, in most cases, a number at the end indicating erosion status. For rating soil series as described here, only the soil series portion of the soil phase is used.

Important characteristics of the soil series mapped in Figure 1 are listed in Table 5. These characteristics are used to rate the soil series for their suitability for slow rate agricultural irrigation in Table 6. An example of the rating method using the Maner series follows.

The Maner Series has a depth to bedrock of 1.5 to 3 meters. Therefore a rating value of 8 (from Table 4) for slow rate agricultural irrigation was assigned. The predominant slope of the Maner series is 3-8%. This does not correspond directly to the score ranges of 0-5% and 5-10% in Table 4. Therefore an intermediate rating value of 7 was assigned. The limiting permiability of the Maner series is 0.5-1.5 cm/hr. Using Table 4 a rating value of 8 was assigned.

Summing the assigned rating values the total is 23. Comparing this total to the rating scale in Table 4 a rating of good is found. The ratings of the other four soils in Table 6 were arrived at in the same way. The reader might try to duplicate the results in Table 6 starting with Tables 4 and 5. A map can be made from Figure 1 using the ratings in Table 8, rating the whole figure for agricultural irrigation. (Figure 2) Similar rating and mapping for overland flow and rapid infiltration can be done. Results should show the area of Figure 1 as mostly good for overland flow and mostly poor for rapid infiltration.

Potential Use of a Matrix. The type of simple rating matrix presented is usable at a planning level of site identification and evaluation. Rating soil series in the vicinity of a town and compiling a color coded map of soil series ratings will emphasize areas of high suitability. Selected areas could be compared for their potential for various land treatment processes.

At the areawide or regional scale, an estimate of technical potential for land treatment can be made based on soil series of the region. A preliminary assessment of soils of a region will inform local planners of land treatment potential.

Table 5. Characteristics of Soil Series Mapped in Figure 1 (Adapted from 8)

Map Symbol	Soil Series	Depth to Bedrock (m)	Slope (%)	Limiting Permeability (cm/hr)
Md	Maner	1.5-3	3-8	1.6-5
Ch	Chestor	1.2-3	8-12	0.5-1.6
Gh,Gc	Glenelk	1.5-3	3-8	0.5-1.6
Gm	Glenvilla	1.5-2.4	0.3	0.15-0.5
Wo	Worshim	1.2-2.1	3-8	0.15-1.6

Table 6. Example Suitability Rating of Soil Series From Figure 1 for Slow Rate Irrigation.

Soil Series	Soil Property Rating Value			Total	Rating
	Depth to Bedrock	Slope	Limiting Permeability		
Maner	8	7	8	23	good
Chestor	7	5	5	17	fair
Glenelk	8	7	6	20	good
Glenvilla	8	7	3	18	fair
Worshim	6	8	4	18	fair

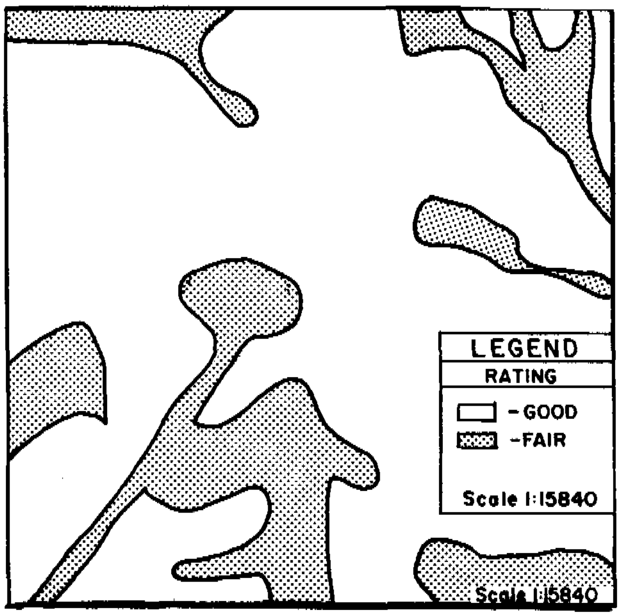

Figure 2. Suitability Map for slow rate agricultual irrigation of soils in Figure 1.

SUMMARY

A matrix rating system of significant soil properties allows rapid screening of potential land treatment site areas. Results show areas of good, fair and poor suitability and the amount and distribution of these soils. The rating will allow preliminary site and land treatment process identification, but must be supplemented by field work and consultation with a knowledgeable soil scientist prior to final design.

NOTE: The matrix approach is being pursued further and examples of utilization and improvement will be reported upon in the future.

ACKNOWLEDGEMENTS

The rating matrix was developed, in part, during a study of "Feasibility of Land Treatment of Wastewater in the Metropolitan Region in Nashville, Tennessee Urban Study Area". (see 1 below). The author acknowledges John A. Powers and Douglas Griffes, Metcalf and Eddy, Inc. for their assistance; and Carolyn Merry and Dr. H. McKim, CRREL, Hanover, New Hampshire.; Ronald Crites, Metcalf and Eddy, Inc.; and Dr. Raymond C. Loehr, Cornell University for their critical review.

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SITE SELECTION

FEASIBILITY STUDY AND PRELIMINARY SITE IDENTIFICATION FOR LAND TREATMENT IN MIDDLE TENNESSEE

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ABSTRACT

The methodology and findings of a study of land treatment feasibility in a 12,200 km^2 area in Middle Tennessee is described. The study is conducted in a two-step approach to determine the technical feasibility for four major land treatment processes: agricultural irrigation, woodlands irrigation, overland flow, and rapid infiltration.

The first step is an areawide feasibility assessment based on available planning documentation with emphasis on the effects of land use, soils, and slope. The second step is a source-specific analysis of land treatment potential in the vicinity of selected major wastewater sources. The procedure involves a more detailed approach than used in the first step, as many additional factors are considered. Alternative site areas are delineated, within which specific sites may be located.

PURPOSE

The purpose of the study was to assess the technical feasibility of land treatment of municipal wastewater in the 10-county Metropolitan Region of Nashville, Tennessee. The project is part of the wastewater management component of the U.S. Army Corps of Engineers' Metropolitan Region of Nashville, Tennessee, Urban Study [1]. The final report serves as a basis for land treatment implementation studies by federal, state, and local agencies and will provide planning options for long-range needs.

The study was limited to technical considerations related to the identification of land areas suitable for land treatment. It was not the purpose of the study to explore cost factors, implementation considerations, or to provide comparative ratings of sites. These considerations are the subject of a second phase of the project, which is a more detailed assessment for a selected community. Only the procedures and results of the first phase are reported here.

METHODOLOGY

The methodology consists of assessing the feasibility of land treatment in the study area at two distinct levels of approach: areawide feasibility and source specific feasibility. The areawide feasibility is determined in order to indicate the technical potential for land treatment in general terms throughout the study area. The source specific feasibility is determined for the purpose of demonstrating the opportunities for land treatment in the vicinities of existing wastewater sources.

Areawide Approach

The methodology for the areawide assessment is one in which broad land areas are classified for their potential as suitable site areas for land treatment. It is a no-constraints approach in that no predetermined factors are considered which would influence or limit the approach. Examples of such constraints would be the limitation of the analysis to only selected types of land treatment systems, or the consideration of only selected categories of land use for analysis.

Certain technical factors are analyzed to classify the various portions of the study area in terms of high, moderate, and low potential for the principal land treatment application methods (agricultural irrigation, woodland irrigation, overland flow, and rapid infiltration). No areas are entirely excluded from further consideration, because in many cases relatively small but high potential areas are not apparent at the areawide scale. Those areas that might otherwise be excluded are often adequate for treating the smaller flows in a study area.

Source Specific Approach

The methodology for the source specific assessment is a source-constrained approach in that potential site areas are identified within a reasonable proximity of selected wastewater sources. The investigation process initiates at the wastewater source and extends outward for a radius of several kilometres from the source, generally not exceeding 10 km, or until a suitable number of site areas are found.

The analysis typically begins with the examination of aerial photographs of existing land use and maps of projected land use surrounding a selected source to locate possible areas for sites. The largest identifiable areas are selected. This is followed by a review of topographic features that affect site selection, such as slope and local relief. Soil characteristics are simultaneously evaluated to determine land treatment potential. Since there is a significant difference in the level of detail of soils information between areas with soil surveys and areas without surveys, two distinct approaches to evaluation of soils are followed, as described later. After the soils are evaluated, other technical characteristics of the site area such as parcel sizes, geology, and groundwater considerations are identified which could influence further investigations into the feasibility of the area.

Finally, site inspections of the identified areas are conducted to ensure that major features of the areas are correctly identified and that other important aspects are not overlooked. In some cases, this results in a further refining or complete elimination of an identified area.

TECHNICAL CONSIDERATIONS

Although a number of considerations are possible in analyzing the technical potential for land treatment, an attempt should be made to limit the factors to those considered most significant and consistent with the level of detail employed at the different stages of the study. These factors are described in the following paragraphs for the areawide and source specific analyses.

Areawide Analysis

The process of identifying and rating the potential of broad areas for land treatment requires the use of identification factors that are definitive and that correspond to easily identifiable physical features. The factors must be on a scale consistent with the degree of detail of information available for areawide analyses. Generally, the factors considered most suitable to produce the desired results at an areawide scale include land cover, slope, and soils. A rating system for evaluating each of these factors is illustrated in Table 1. A brief discussion of this system follows.

Table 1. Rating Factors For Identifying Potential for Land Treatment In An Areawide Analysis

	Type of system			
	Agricultural irrigation	Woodlands irrigation	Overland flow	Rapid infiltration
Land use				
Open and cropland	High	Moderate	High	High
Partially forested	Moderate	Moderate	Moderate	Moderate
Heavily forested	Low	High	Low	Low
Built up	Low	Low	Low	Low
Slope				
0 to 12%	High	High	High	High
12 to 20%	Moderate	High	Moderate	Moderate
>20%	Low	Low	Low	Low
Soils[a]				
XYZ association	High	High	Moderate	Low

a. Actual soil association ratings not shown.

Land Use. The existing land use of an area has an obvious effect on suitability for land application and is reflected in the table. Land use information is obtained from small scale planning documents that are considered adequate for the areawide analyses.

Slope. The most important topographic feature for land application systems is the predominant slope of the land. Excessive slope may cause an undesirable amount of erosion and runoff to occur, especially during wet-weather conditions, and may also require large capital expenditures for earthwork.

In Table 1, the ratings are high for all types of land treatment systems where slopes of 0 to 12% occur. The topography is considered sufficiently flat so that runoff considerations and site preparation requirements are minimal. Areas with predominant slopes of 12 to 20% are less desirable, although they are not considered highly restrictive to the potential of an area. One exception is that of woodlands irrigation, in which the forest debris and organic mat at the surface are deterrents to runoff and erosion. Thus, the ratings for the 12 to 20% slope category are high for woodlands irrigation and moderate for agricultural irrigation, overland flow, and rapid infiltration. Slopes in excess of 20% are considered to be generally restrictive for all land treatment systems.

Soils. Soil characteristics are usually the most important factors to be considered in evaluating the technical feasibility of land treatment. The most appropriate soil classification unit for evaluating the various properties is the soil series. For mapping of large areas, however, the most appropriate unit is the soil association, which is a combination of soil series in a defined proportion.

A system was developed for rating soil series on the basis of soil depth, permeability, and slope. This rating system is described in detail in another transaction in this volume (Moser). Once each series is rated, soil associations are evaluated by combining the ratings of each of their component series. The resulting evaluation essentially provides an indication of the probability of finding soils suitable for a particular land treatment process within the soil association area.

The results of these procedures are presented on maps indicating broad areas of potential. To develop these maps, the rating factors introduced in the previous section are translated graphically into screening overlays for each method of land treatment as it is affected by each rating factor. For example, one overlay shows high, moderate, and low potential land cover areas for agricultural irrigation systems. The overlays for each rating factor are then consolidated to produce a composite map identifying the potential of areas for each method of land treatment. In order to interpret the overlapping effect of different ratings in a given location, definitions of high, moderate, and low potential are developed and are presented in Table 2. Some judgment is also exercised in working with the overlays to minimize the presentation of relatively small mapping units.

Table 2. Definition Of High, Moderate, And Low Potential Areas

Rating of land cover, slope, and soils	Designated potential
High potential for all three factors	High
Any combination of high or moderate potential ratings (except all three high potential)	Moderate
Any combination of factors which includes at least one low potential rating	Low

It is important to note that the figures produced from this procedure are based on generalized information to give an areawide perspective. Actual sites of high technical suitability may be found outside of the designated high potential areas. It should also be noted that the results pertain only to the physical potential for land treatment and do not take into consideration legal, political, environmental, social, or economic factors, nor are they related to wastewater sources.

Source Specific Analysis

The source specific analysis provides identification and preliminary evaluation of potential land treatment site areas for selected communities.

The term 'site area' is used to designate areas with potential for land treatment within which a site could be located. Thus, site areas are generally larger than required for a given method of application.

The level of detail of a source specific analysis depends on the detail of information available. A greater amount of detail is possible in counties with completed soil surveys than in counties without surveys. A soil survey allows a more detailed evaluation because the soil mapping units (covering a minimum area of 3 to 4 ha) can be evaluated directly for land treatment suitability. In areas without soil surveys, soils are mapped according to soil associations. Soil association mapping only generally describes the extent of a group of soil series that are found together, the properties of which are not homogeneous. Areas mapped by soil association are evaluated by estimating the potential for finding suitable soils within the soil association of the site area.

The factors considered most important in identifying and evaluating potential site areas are discussed in the following paragraphs.

Wastewater Flows and Quality. Flows are the basis for determining land area requirements when considered in conjunction with application rates. These factors are described below. Quality is a factor in determining nutrient loadings and site life. For municipal systems, hydraulic loadings normally govern the capacity of a site.

Water Quality Standards. The water quality standards for the receiving stream or effluent quality are important as factors in selecting the land treatment process to be used. Another consideration in the planning stages may be the impact of water quality standards on a conventional system if a land treatment system were not implemented.

Land Area Requirements. The area requirements of a prospective site are determined primarily on the basis of application rates, which are a function of the hydraulic capacity of the site for the greatest projected flow (design flow) in the life of the system. Estimates of field area requirements should be conservative in order to compensate for the many uncertainties normally associated with a feasibility analysis.

Proximity to Source. The distance of a prospective site area from the wastewater source is obviously important from a cost standpoint. However, there are a number of factors which may offset the costs related to distance, such as the lower cost of property more remote from the community. Since cost constraints are not directly considered, there are no guidelines which establish generalized limitations of distance from source.

Land Use. The land use factors considered important include urban development, vegetative cover, and parcel size.

Urban development is considered an obstacle to implementation of a land treatment site and is generally avoided wherever existing or projected development is indicated.

The vegetative cover of an area affects the type of land treatment system to be used. Normally, only predominantly nonforested areas are identified for agricultural irrigation, overland flow, and rapid infiltration systems. Although it is possible to clear a forested area for one of these systems, this would be an option developed in a detailed study. Conversely, only presently forested areas are identified for woodlands irrigation, although a woodlands system could be developed in open land (e.g., tree farms).

Parcel sizes within the identified site areas are important secondary considerations to the implementation potential for a site. In general, large parcels should be sought in order to minimize the number of parties involved in a lease agreement or land purchase. For this reason, predominant parcel sizes in the vicinities of identified site areas are determined.

Topography. The topography of a site area directly affects land preparation requirements, application rates, and runoff control. In the source specific analysis, local relief is taken into account by site area inspection and review of large-scale maps.

Soil Characteristics. As described earlier, soil characteristics are perhaps the most important of all factors considered. In the identification of site areas in this analysis, the appropriate soil series or soil association is evaluated as soil association is evaluated as previously described to delineate potential site areas from a soils standpoint. In the areas which have soil surveys, site areas and potential methods of application are identified with a higher level of confidence than in unsurveyed areas. It should be noted that, particularly in the case of areas without surveys, soil characteristics of specific sites can vary from the generalized characteristics reported. Further evaluations of potential sites would, therefore, require field soils investigations.

Geology. Important geologic factors include fracturing and solutionization characteristics of the bedrock which provide possible short-circuiting opportunities for drainage to groundwater. Generally, however, this is a problem only when the soil cover is insufficient to provide adequate treatment of the percolating wastewater. Actual identification of these problem areas can only be accomplished on a site specific basis and is not incorporated into a feasibility land study.

Groundwater Considerations. The primary concern for groundwater at the feasibility level is the depth to groundwater. Existing groundwater quality is a design consideration and must be analyzed on a site specific basis.

Hydrology. Hydrologic factors which affect site selection include surface drainage (runoff), subsurface drainage, and the susceptibility to flooding. All of these are design considerations which cannot be adequately addressed without detailed field investigations. It is noteworthy that soils generally well suited to land treatment are often found along streams and rivers where they are subject to flooding. Since flooding problems can often be mitigated by storage and proper system management, site identification is not restricted by flooding potential.

STUDY RESULTS

Following the study methodology described, the 10-county area (Figure 1) surrounding Nashville was analyzed for the technical feasibility for land treatment. The study area includes four counties with completed soil surveys, and one county with a nearly complete survey. All of the technical factors previously described were considered to some extent, although many were found to be useful only as general points of information and were not evaluated in detail.

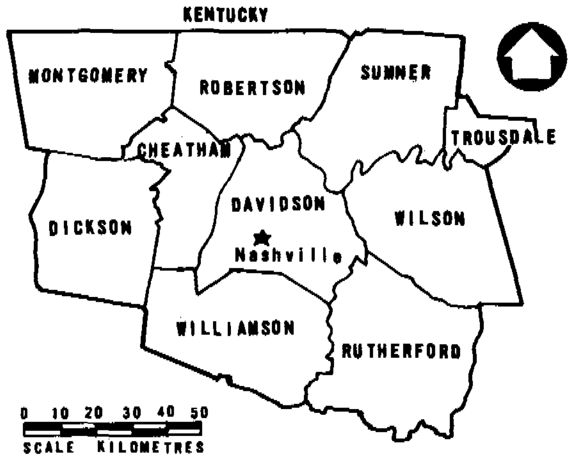


Figure 1. Study Area - Metroplitan Region of Nashville, Tennessee

The estimated application rates and land area requirements for treating a unit flow of 50 L/s of wastewater in the study area are presented in Table 3. The application rates are relatively low because many soils in the study area are of slow to moderate permeability and of relatively shallow depth. The land area requirements are used in both the areawide assessment (see Figure 2) and the source specific assessment (see Table 4).

Table 3. Application Rates and Land Area Requirements, Nashville, Tennessee

	Application rate, cm/wk	Land area required, ha
Agricultural or woodlands irrigation	0.5	255
Overland flow	2.5	47
Rapid infiltration	6.0	29

In order to assess the effectiveness of the areawide and source specific methodologies and to illustrate the use of the information presented, study results for two of the 10 counties are presented. Robertson County is a soil surveyed county in the north central study area, and Sumner County is an unsurveyed county bordering Robertson County to the east. These counties were also selected to compare the effects of different levels of soil information on the analysis. Although they do not represent all the features of the study area, these counties are physiographically similar and each contains communities representative in size of many of the communities studied.

Areawide Analysis

Results of the areawide analysis are shown in Figure 2 for the two counties (original scale 1:250,000). This figure illustrates the broad areas identified with potential for agricultural irrigation. The high potential areas are those with flat to gently rolling topography and deep, moderately permeable soils. These areas are presently farmed for field crops and pasture. Robertson County represents the most extensive high potential areas of all the counties in the study area.

studied, based on the values derived in Table 3 and using projected flows for the year 2000. These area requirements provide a perspective of the relative probability for finding sites with moderate to high potential for land treatment. It is inferred from this figure that there should be numerous opportunities for finding high potential areas in the vicinities of Springfield, Greenbrier, and Portland. Likewise, the prospects seem moderately good for finding agricultural irrigation sites in the vicinity of Gallatin. It appears that the least opportunities are available for Ridgetop and Hendersonville. These observations correlated closely with the information developed in the source specific analysis.

Similar analyses were performed for woodlands irrigation, overland flow, and rapid infiltration. The potential for each of these processes was less extensive than for

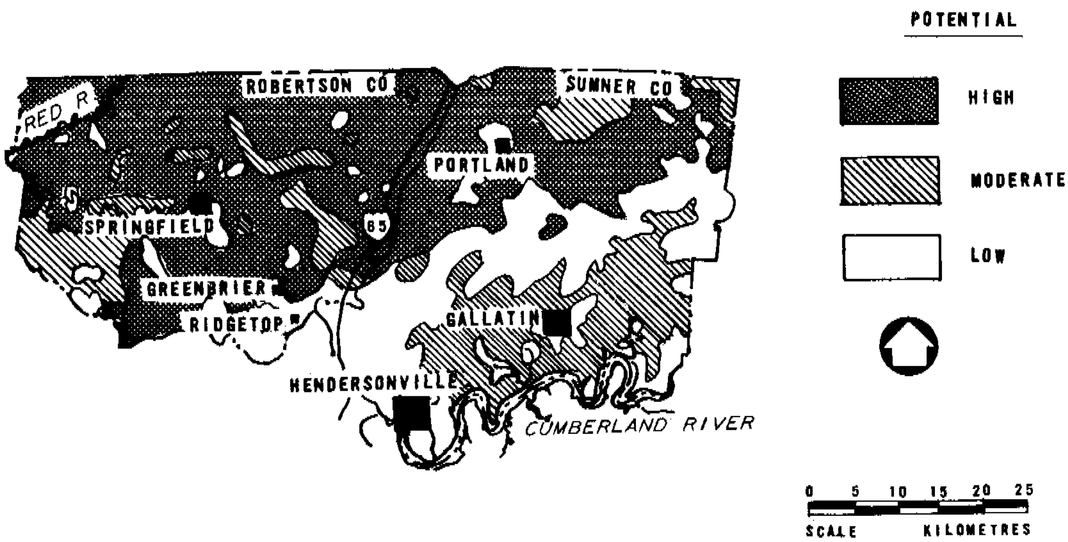


Figure 2. Potential for Agricultural Irrigation, Robertson and Sumner Counties

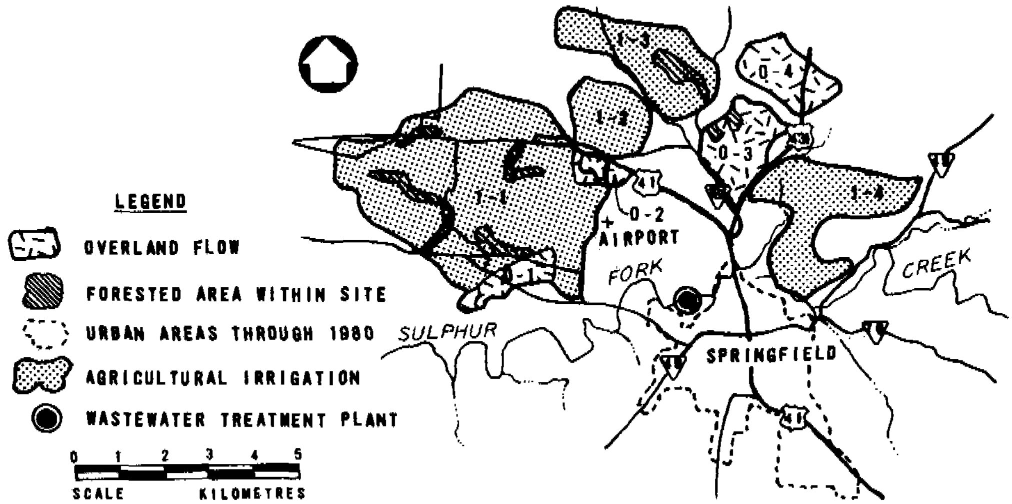
The moderate potential areas generally have good soils but are partially forested or have moderate to steep slopes. In the southern part of Sumner County, the soils of the flat, open areas are generally shallow and not considered as having high potential for irrigation. Low potential areas are generally those with steep slopes or heavy forestation.

Superimposed on Figure 2 are graphical representations of the land area requirements for agricultural irrigation for each of the communities

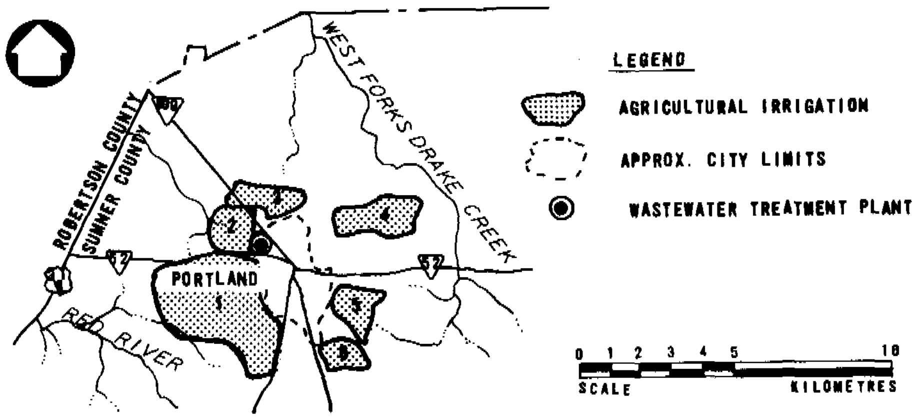
agricultural irrigation and, for simplicity of presentation, are not included here. The results of these analyses were also confirmed by the source specific analysis.

Source Specific Analysis

For presentation purposes, the results of the source specific analysis are limited to one community in each county, and are shown in Figure 3 and Table 4. The site areas were identified in accordance with the



POTENTIAL LAND TREATMENT SITE AREAS FOR SPRINGFIELD



POTENTIAL LAND TREATMENT SITE AREAS FOR PORTLAND

Figure 3. Potential Land Treatment Site Areas for Portland and Springfield

methodology and factors described earlier, and the table provides a generalized summary of important features of each site area.

The distinction between surveyed and unsurveyed areas is reflected in the figure and the table. In surveyed areas, the potential for land treatment is more accurately determined because of better soil information and, as a result, the site areas can be delineated at a relatively large scale of mapping; hence, the larger scale for Springfield in Figure 3. The original mapping scales for soil evaluation were 1:16,000 for soils mapped areas and 1:250,000 for unmapped areas. In the table, the surveyed areas are described in terms of predominant soil series; whereas the unsurveyed areas are described in terms of predominant soil association.

Visits were conducted to the identified site areas to verify the potential of the areas and to finalize site area boundaries. In some cases, recent housing developments were found within the boundaries and some areas were eliminated. Generally, however, the analytical procedure for identifying site areas proved to be fairly accurate.

CONCLUSIONS

Land treatment was found to be feasible by one or more land treatment methods throughout most of the 10-county study area. High potential areas with sufficient land availability for irrigation or overland flow sites were identified in the vicinities of all the mmunities studied (largest projected

Table 4. Potential Land Treatment Site Areas For Portland and Springfield

Site area	Total area, ha[a]	Distance from source, km	Predominant soil association or series	Current land cover or use	Predominant parcel size, ha	Elevation difference, source to site area, m	Remarks
Portland (unsurveyed)			Association				
1	975	1.6-5	Pickwick-Pembroke-Cumberland	Cropland and pasture, some forest	40	(-3) to 6	Broad, flat to rolling slopes; pasture, field crops and occasional woodlands, few houses; large parcels.
2	170	0.8	Pickwick-Pembroke-Cumberland	Cropland and pasture	12	(-9) to 3	Very broad, gentle slopes, pasture and field crops; few houses; moderate parcel size.
3	150	0.8-1.6	Pickwick-Pembroke-Cumberland	Cropland	16	(-3)	Flat to gently sloping; field crops; several houses, factories south of site; moderate parcel size.
4	275	3.2	Pickwick-Pembroke Cumberland	Cropland and pasture, some builtup	20-40	(-3) to 3	Generally flat with choppy micro-relief; open and cropland; karst topography; subdivisions near site area; moderate parcel size.
5	210	4.8-6.4	Pickwick-Baxter-Cumberland	Cropland and pasture, some forest	12	(-12) to 0	Rolling hills at the head of a drainage basin; pasture, field crops; few houses, moderate size parcels.
6	145	6.4	Pickwick-Baxter-Cumberland	Cropland and pasture, some forest	12	(-3) to 6	Broad, flat to gently sloping; field crops; few houses; industrial area to north; fountain head to south.
Springfield (surveyed)			Series				
I-1	1785	4	Pembroke, Baxter, Dickson, Mountview	85% farm 15% woods	30-60	15 to 45	Large open flat to rolling fields; mostly agriculture, some woodland; few houses; few small parcels; some sinkholes in northern part of site.
I-2	270	4	Mountview, Dickson, Pembroke, Pickwick	90% farm 10% woods	60	45	Flat to gently rolling fields; agricultural cropland; scattered sinkholes in western site area; few houses; moderate to large parcels.
I-3	440	4.8	Pembroke, Baxter, Hamblen, Crider	80% farm 20% woods	12-200	45	Flat to rolling; poorly drained woodland in center of site; karst topography; dissected by 2 major roads; several small and one very large parcel.
I-4	595	4	Pembroke, Crider, Dickson	80% farm 20% woods	10	0 to 38	Gently rolling farmland dissected by stream valleys; karst topography mixed with developed drainage; scattered sinkholes; few houses; many small parcels.

a. Portland land area required = 200 ha; Springfield land area required = 530 ha, based on land area requirements in Table 3 and projected flows for the year 2000.

population--73,000) except metropolitan Nashville (projected population--626,000). Rapid infiltration was generally not feasible because of slow to moderate soil permeabilities throughout the study area.

The methodology used in the areawide analysis was found to provide fairly accurate and consistent predictions of land treatment feasibility for the scale at which the study was conducted. This was confirmed by the identification of potential site areas in the source specific analysis within the generalized areas of potential identified in the areawide analysis. With some exceptions, the definitive distinctions expected in the source specific analysis between soil surveyed areas and unsurveyed areas were not readily apparent in the presentation of the results. Since soil surveys do provide relatively detailed information for a feasibility level study, it may be concluded that land treatment potential may be determined with a higher degree of confidence using soil surveys than is possible if surveys are not available. Field studies would be necessary to fully evaluate the technical suitability of a site area and to provide preliminary design criteria for a land treatment system.

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SITE SELECTION

PHOSPHATE ADSORPTION CAPACITY AND CATION EXCHANGE CAPACITY OF 35 COMMON SOIL SERIES IN NEW YORK

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A cross section of New York State soils were analyzed for phosphate adsorption capacity, cation exchange capacity and related parameters. When the soils were grouped on the basis of parent materials or on the basis of B horizon taxonomy, there were highly significant differences in these chemical properties between the subgroups. For phosphate adsorption, tills were better than outwashes and acid soils better than calcareous ones. For cation exchange capacity (CEC), however, the calcareous soils were slightly better than the acid ones, and the clays gave the highest total capacities. It was also found that the phosphate adsorption capacity was greatest in the A & B horizons and decreased with increased depth.

INTRODUCTION

There has been renewed interest in putting wastewater on land. This has proved especially economical for small plants in unpopulated areas where suitable land is available. Soils have considerable ability to adsorb phosphates and heavy metals in wastewater and to protect water courses from these compounds. The heavy metal retention capacity is often limited to 10% of the soil cation exchange capacity (1).

Phosphate and heavy metals can be removed by several mechanisms: (A) Rapid removal or sorption. (B) Slow mineralization and insolubilization. (C) Plant uptake and (D) Biological immobilization (2, 3, 4, 5, 6). Rapid removal submechanisms could include ion exchange, chemical reaction and physical adsorption. Although the exact submechanisms are not clear, the Langmuir isotherm has been used successfully by a number of authors to describe this rapid removal. This article will give data on this rapid removal ability for 35 New York soil series. After 2-5 days (the period of an isotherm test) the additional P removal on soils slows and is a topic of research. Some of this research data indicated that the ratio of four months P retention to the 5 day retention was in the 1.5 to 3.0 range for a number of soils (7). The range of P adsorption values from the isotherm tests on many soils is much greater (approximately 1-100 range). Even though the isotherm values are of the main parameters characterizing soil adsorption of phosphate, other factors including pH, contacting solution, temperature, redox potential, oxygen status, soil moisture tension, and time also have an effect (8, 9). Equations for predicting the movement of phosphate through soils as a "shock layer" were given by Shaw and Novak, et al. (10, 11).

Vijayachandran and Harter (12) studied 19 soils and correlated PO_4 adsorption with other soil chemical properties. Leachable aluminum and organic matter were highly correlated. Ellis and Erickson (5), in a study of 29 Michigan soil profiles for phosphate adsorption, found that for sandy soils and podzols, the B horizon generally adsorbed the most phosphate followed by the A and C horizons. Bailey (13) noted that acid soils and soil formed on igneous rocks had higher phosphate adsorption capacities than other soils.

METHODS AND PROCEDURES

As indicated in Table 1, 35 soil series of varying parent material were sampled. A pit was generally dug with a backhoe and the major subhorizons within the A, B and C horizons were sampled. Analyses included bulk density, pH, size distribution, and cation exchange capacity on all samples. Fe, Al and Ca mineralogy were determined on many of the C horizon samples. Deep samples (8-26 m; 28-85 ft.) from 13 areas in New York State were also analyzed for phosphate adsorption capacity. The pH was determined on a 1:1, soil:distilled water mixture. The soils were first air dried, crushed and sieved. Chemical analyses were performed on the size fraction less than 2 mm. Cation exchange capacity (CEC) was determined using the standard

Table 1. SOIL SAMPLES LOCATIONS

No.	Soil Series	Parent Material	Comments	Horizons(Depth in cm)
1	Adams	Sandy outwash		A12, B21ir, B22, B23, C1, C2 10 20 51 74 117 183
2	Becket 1	Granite, gneiss and coarse till	C fragipan	Ap, B21ir, B22ir, B23, Cx 13 20 38 74 127
3	Becket 2		C fragipan	O&A, B21ir, B22ir, B23, IICx 18 33 46 74 183
4	Bangor	Granite, gneiss and till		A1, B21ir, B22ir, B23, B3, C1, C2 10 23 30 51 71 97 127
5	Charlton	Granite, gneiss and till		Ap, B21, B22, B3, C1, C2 15 25 48 76 127 203
6	Chenango	Gravel, coarse outwash	C calcareous	Ap, B21, B22, B23, C 20 53 127 191 216
7	Collamer	Silty lake beds	C calcareous	Ap, B&A, B2t, C1, C2 25 36 66 122 229
8	Colonie	Sandy outwash		A, B21, B22, B23-4, B25-6, C1, C2, C3 5 28 33 91 178 234 366 457
9	Colton	Sandy outwash terrace		A2, B21h, B22ir, B23, C1, C-25 5 13 30 51 76 762
10	Elmwood	Sand over lake beds and clay	C calcareous	Ap, B, A'2, II'2, IIC 23 43 53 124 213
11	Elnora	Sandy outwash		A, B21, B22, B23, C1, C2 15 46 69 102 183 213
12	Erie	Shale, till	B fragipan C calcareous	Ap, B2, A'2, B'xt, B3, IIc 18 36 61 84 142 198
13	Essex	Till, granite, gneiss	C fragipan	A1, B21, B22, B3, Cx1, Cx2 15 30 46 61 91 150
14	Gilpin	Residuum, shale, siltstone		A&O, B21, B22t, B23, C 5 15 43 58 76
15	Hermon	Till, granite, gneiss		Ap, B21hir, B22, B3, C1, C2, C3 25 33 43 64 114 127 203
16	Hinckley	Gravelly outwash	C2 calcareous	Ap, B21, B22, C1, C2, Coarse/Fine sand 13 25 71 157 203 1220 1830
17	Honeoye	Loamy till with sandstone, limestone and shale	C calcareous	Ap, B&A, B21t, B22t, IIC1, IIC2 25 38 61 81 140 208
18	Howard	Gravelly outwash terrace	C calcareous	Ap, B, A'B, B2t, C 23 46 66 112 198
19	Lackawanna	Sandstone, till	B fragipan	A&O, B21, B22, B<, C 8 23 43 124 216
20	Langford	Shale, till	B fragipan	A1, B21, B22, B3, A'2, B'x, IIC 10 43 61 84 104 147 211
21	Lansing	Sandstone, shale and limestone till	C calcareous	Ap, B4A, B2t, C1, C2, Danley Bit 20 33 76 137 178 366-457
22	Lordstown	Grey shale and sandstone till		Ap1, Ap2, B2, C1, C2, C3 6 18 33 56 102 122
23	Mardin	Sandstone and shale till	B fragipan	Ap, B21, A'2, Bx, C1, C2 25 43 51 89 145 191
24	Morris	Red sandstone and shale till	B fragipan	Ap, B2, Bx, C 15 33 165 203
25	Ontario	Limestone, sandstone and shale till	C calcareous	Ap, B1, B2t, C1, C2 23 43 81 117 152
26	Oquaga	Red shale and sandstone till		O2, B1, B21, B22, C 0 13 30 64 81
27	Palmyra	Gravelly outwash	C calcareous	Ap, B1, B2t, C1, C2 20 43 76 119 178
28	Paxton	Schist and granite till	Fragipan	A1, B1, B2, Cx1, Cx2 15 36 71 102 152
29	Rhinebeck	Clayey lake beds	C calcareous	Ap, B4A, B21t, B22t, IIIC1, IIIC2 18 33 43 76 178 183
30	Riverhead	Gravelly outwash		A1, B21, B22, B31, IIB32, IIIC1, C2, C3 3 18 41 61 94 152 183 213
31	Schoharie	Silty clay beds	C calcareous	Ap, B&A, B21t, B22t, C1, C2, C3, C4 15 25 43 64 91 152 191 287
32	Shapleigh	Granite till		A2, B21, B22 3 10 51
33	Sodus	Sandstone till	C calcareous	Ap, B2, A'2, B'x, C 15 36 61 91 152
34	Vergennes	Lacustrine or marine clay		Ap, B&A, B2t, C 23 38 64 152
35	Wareham	Wet sands, outwash		Not designated 0-183 to 518
36	Windsor	Sandy outwash		A, B21, B22, C1, C2, C3 36 51 71 127 157 188
37	Beach Sand			0-15

ammonium acetate procedure, and phosphate capacity was established with a .01 M NaCl contacting solution at 25°C. More details on the methods and data are given in another report (14).

RESULTS AND DISCUSSION

For all horizons of the 35 soils samples, the phosphate adsorption maximum (B max) ranged from 2.8 to 277 mg P/100 gm of soil and averaged 38.2. The amount of P adsorbed at a 5 mg P/l equilibrium concentration (x/m @ 5) and the CEC also varied over a 1-100 factor range. Table 2 gives the horizon means for all the soils. The B horizon B max and x/m @ 5 were on the average 2.5

Table 2. HORIZON MEANS

	Thick. cm	Bulk Den.* gm/cc	B max* mg P/100 gm	x/m @ 5* mg P/100 gm	CEC* meq/100 gm
A	6.3	1.32	45.7	30.5	18.7
B	29	1.60	42.9	33.4	8.8
C	39	1.72	18.9	12.2	6.5

*Corrected for the % of coarse fragments (CF) greater than 2 mm.

times that for the C horizon. For some soils this factor was 5.0, and .6 m (2 ft.) of B horizon was equal to 3.0 m (10 ft.) of C horizon in total capacity to remove phosphate. This is of importance; in most wastewater rapid infiltration systems, the B horizon is generally not utilized. Modified designs to allow application of the wastewater above the B horizon are needed if phosphate removal is to be maximized. For CEC, the A horizon has the greatest unit capacity but the lowest total capacity due to its small thickness.

To simplify the data for analysis, a single A horizon and thickness-weighted B & C horizons were obtained by calculation for each soil from the detailed horizon data. The soils were then grouped by four methods. The first grouping utilized the new national soil classification (Table 3) and had 8 subgroupings:

Table 3. GROUPING I

Alfisols	Entisols	Inceptisols	Spodisols	Ultisols
2 Aqualfs	2 Psamments	1 Aquepts	5 Fragiorthods	1 Hapludults
9 Udalfs		7 Ochrepts	7 Haplorthods	

The numbers refer to the number of soils in that subgroup. Grouping system II is given in Table 4, based on a 1/250,000 scale soil map of New York State by Arnold et al. (15). Grouping III was based on a parent materials classifica-

Table 4. GROUPING II KEY FOR WEIGHTED HORIZONS

Key No.	Description and Soils Included that were Sampled
0.	Other areas not classified: 14* Gilpin.
1.	Areas dominated by coarse textured mesic soils with sand and gravel substrata. -Dominantly to excessively well drained: 8 Colonie, 11 Elnora, 16 Hinckley, 36 Windsor
2.	Areas dominated by medium and moderately coarse textured mesic soils with sand or gravel substrata. -Dominantly well drained: 6 Chenango, 18 Howard, 27 Palmyra, 30 Riverhead
3.	Areas dominated by medium and moderately coarse and coarse textured frigid soils with sand and gravel substrata. -Dominantly to excessively well drained: 1 Adams, 9 Colton
4.	Areas dominated by medium and moderately coarse textured mesic soils with compact loamy substrata. -Dominantly and moderately well drained with a calcareous substrata: 12 Erie, 17 Honeoye, 21 Lansing, 25 Ontario
5.	Areas dominated by medium and moderately coarse textured mesic soils with fragipans and compact substrata. -Dominantly and moderately well drained: 13 Essex, 19 Lackawanna, 20 Langford, 28 Paxton, 33 Sodus
6.	Areas dominated by medium and moderately coarse textured frigid soils with fragipans or compact substrata. -Dominantly well drained soils having sand or gravel substrata: 2 Becket No. 1, 3 Becket No. 2, 4 Bangor, 15 Hermon
7.	Areas dominated by coarse to fine texture, stone free soils with silty or clayey substrata. -Dominantly to moderately well drained: 7 Collamer, 8 Elmwood, 29 Rhinebeck, 31 Schoharie 34 Vergennes
8.	Areas dominated by moderately deep and shallow soils over bedrock. -Dominantly well drained: 5 Charlton, 22 Lordstown, 23 Mardin, 24 Morris, 26 Oquaga

*These numbers are from an alphabetical listing of the soils from 1 to 36.

tion taken from a New York State soil map by Cline (16). This grouping system is given in Table 5 and had 7 subgroups. Grouping IV combined aspects of Grouping I and III (parent materials) and had 6 subgroups. It was based on the taxonomy of the B horizon (14).

The data was then subjected to a two-way (subgroups and horizons) analysis of variance using Scheffe's approximation for unequal numbers of soils per subgroup via the NWAY 1 computer program (17). This was done for the first 3 grouping systems and for each of the variables indicated in Table 6. The F values given were highly significant for most parameters with respect to horizons and with respect to subgroups. Grouping systems II and III had the highest F values and significance levels.

The next step involved looking at the A, B or C horizons individually within the previous grouping systems noted in Table 7. Here, a one-way (soil subgroups) analysis of covariance was

Table 5. GROUPING III KEY FOR WEIGHTED HORIZONS

1. Granite, gneiss till

2, 3*	Becket 1, 2
4	Bangor
5	Charlton
13	Hermon
28	Paxton
13	Essex

2. Sand and Gravel Outwash

1	Adams
9	Colton
8	Colonie
16	Hinckley
30	Riverhead
36	Windsor
11	Elnora

3. Sand and Gravel Outwash, Calcareous

6	Chenango
27	Palmyra
18	Howard

4. Silt and Clay Beds

31	Schoharie
7	Collamer
29	Rhinebeck
34	Vergennes
10	Elmwood

5. Sandstone, Siltstone and Shale till

23	Mardin
24	Morris
26	Oquaga
19	Lackawanna
20	Langford
22	Lordstown

6. Limestone, Sandstone and Shale till Calcareous

12	Erie
17	Honeoye
25	Ontario
33	Sodus
21	Lansing

7. Shale and Siltstone

14	Gilpin

*These numbers are from an alphabetical listing of the soils from 1 to 36.

Table 6. F VALUES FOR TWO-WAY ANALYSIS OF VARIANCE

Variable	Grouping I Sub-Groups	Grouping I Horizons	Grouping I Interact	Grouping II Sub-Groups	Grouping II Horizons	Grouping II Interact	Grouping III Sub-Groups	Grouping III Horizons	Grouping III Interact
Thick. cm	7.0	55.2	1.92!!	4.73	61.7	NS	3.58	40.2	NS
B.D. gm/cc	4.67	26.8	NS	3.85	33.5	NS	3.72	29.5	NS
Coarse Frag. %	2.87!	NS	NS	7.09	2.57**	NS	25.2	5.26	NS
Silt and Clay %	8.42	NS	NS	46.3	3.23*	NS	6.15	4.34!	NS
pH Units	10.8	3.37!!	NS	18.0	6.10	NS	33.7	6.17	NS
B max mg P/100 gm	7.16	7.26	NS	14.2	15.5	NS	11.4	8.73	NS
x/m @ 5 mg P/100 gm	5.94	4.49	NS	12.0	10.5	NS	9.06	5.73	NS
CEC meq/100 gm	2.21*	6.15	NS	4.41	13.7	NS	5.68	12.1	NS
B max - kg P	NS	30.9	NS	6.79	70.4	1.92!!	5.31	49.3	NS
x/m @ 5 kg P	2.20*	27.4	NS	8.28	71.7	2.25	4.43	41.2	NS
CEC kg eq	NS	9.9	NS	7.09	22.1	NS	8.83	23.3	2.57
Deg. of Freedom	7/104	3/104	21/104	8/100	3/100	24/100	6/108	3/108	18/108

NOTE: NS = Not significant; values not footnoted are significant at .005
! = Significant at .01 * = Significant at .050
!! = Significant at .025 ** = Significant at .070

Table 7. F VALUES FOR COVARIANCE ANALYSIS WITH SOILS WITHIN SUBGROUPS AS REPLICATES WITH B.D., S+C AND pH AS COVARIATES

	Grouping I#			Grouping II			Grouping III			Grouping IV
	A	B	C	A	B	C	A	B	C	B##
B max	-	3.05!	NS	2.81!	7.85	NS	4.89	4.41	NS	12.61
x/m @ 5	-	2.77*!	NS	-	7.02	NS	-	4.10*	NS	11.18
CEC	-	2.88!	NS	NS	2.11!*	NS	3.64**	2.73*!	1.79!!	4.51

NOTE: If not noted, values are highly significant at the .005 level
NS not significant
* significant at .006 at 6/24 DF *! significant at .050
** significant at .010 !* significant at .100 at 8/22 DF
! significant at .025 # @ 7/23 DF
!! significant at .035 at 6/24 DF ## @ 5/25 DF

performed using the BMD04V program (18) for each of the 4 grouping systems. These analyses apply to unit capacities, not total mass capacities for the horizons. Either B max, x/m @ 5, or CEC was selected as the dependent variable with bulk density (B.D.), % silt plus clay (S+C) and pH as the independent covariants. Many of the F values were highly significant, especially for the B horizon. This indicated that the chemical properties are significantly different for different soil taxonomy subgroups. The mean values for Grouping III are given in Table 8 and the ranking of these means in Table 9. Here, several important trends were noted. The best soils for removing phosphorus were the acid tills. The acid tills were better than the calcareous tills, and the tills were better than the outwashes. Again, within the outwashes, the acid soils removed more phosphate than the calcareous soils. For CEC, the silt and clay beds

Table 8. GROUPING III MEANS

Sub Group	2	1	3	4	6	7	5	Ave.
Horizons				B max - mg P/100				
A	26.8	67.9	26.6	48.4	40.7	64.1	50.2	45.7
B	33.1	81.2	19.1	39.6	25.9	70.9	37.9	42.9
C	12.9	18.9	5.0	31.4	13.4	49.0	22.3	18.9
T	20.9	45.2	13.9	35.4	22.6	65.3	31.9	30.9
				x/m @ 5 - mg P/100 gm				
A	18.5	44.9	20.3	34.1	21.9	55.2	33.1	30.5
B	23.4	68.1	14.1	27.3	15.8	59.5	28.5	33.4
C	8.6	13.7	2.8	17.7	7.8	38.4	13.9	12.2
T	15.8	35.5	10.4	22.2	13.4	54.3	22.4	22.3
				CEC - meq/100 gm				
A	12.8	31.7	12.4	17.5	19.7	22.6	14.6	18.2
B	5.2	12.1	5.3	16.6	7.6	10.5	6.1	9.0
C	2.2	4.1	3.7	16.1	5.9	14.5	6.6	6.5
T	4.0	9.1	5.1	16.4	7.9	12.2	7.1	8.3
				B max - kg P/sq. m				
A	.048	.141	.077	.140	.099	.028	.083	.096
B	.351	.690	.301	.310	.416	.553	.541	.448
C	.212	.307	.047	.681	.210	.144	.306	.300
T	.602	1.202	.433	1.138	.745	.731	.908	.869
				kg P/sq. m				
A	.037	.086	.059	.099	.051	.024	.059	.064
B	.268	.575	.231	.212	.255	.464	.348	.338
C	.125	.218	.029	.371	.117	.113	.178	.180
T	.432	.932	.327	.693	.431	.607	.612	.601
				CEC - kg equivalent/sq. m				
A	.023	.063	.037	.052	.052	.010	.021	.039
B	.060	.103	.076	.086	.111	.082	.094	.087
C	.048	.062	.041	.377	.083	.042	.091	.111
T	.132	.250	.158	.562	.258	.137	.210	.251

*NOTE: All values reduced by a linear correction for % coarse fragments (CF).

Table 9. RANKING OF SOIL SUBGROUPS IN GROUPING III

Rank for # R	Rank for # CEC	Group	Parent Material	Comments*
1	3	1	Granite, gneiss till	Acid tills dominate in the New York Adirondacks S + C = 26.1%
2	4	5	Sandstone, siltstone and shale till	These soils often have high water table or rock limitations S + C = 45.4%
2	5	7	Shale and siltstone	Only one soil, "Gilpin," sampled has high rock limitations S + C = 62.9%
2	1	4	Silt and clay beds	Soils high in clay, impermeable with occasional high water tables S + C = 85.2%
3	2	6	Limestone, sandstone and shale till, calcareous	Calcareous tills, often in Western New York State S + C = 45.2%
4	7	2	Sand and gravel outwash	Outwash S + C = 12.5%
5	6	3	Sand and gravel outwash, calcareous	Calcareous outwash, often in Western New York State S + C = 13.8%

*Average % silt + clay for the total horizon samples.
#Rankings based on an average of ranking values for B or Total Horizons in Table 7. However, when the Group VII "Gilpin" soil came close to a tie in the rankings, it was put on the next lower rank, due to the shallow rock limitation.

had the highest capacity and the calcareous soils were better than the acid ones. The tills were again better than the outwashes. The complete data set and means for the weighted B horizon via Grouping IV are listed in Table 10. Here, the soils having highly spodic B horizons gave the highest phosphate adsorption. Correlations and regression equations were developed for the soils within the subgroups in another report (14).

The soil with the highest capacity to remove phosphate was the Becket series, which is an acid till, sampled in the New York State Adirondacks. Data for two sets of samples of this soils

Table 10. GROUP IV DATA SET FOR WEIGHTED B HORIZONS

		Coarse Fragments%	Silt and Clay%	Bulk Density	pH	B max	x/m @5	CEC
					Highly Spodic Horizon, Spodosols Subgroup I			
Hermon		11.20	8.13	1.53	7.20*	59.42	47.70	9.26
Colton		34.44	12.61	1.52	5.16	65.11	55.39	8.47
Bangor		23.33	23.09	1.59	5.20	80.22	65.47	6.54
Becket 2		17.14	18.62	1.42	4.97	154.34	135.82	19.39
Becket 1		23.75	29.85	1.47	5.31	147.01	130.17	17.92
	Mean	21.97	18.46	1.50	5.57	101.22	86.91	12.32
					Color B Horizon, Primarily Inceptisols, Subgroup II			
Adams		1.52	2.64	1.52	5.32	29.29	26.71	5.98
Colonie		.129	8.32	1.48	5.68	20.31	13.46	3.31
Elnora		.000	11.62	1.62	5.81	11.59	8.53	4.14
Windsor		8.57	15.00	1.70	6.40	23.62	9.42	2.38
	Mean	2.55	9.40	1.58	5.80	21.20	14.53	3.95
					Argillic Horizon, Alfisols, Subgroup III			
Howard		40.26	18.29	1.67	7.04	20.56	17.11	6.51
Palmyra		36.23	22.96	1.60	7.15	19.88	12.07	6.59
Ontario		11.74	42.61	1.61	7.23	19.06	9.34	8.09
Lansing		27.24	49.44	1.76	6.78	32.91	18.27	8.46
Honeoye		12.64	51.36	1.41	7.18	22.27	13.63	8.92
Rhinebeck		.87	77.13	1.63	6.87	35.36	21.55	8.77
Erie		16.96	53.33	1.72	7.54	34.80	21.77	7.45
Langford		20.22	53.51	1.45	5.64	50.45	40.19	6.72
Schoharie		2.28	86.69	1.69	6.20	37.41	27.27	21.20
Collamer		1.00	96.75	1.60	6.03	47.70	24.24	20.13
Vergennes		.000	99.00	1.52	6.21	57.28	47.78	26.50
	Mean	15.40	59.19	1.61	6.72	34.36	22.99	11.76
					Argillic Horizon, Ultisols, Subgroup IV			
Gilpin		18.62	63.10	1.46	4.20	70.91	59.52	10.52
					Cambic Horizon, Inceptisols, Subgroup V			
Chenango		70.78	13.32	1.47	6.16	16.76	13.27	2.89
Riverhead		12.08	22.42	1.60	5.09	33.10	24.72	2.29
Lackawanna		37.74	30.26	2.09	5.55	31.01	20.92	4.23
Sodus		25.13	33.93	1.90	6.33	20.70	16.16	4.99
Oquaga		36.24	35.36	1.34	4.86	37.96	32.75	7.25
Paxton		14.09	41.91	1.83	5.66	26.50	15.03	12.27
Morris		22.95	48.19	2.00	6.33	17.50	8.97	7.70
Elmwood		4.67	48.33	1.33	6.63	20.11	15.54	6.46
Lordstown		31.71	50.29	1.63	4.63	50.93	40.55	6.73
Mardin		30.80	51.92	1.73	5.25	39.40	27.60	4.33
	Mean	28.62	37.59	1.69	5.65	29.40	21.55	5.91
					Cambic - Spodic Intergrade Horizon, Subgroup VI			
Hinckley		29.70	19.39	1.44	6.16	31.53	25.59	9.81
Essex		16.67	20.33	1.64	6.80	48.73	36.42	8.71
Charlton		10.42	25.75	1.61	6.54	52.18	45.91	10.54
	Mean	18.93	21.82	1.56	6.50	44.15	35.97	9.69

*This value may be an error on the high side.

series (which is classified as a spodisol, typic fragiorthod, coarse loamy, mixed frigid) are given in Table 11.

The mineralogy data for the clay, silt and sand size fraction on the C horizon was obtained. The data on the sand fraction gave the highest correlations with B max and CEC. For all the soils, percents of quartz, potassium, feldspar and plagioclase feldspar, each had correlation coefficients of -.5 to -.7 with B max and CEC. The percents of leachable iron and aluminum and the percents of silt plus clay and mixed rock fragments were positively correlated with B max and CEC. However, when these C horizon soils were broken down into subgroups according to the Grouping III parent materials classification, the absolute values of correlation coefficients decreased with the exception of percent silt plus clay which increased to the .7 to .9 range. In other words, for soils of a broad diversity of parent materials, the minerals gave higher correlation with B max and CEC than for soils of a narrow group of parent materials.

It was noted from the detailed data on the soil horizons that, at constant percent silt plus clay, the soil B max decreased with increasing depth. This trend was further investigated by testing deep outwash and till samples from various locations across New York State. Fourteen depth composited samples ranging from 8-26 m (25-85 ft.) in depth gave an average coarse fragments corrected B max of about 6.7 mg P/100 gm and an x/m @ 5 value of about 3.3 mg P/100 gm, which was significantly lower than the average C horizon samples noted in Table 8 (14).

CONCLUSIONS

It was concluded that for soils up to a 1.8 m (6 ft.) depth in New York State and probably in the northeastern United States, that the most effective and useful way to rate their ability for phosphate adsorption capacity and cation exchange capacity was by grouping them by a parent materials soil classification system or by a B horizon taxonomy system and analyzing a number of soils from each soils subgroup. It was also found that the unit B max (phosphate adsorption capacity) of the B horizon was often over 2.5 tomes that of the C horizon. For phosphate adsorption, tills were better than outwashes and acid soils better than calcareous ones. The Becket series, an acid till in the New York State Adirondacks, was outstanding for phosphate removal. For CEC, however, the calcareous soils were slightly better than the acid ones, while the clays gave the highest total capacities. On the average, soils 8-26 m (25-85 ft.) had a lower B max than did the C horizon of soils at a 1-2 m (3-6 ft.) depth.

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Table 11. BECKET SOIL SERIES DATA

	Horizon - Becket 1					Horizon - Becket 2@				
	A_p	B 21	B 22	B 23	Cx	O & A	B 21	B 22	B 23	Cx
Thickness - cm	13	8	18	25	64	18	15	13	25	119
pH - units	5.1	5.0	5.3	5.3	5.9	4.4	4.91	4.95	5.0	5.2
B max - mg P/100 gm*	125.0	277.2	193.4	75.5	29.6	56.7	240	161.5	83.5	13.7
x/m @ 5 - mg P/100 gm*	104.1	221.2	180.1	67.9	27.0	6.3	214	135.7	88.8	10.0
x/m @ 5 - mg P/100 gm*	41.6	78.5	111.1	35.8	15.2	.7	109.2	55.6	45.4	3.0
CEC meq/100 gm*	26.5	33.7	19.8	11.8	6.5	76.7	35.5	16.9	10.9	1.7
Silt + clay % <.062 mm	43	43	34	23	24.5	21	21	17	18	16.5
Clay % <.002 mm	2	2	3	0	0.5	2	2	2	1.5	.5
Coarse fragments % >2 mm	12	12	17	32	22.5	15	20	15	16.5	21.5
Al by 2N HCl -%			1.91		0.89			2.87		.28
Fe by 2N HCl -%			1.79		1.17			2.63		1.12
Fe by B -% #			1.51		0.29			2.46		.13

*Corrected for % coarse fragments.
#Ammonium Citrate and Sodium dithionate leach.
@For the total B Horizon composite: TOC = 2.64%, volatile solids = 5.46%, Fe by B = 1.36%, Al = 1.01%.

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DESIGN CRITERIA

TWO PHASE EVALUATION OF LAND TREATMENT AS A WASTEWATER TREATMENT ALTERNATIVE - A RATIONAL APPROACH TO FEDERAL AND STATE PLANNING AND DESIGN REQUIREMENTS

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In order to include land treatment as a viable alternative in federal, state and local wastewater management plans and design efforts, a straightforward two-phase evaluation scheme was developed. This approach was found necessary to allow the consulting engineer to meet the dictates of both the Water Pollution Control Act Amendments of 1972, PL 92-500, and the Clean Water Act of 1977, PL 95-217, as well as budget adequate fees for the detailed and oft times expensive evaluation of the land treatment alternative.

Phase 1 involves a preliminary screening technique utilizing existing topographical data, agricultural data, soils maps, SCS Reports, climatological data and general design criteria. In this phase, potential sites, loading limitations and transmission routes are identified. Early impact identification and evaluation factors are used to highlight and optimize the engineering and scientific tasks and associated levels of effort which will be necessary in the Phase 2 detailed evaluation and study phase.

Phase 2, the second and more detailed phase,will determine the cost-effectiveness of the land treatment alternative(s), and quasi-final design criteria. The general breakdown of results from the second phase evaluation includes: wastewater composition and quantification, hydraulic loading, organic and nutrient loading, host site plant and soil renovation capacities (Phosphorus Adsorption Isotherms, Cation Exchange Capacity, etc.), geohydrological and groundwater flow analysis, pretreatment and storage requirements and effluent suitability for irrigation (salinity, electrical conductivity, sodium adsorption ratio, nutrients, heavy metals and toxic components).

A simple checklist has been developed for both the Phase 1 and 2 evaluation steps.

A. INTRODUCTION

The planning, design and construction of land application systems for the treatment of wastewater requires consideration of probably the broadest range of technical, environmental and socio-economic factors of all the alternatives in the wastewater treatment field today. The three basic land treatment methods: spray (slow rate) irrigation, overland flow and rapid infiltration can effect the subsurface and groundwater environments as well as surface and air environments. The public health, social and economic aspects of land treatment can also be quite complex thus emphasizing the importance of timely and accurate planning efforts prior to design and construction.

In addition, land treatment systems planned for the United States must comply with a number of federal, state and local regulatory requirements which necesitate extensive commitment of time and resources to the evaluation of potential land treatment sites and system components.

In order to insure both the timely and accurate evaluation of land treatment alternatives as well as maximize the direction and effectiveness of detailed site investigations, a two phase planning effort is proposed. The first phase would generally screen the study area for potential sites utilizing early evaluation indicators as "planning tools" to identify the most promising land treatment sites for further study as well as emphasizing the topics and data gathering efforts needed in the second phase to fully define the feasibility and potential environmental impacts of such sites.

B. FEDERAL AND STATE REGULATORY STATUS

The federal (USA) statutory basis for consideration and funding of land application systems for the treatment of municipal wastewater is the Federal Water Pollution Control Act Amendments of 1972 (Public Law 92-500). This is solidly reinforced by the recent passage of the Clean Water Act of 1977 (PL 95-217).

In accordance with the legislative dictates of Public Laws 92-500 and 95-217 a number of regulations have been issued pertaining to land application. They are:

- Areawide Waste Treatment Management (Section 208), 40 CFR 35, subpart F, published in the Federal Register, May 13, 1974.
- Grants for Construction of Treatment Words (Section 201).
- Guidance for Facilities Planning (May 1975), and 40 CFR 35.917.
- Cost-Effectiveness Analysis Guidelines, 40 CFR 35, Appendix A published in the Federal Register, September 10, 1973.
- Secondary Treatment Information (Section 304(d)(f)) 40 CFR 133 published in the Federal Register on August 17, 1973.
- Alternative Waste Management Techniques for Best Practicable Waste Treatment (Section 304(d)(2)), published in the Federal Register on February 11, 1976.

In addition to the existing federal statutes and regulations, the Administrator of the EPA, in his letter of October 3, 1977, has further defined EPA policy on land application. In this letter, he requires Regional Administrators to preferentially consider land application as an alternative wastewater management technology; ... and that if an applicant for construction grant funds does not recommend a method that encourages water reclamation and reuse, then the applicant should be required to provide complete justification for the rejection of land treatment.

Other related federal laws which effect the feasibility of the land application waste treatment alternative are:

- National Environmental Policy Act (PL 91-190)
- Safe Drinking Water Act (PL 93-532)
- Toxic Substances Control Act (PL 94-469)
- Resource Conservation and Recovery Act (PL 94-580)

The individual states and munnicipalities within the United States have also adopted a wide range of technical requirements in reviewing land treatment systems. In most cases, however, specifics of these requirements are not published and approval of land application systems is on a case by case basis.

C. THE FEDERAL WASTEWATER TREATMENT CONSTRUCTION GRANTS AND FACILITIES PLANNING PROCESS

The previously listed federal statutes, regulations and policy guildelines attributable to PL 92-500 and PL 95-217, emphasize both the requirements and overall need for accurate and timely evaluations of the land treatment alternative in meeting the wastewater treatment needs of a given community. In order to assist such communities in the planning, design and construction of such facilities, the U.S. Environmental Protection Agency, in cooperation with the individual states, has instituted a construction grants-in-aid program with three basic steps: Step 1, facility planning; Step 2, design; and Step 3, construction. A schematic flow diagram of this process is shown in Figure 1.

With respect to land treatment, the most important, yet complex stage in the process is the Step 1, facilities planning step. It is the basic purpose of the facilities plan to define the wastewater treatment needs and goals of a study area, and after developing alternatives, recommend the most cost-effective and environmentally sound means to meet those goals.

Recent federal guidelines have established the additional facilities plan-ing requirement to preferentially consider land treatment as wastewater treatment alternative as wlei as to thoroughly document any reasons for rejection of land treatment in the facilities plan and/or environmental assessment. The following sections outline a two phase pro-cess by which the environmental engineer or planner can effectively implement these basic facilities planning requirements, in a rational manner, to maximize the commitment of resources and insure the effectiveness of subsequent land treatment evaluation efforts.

PHASE I

1. Identify potential land applica- sites by general "process" suitability. Land treatment alternatives can be divided into two categories: "process" alternatives and "system" alternatives. Land treatment process alternatives refer to the basic process, i.e., spray irrigation, overland flow, etc. System alternatives, on the other hand, include all elements and unit processes of the land treatment facility, i.e., storage, transmission, discharge point, etc. Figure 2, illus-

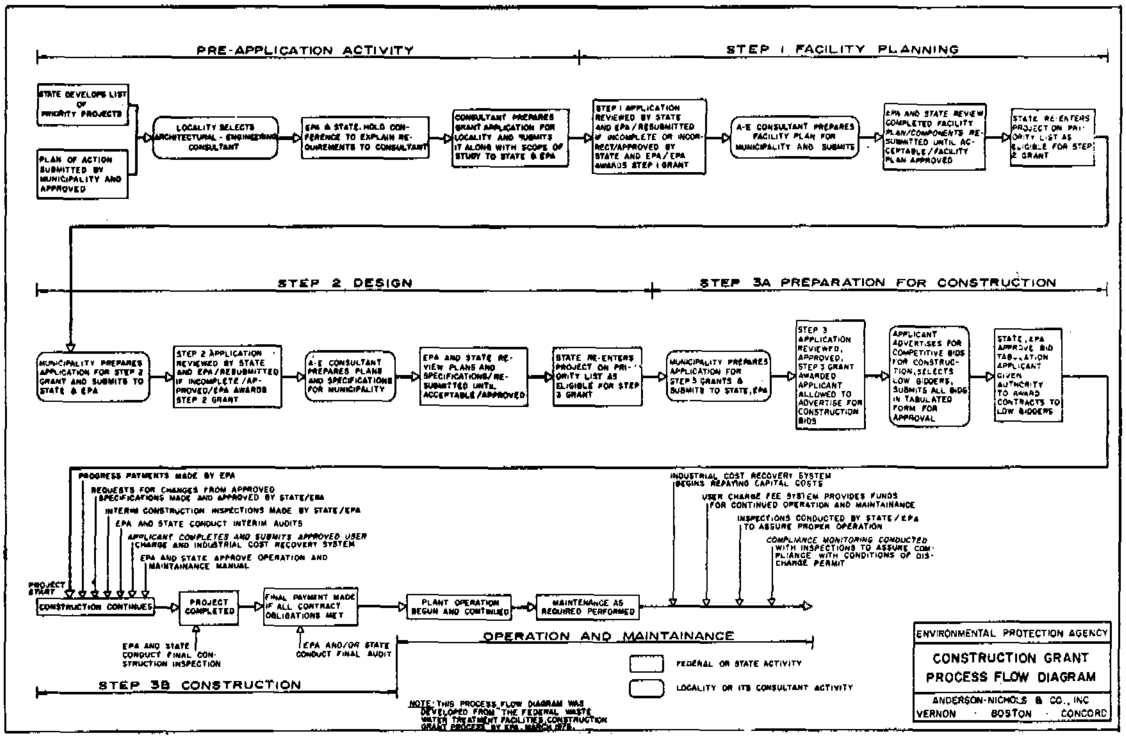

Figure 1. U.S. Environmental Protection Agency Construction Grant Process Flow Diagram.

D. TWO PHASE EVALUATION OF LAND TREATMENT AND CASE STUDY

In order to rationally develop land treatment alternatives for a given study area in an accurate yet timely manner, a two phase process can be used. Simply stated, the procedure starts with a broad, general and optimistic view of land treatment suitability and wastewater treatment needs in Phase 1 to develop early evaluation impact indicators of the various alternatives with the objective being to highlight and gradually focus in on the most promising land treatment alternatives for further study in Phase 2.

An outline of the two phase process is as follows:

trates schematically these various "process" and "system" alternatives. The Phase 1 site screening procedure will generally identify land areas within the study area as to their "suitability" (not feasibility) for the three major "process" alternatives:

- spray (slow rate) irrigation
- overland flow
- rapid infiltration

It is important to remember that the first screening of potential sites should utilize broad and general suitability criteria, with emphasis on identifying as many "potential" sites as possible, and leaving detailed judgements of each site for the latter steps of the Phase 1 procedure.

The key features of the Phase 1 site

identification procedure are shown in Fifure 3. The emphasis is on categorizing sites initially by process suitability i.e., spray irrigation, overland flow, etc., as well as their general suitability to meet water quality discharge requirements and land use constraints. The quantitative screening elements of the procedure, i.e., slopes, soils characteristics etc., will be dependent on continuing advancements in the state-of-the-art of land treatment as well as any unique features pertinent to specific study areas. The environmental planner, therefore, can incorporate more or less detailed site selection criteria within the basic screening methodology. A summary of published operational data on the three major land application processes is given in Table 1 and may be used as a baseline guide by the environmental planner in the initial site identification procedure.

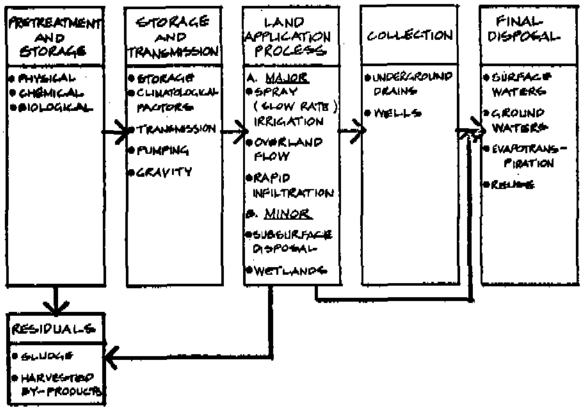

Figure 2. Land Treatment "Process" and "System" Alternatives.

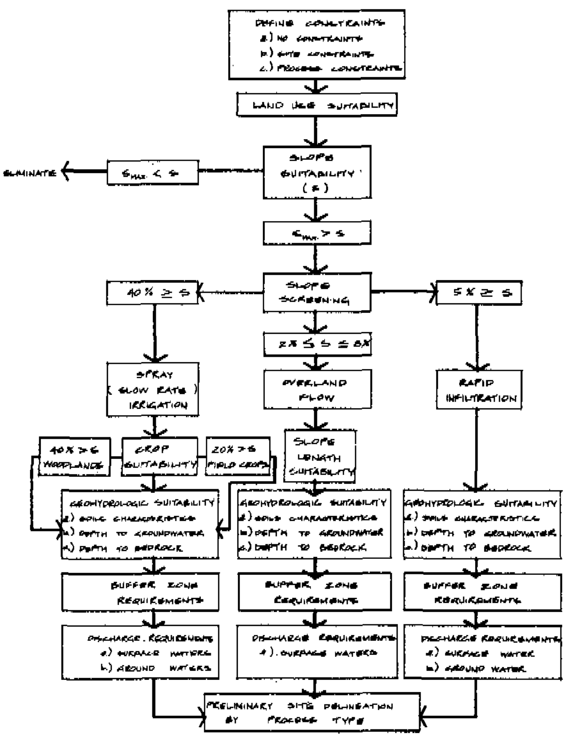
Figure 3. Flow Diagram for the Phase 1 Identification of Land Treatment Sites by Process Suitability.

Table 1. Summary of Published Site Operational Data for Land Application Processes (Source: Ref. 1,2,3,4).

Process Operational Data	Spray Irrigation (Slow Rate)	Overland Flow	Rapid Infiltration
I. Site Characteristics			
a) Soil Texture	clay loams to sandy loams	Clays and clay loams (1)	Sands and sandy loams
Permeability (in/hr)	0.06 - 20.0	0.2	2.0 - >20.0
Cation exchange capacity (meq/100g)	13 - 27	22 - 63	0 - 6.0
b) Topography Slope	0 - 20% cultivated 20 - 40% non-cultivated	2 - 8% (3)	<5% (2)
Relief	Varies	Varies	Varies
Flood Potential	Minimal	Minimal	Minimal
Vegetation	Field and forage Crops Woodlands	Perennial Grass	N/A
c) Geology Depth to Bedrock	5.0' min.	2.0' min.	15.0' min.
Depth to ground-water	2.0' min.	N/A	10.0' min.
d) Land Use Buffer zones	May conflict 200 - 1000'	May conflict 200 - 1000'	May conflict 200-1000'
II. Wastewater Loading			
a) Annual Application Rate (ft/yr)	2 - 20	10 - 70	20 - 560
b) Weekly application Rate (inches)	0.5 - 4.0	2.5 - 16.0	4.0 - 120.0
c) Storage (months)	3 - 8	3 - 8	Usually none

(1) Requires impervious strata at shallow depths
(2) Greater slopes are possible but require extensive earthwork.
(3) Overland flow slope lengths typically 120 to 150 feet long.

A description of the individual elements of the screening procedure shown in Figure 3 is given below:

a. Define constraints on the development of land treatment alternatives.

No constraints: all land within the study area can be evaluated for the entire range of land application processes and various system combinations.

Site constraints: Only predetermined sites can be used and land application processes are evaluated to match the given sites.

Process constraints: The study begins with a limitation on the land application processes which can be used and potential sites are identified only for those processes.

b. Land Use Suitability--Sites should not seriously conflict with existing land use but should reinforce land use patterns. The drainage basins of surface water supplies, national parks, heavily used recreational areas, developed urban areas and historic sites may be areas of exclusion for land application systems because of committed land use or sensitive environments.

c. Identify sites by process suitability using Table 1.

d. Define buffer zone requirements for potential land treatment sites. Typical values of from 200 to 400 feet can be used for the initial Phase 1 effort.

e. Define water quality standards and discharge limitations. The quality of land treatment effluents must logically meet the prevailing discharge standards for each receiving waterbody if water quality goals are to be met. Although detailed analyses are required to determine if a given land treatment site can produce an effluent in compliance with specific water quality standards and discharge limitations, the following preliminary screening criteria can be used to help categorize potential sites by process suitability.

- In general, all land application processes will produce an effluent of secondary quality suitable for discharge to major watercourses.
- Water quality limited watercourses of medium to small size are most probably suitable for direct or indirect discharges from spray (slow rate) irrigation and rapid infiltration systems.
- Small environmentally sensitive waterbodies including Class A streams, lakes and groundwater aquifiers most probably can receive effluents from spray (slow rate) irrigation systems which utilize harvested crops for maximum renovation of applied water.

It should be emphasized that the above general screening criteria is only a tool which may be of use during the preliminary site identification procedure and should not be the basis for final exclusion of any given site.

f. Define climatological constraints. In general, land treatment processes which depend on crop or plant uptake for wastewater renovation or which are adversely affected by freezing must apply wastewater on sites during the warmer periods of the year. These climatological factors in essence require storage of wastewater during certain periods of the year and so require higher wastewater loading on potential sites than that which would be dictated by the annual average flow rate.

g. Construct a composite site screening and evaluation map. The combined effects of land use, soils, topographic, geologic and hydrologic characteristics as well as water quality limitations of an area can be simply organized to determine the suitability of land application sites, on a preliminary basis, using three basic map overlays as tools:

- United States Geological Survey Quadrangle Maps
- Soil Conservation Service Soil Maps
- Land Use Maps (Zoning, planning agencies, etc.)

A grid system will normally prove useful in transferring data from the individual map overlays to the composite site screening map. The individual sites which are identified in the Phase 1 screening procedure are then transferred to the simple composite site selection map and evaluation for further analysis.

2. Preliminary Development of Wastewater Loading Rates

Wastewater quantity and quality must be accurately determined during the Phase 2 effort in order to fully assess the feasibility and environmental impacts of any of the land treatment sites which have been previously identified. However, a rough approximation of wastewater flow rates (i.e. 100 gpd/capita) and population projections (i.e. 50% increase during the planning period) can be used during the Phase 1 process to develop effective early evaluation indicators.

3. Preliminary Identification of Potential Land Treatment Impacts And Areas of Study

As was the case in initially identifying potential sites, the emphasis in the initial listing of impacts and study areas should be to include as many factors, both positive and negative as possible, leaving detailed judgements and evaluations for the latter phases of the planning effort. Table 2 can be used as a guide in initially identifying and developing these impacts and areas of study.

4. Early Evaluation Indicators for the Identification of the Most Promising Sites for Further Study

As outlined in the previous sections, the land application alternative can include an extensive list of options and areas for investigation. In order to insure that sufficient efforts are focused on significant land treatment impacts at an early stage in the facilities planning/ environmental assessment process, a systematic but flexible early evaluation system can be developed and used as a "tool" to determine the most promising land treatment sites and areas for future study.

The first step in the early evaluation procedure requires development of a single composite map of the planning area and adjacent communities with overlays of the following basic characteristics.

- Location of preliminary land application sites and suitability by process type(s)

Table 2. Potential Land Treatment Impacts and Areas of Study (Source: Ref. 3).

MAJOR ENVIRONMENTAL IMPACT COMPONENTS	DESCRIPTION
A. Environmental Effects	
1. Soil	Changes in groundwater levels, drainage areas, local climate, organic/inorganic effects on soil hydraulic capacity, soil chemistry and toxic elements
2. Vegetation	Toxic effects, changes in groundwater levels, local climates, reduced growth due to hydraulic loadings or poor aeration
3. Groundwater	Effect groundwater levels, rate and direction of flow, changes in quality and build up of toxic contaminants
4. Air Quality	Formation of Aerosols and odors
5. Animal and Insect Life	Disrupt food chain, migratory routes and habitats of certain species, alter groundwater tables, encroach sites of rare and endangered species
6. Climate	Increase local hudidity and decrease temperature, changes in microclimate near application site
7. Surface Water	Increases or decreases in rates of flow, changes in water quality, change in drainage basin hydrological characteristics
8. Geologic Formations	Transmission of contaminants through bedrock discontinuous, creation of discontinuous by percolating wastewater
B. Public Health Effects	
1. Groundwater	Build up of toxic contaminants, pathogens, heavy metals, reduction in quantity of potential drinking water supplies
2. Insects and Rodents	Contamination of insects (mosquitoes) and rodents from wastewater/sludge contaminants, breeding pathogenic vectors
3. Site Runoff	Contamination of populated areas
4. Aerosols and Odors	Transmission of pathogens in aerosols, odor effects on general health
5. Crops/Food Chain	Build up of toxic contaminants in the food chain via direct consumption or food chain intermediate (i.e. pigs, beef)
6. Noise and Traffic	Crop harvesting activities, primary and secondary growth effects, pretreatment facilities
7. Surface Water	Changes in quality and quantity of potential surface drinking water supplies
C. Social/Aesthetic Effects	
1. Land Use	Possible conflicts with surrounding land use: residential, commercial, industrial, recreational, urban, agricultural, wilderness and green belts
2. Community Growth	Increases in growth due to sewer system availability, decreases in growth due to resource or local services, committments
3. Relocation of Residents	Possible relocation of residential, commercial and farm buildings, schools, cemeteries and churches
4. Greenbelts/Open Space	Disruption of local scenic character, enhancement of local scenic character
5. Recreational Activities	Effects on wild and scenic rivers, archeological, historical and geological sites
D. Economic Effects	
1. Loss of Tax Revenues	Loss of taxable land as a result of governmental purchase
2. Land Devaluation	Change in value of land used or adjacent land
3. Energy Committment	Pretreatment facility, transmission of wastewater to application area, harvesting crops
4. Resource Committment	Land, chemicals, supplemental fertilization
5. Groundwater	Increase or decrease in quantity of groundater, change in quality of groundwater
6. Surface Water	Increase or decrease in quantity of surface water, change in quality of surface water
7. Revenues	Sales of crops, renovated water, irrigation water or leaseback of purchased land
E. Legal Effects	
1. Water Rights	Conflicts with Riparian, appropriative or combination water rights of natural watercourses, surface waters, percolating groundwater
2. Implementation Authority	Ability to purchase and use land application area, conflicting or overlapping political jurisdiction, public information and acceptance
3. Existing Regulations and Plans	Possible conflicts with comprehensive master plans, zoning, wastewater treatment regulations and standards

- Approximate boundary of major developed areas by land use.
- Location of surface and groundwater drinking water supplies and surface drainage divides.
- Approximate downstream center of the wastewater collection system and radii distances to potential sites.

Secondly, establish the preliminary wastewater quality and quantity projections for the study area. A reasonable percent increase in population (such as 50% can be assumed during the Phase 1 process for this initial assessment procedure. It should be emphasized that these initial population and wastewater generation projections are for preliminary evaluation procedures only and that detailed projections of wastewater quantities will be performed during the facilities planning effort. Major industrial or commercial wastewater loadings should be included even if extremely rough estimates are used.

Thirdly, estimate the approximate wastewater hydraulic loading on each potential site by determining a hydraulic loading factor (H.L.F.) for each site using equation 1.

H.L.F. = Wastewater Applied/Unit Average Daily Flow Equivalent Loading of the Application Site (1)

The individual components of Eq. 1 are described below.

Wastewater Applied

- Spray Irrigation
- Overland Flow
- Rapid Infiltration

= Annual Average Daily Flow Rate (gpd)

Unit Average Daily Flow Equivalent Loading of the Application Site

The equivalent hydraulic loading, in terms of the annual average daily flow, on the site area if wastewater was applied at 1 inch per week during the application period. The unit equivalent loading values for the major land treatment processes and assumed application periods are given below:

- Spray (slow rate) irrigation, 6 month application = 1940 gpd/acre
- Overland Flow, 6 month application period = 1940gpd/acre
- Rapid Infiltration, 12 month application period = 3880gpd/acre

Adjustments to the unit average daily flow equivalent loading values can be easily made to account for shorter application periods (i.e., specific crop growing seasons, state requirements etc.) by using the following relation:

$$\text{Unit Average Daily Flow Equivalent Loading} = A \times \frac{B}{26} \qquad (2)$$

where: A = Unit Average Daily Flow Equivalent for a 6 month (26 week) application period(i.e. 1940 gpd/acre)

B = Assumed application period in weeks

Once the H.L.F. is determined for each site table 3 can be used as an approximate guide in relating land treatment impact components to the H.L.F. The early impact identification and evaluation factors in Table 3 indicate increasing degree of severity and need for study of the various impacts on a scale from 1 to 3 with "+" or "-" for emphasis. Beneficial impacts such as crop revenue, irrigation water for golf courses, etc., are indicated with a value of 1.

It should be noted that the H.L.F. is essentially the application rate in inches per week which would be required on a specific site area to hydraulically dispose of the average daily wastewater flow from a given study area. A similar relation for sludge loading of a land application site can also be developed, perhaps in terms of a solids loading factor (S.L.F.) in which annual application of sludge in tons of solids per acre per year is calculated for each potential site and from which another site of early impact identification and evaluation factors can be assessed.

It should be emphasized again, that the early evaluation rating system is basically a subjective tool to optimize study efforts and not a substitute for detailed facilities planning analyses.

Table 3. Early Impact Identification and Evaluation Factors.

Major Environmental Impact Components	Spray Irrigation (Slow Rate)			Overland Flow			Rapid Infiltration		
	H.L.F. ≤1	H.L.F. 1-4	H.L.F. ≥4	H.L.F. <5	H.L.F. 5-10	H.L.F. >10	H.L.F. ≤20	H.L.F. 20-40	H.L.F >40
A. Environmental Effects									
1. Soil	1	2	3	1	1	2	2	2	3+
2. Vegetation	1	2	3	1	2	3	1	1	1
3. Groundwater	1	2	3	1	1	1	2	3	3
4. Air Quality	2	2	3	1	1	2	1	1	2
5. Animal and Insect Life	2	2	3	2	2	2	1	1	2
6. Climate	1	1	2	1	1	1	1	1	1
7. Surface Water	1	1	2	3	3	3+	1	1	1
8. Geologic Formations	2	2	3	1	1	1	2	2	3
B. Public Health Effects									
1. Groundwater	2	2	3	1	1	1	2	3	3+
2. Insects and Rodents	2	2	3	2	2	3	1	2	3
3. Site Run-off*	1	1	2	2	3	3+	1	1	1
4. Aerosols and Odors	2	3	3+	1	2	2	1	1	2
5. Crops/Food Chain	2	2	3	2	2	2	1	1	1
6. Noise And Traffic	2	2	3	2	2	3	2	2	3
7. Surface Water	1	2	3	3	3	3	2	2	2
C. Social/Aesthetic Effects									
1. Land Use	3	2	2	2	3	3	1	2	2
2. Community Growth	2	2	3	2	2	3	2	2	3
3. Relocation of Residents	2	2	2	2	2	2	1	1	2
4. Greenbelts/Open Space	2	2	2	2	2	2	2	2	2
5. Recreational Activities	2+	2+	3+	2+	2+	2+	1	1	2
D. Economic Effects									
1. Loss of Tax Revenues	2+	2+	2+	2	2	2	1	1	1
2. Land Devaluation	2	2	3	2	2	2	2	2	2
3. Energy Commitment	2	2	3	2	2	3	2	2	2
4. Resource Commitment	2	2	3	2	2	3	2	2	2
5. Groundwater	2	2	3	1	1	1	2	3	3
6. Surface Water	1	2	2	2	3	3	1	2	2
7. Revenues	1	1	1	1	1	1	2	2	2
E. Legal Effects									
1. Water Rights	3	3	3	2	3	3	3	3	3
2. Implementation Authority	2	2	3	2	3	3	2	3	3
3. Existing Regulation and Plans	2	2	2	2	2	2	2	2	2

* Rating values will vary with socio economic and site location factors. Environmental impact identification factors indicate increasing degree of adversity from 1 to 3 with + or - added for emphasis. Beneficial impacts are indicated with a value of 1.

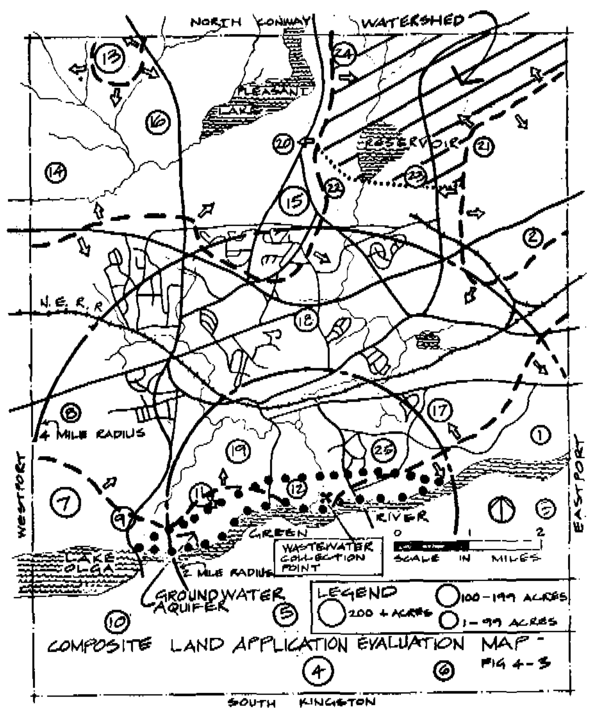

Figure 4. Phase 1 Composite Site Selection and Evaluation Map for Tinkersville, New England.

As an example of the Phase 1 procedures, a composite site selection and evaluation map for Tinkersville, a hypothetical community in New England has been developed and is shown in Figure 4. A rough projection of wastewater flows for the study area indicate that a value of 1.7 MGD for average daily flow can be used in the Phase 1 analysis.

A summary of early evaluation impact identification factors for this example are given in Table 4 and indicate, on a preliminary basis, that rapid infiltration sites 16 and 17 and spray irrigation site 7 are the most promising land application alternatives to be investigated in Phase 2. In order to fully evaluate sites 7 and 17, a comprehensive and detailed wastewater and soils renovation analysis will most likely be required to insure protection of the ground water aquifer to the south. A similar effort is indicated for site 16 to insure protection of the small Class A streams and future surface water supplies nearby. Energy demands and operational costs are also highlighted at site 16.

can be found in published literature (1,2, 3,4). Table 5, however, was developed to assist the environmental planner in establishing budget requirements and manpower estimates for both in Phase 1, preliminary development of land application alternatives and the subsequent Phase 2, detailed evaluations of specific sites. The range of values shown in Table 5 for the Phase 1 preliminary site screening efforts roughly corresponds to communities with sewered populations of less than 1000 to greater than 30,000. In addition, the range of manpower effort for the Phase 2, detailed site investigations roughly correspond to specific land application sites of less than 50 acres (20.3 ha) utilizing land treatment processes without storage to land application sites of 500 acres (203 ha), utilizing processes that do require wastewater storage. It should be noted that any given site chosen for Phase 2 evaluation

Table 4. Summary of Environmental Impact Early Warning Indicators for Tinkersville.

					ENVIRONMENTAL IMPACT COMPONENTS				
			SI		(A) Environmental	(B) Public Health	(C) Social Aesthetic	(D) Economic Effects	(E) Legal
Site I.D.	Acres	Suitability	OF	RI	1 2 3 4 5 6 7 8	1 2 3 4 5 6 7	1 2 3 4 5	1 2 3 4 5 6 7	1 2 3
1	70	SI	12.5		3+ 3+ 3 3 2 2 2+ 3	3 3 2+ 3 3 2 3+	2 3+ 2 2 2	2 2 3+ 2+ 2 3+ 1	3+ 3+ 2
2	65	SI	13.5		3+ 3+ 3 3 2 2 2+ 3	3 3 2+ 3 3 2 3+	2 3+ 2 2 2	2 2 3+ 2+ 2 3+ 1	3+ 3+ 2
3	50	SI	17.5		3+ 3+ 3+ 3 2 2 2+ 3	3 3 2+ 3 3 2 3	2 3+ 2 2 2	2 2 3+ 2+ 2 3+ 1	3+ 3+ 2
4	200	SI	4.4		2+ 2+ 3 3 2 2 2+ 2+	3 2 2+ 2+ 2+ 2 3	2 2 2 1 2+	2+ 2 3+ 2+ 2 3+ 1	3+ 3+ 2
5	120	OF	7.3		2 2 1 1 2 1 3+ 1	1 2 3+ 2 2 2 3+	2 2 2 2 2+	2+ 2 3+ 2+ 1 3+ 1	3 3+ 2
6	60	SI	14.6		3+ 3+ 3 3 2 2 2+ 3	3 3 2+ 3 3 2 3+	2 3+ 2 2 2	2 2 3+ 2+ 2 3+ 1	3+ 3+ 2
7	312	SI	2.8		2 2 2 2 2 1 1+ 2	2 2 1 3 2 2 2	2+ 2 2+ 2 2+	3 2+ 3+ 2 2 2+ 1	3 2+ 2
8	90	OF	9.7		2 2 1 1 2 1 3 1	1 2 3+ 2 2 2 3+	2+ 2 2 2 2+	2+ 2 3+ 2+ 1 3+ 1	3 3 2
9	100	SI	[illegible].7		3+ 3+ 3 3 2 2 2+ 3	3 3 2+ 3 3 2 3+	2 3+ 2 2 2	2 2 3+ 2+ 2 3+ 1	3+ 3+ 2
10	80	OF	11.0		2 3 1+ 2+ 2 1+ 3+ 1+	2 3+ 3++ 3 2 3 3+	3 3 2 2+ 2+	2 3 3+ 3 1+ 3++ 1	3 3 2
11	110	OF	8.0		2 3 1 2+ 2 1+ 3++ 1+	2 3+ 3++ 3 2 3 3++	3+ 3+ 2+ 2+ 2+	2 3 3+ 3 1+ 3++ 1	3 3+ 2
12	90	OF	9.7		2 3 1 2+ 2 1+ 3++ 1+	2 3+ 3++ 3 2 3 3++	3+ 3+ 2+ 2+ 2+	2 3 3 3 1 3 1	3 3+ 2
13	220	SI	3.9		2+ 2+ 3 3 2 2 2+ 2+	3 2 2+ 2+ 2+ 2 3	2 2 2 1 2+	2+ 2 3+ 2+ 2 3+ 1	3 3 2
14	70	SI	12.5		3+ 3+ 3 3 2 2 2+ 3	3 3 2+ 3 3 2 3+	2+ 3+ 2 2 2	2 2 3+ 2+ 2 3+ 1	3 2 2
15	275	SI	3.2		2+ 2+ 2+ 2+ 2+ 1+ 1+ 2	2+ 3 2 3+ 2+ 2 2+	3+ 3 3+ 2 2+	3 3+ 3 2 2+ 2+ 1	3 3++ 3
16	150	R.I.		2.9	2 1 2 1 1 1 2 2	2 1 1 1 1 2 2	2 2 1 1 2	2+ 2+ 3+ 2 2+ 1+ 2	3 2+ 2+
17	160	R.I.		2.7	2 1 2 1 1 1 2 2	2 1 1 1 1 2 2	2 2 1 1 2	2+ 2+ 2 2 2+ 1+ 2	3 2+ 2+
18	140	OF	6.3		1+ 2 1 2 2 1 3 1	1 2+ 3+ 2+ 2 2 3+	3+ 2 3 2 2+	3+ 3 1 2 1 3 1	3+ 3++ 2+
19	120	SI	7.3		3+ 3+ 3+ 3+ 3+ 2+ 2+ 3+	3+ 3+ 2+ 3+ 3+ 3+ 3+	3 3 3+ 2 2+	3+ 3+ 2 2 3+ 2+ 1	2 3+ 3
20	80	SI	10.9		3+ 3+ 3+ 3+ 3+ 2+ 2+ 3+	3+ 3+ 2+ 3+ 3+ 3+ 3+	3 3 3+ 2 2+	3 3 2 2 3 2 1	3 3 3
21	65	SI	13.5		3 3 3 3 2+ 2 2+ 3	3 3 2+ 3 3 2 3+	2 3+ 2 2 2	2 2 3+ 2+ 2 3+ 1	2 3+ 2+
22	40	SI	22		3+ 3+ 3+ 3 2+ 2+ 2+ 3	3 3 3 3 3 3 3+	3 3 3 3 3	2+ 3 2+ 2+ 3 3+ 1+	3 3+ 2+
23	150	RI		2.9	2 1 3 1 1 1 2 2	2 1 1 1 1 2 2	2 2 1 1 2	2 2 3 2 2 1 2	3 3 3
24	160	SI	5.5		3 3 3 3 3 2 2 3	3 3 2 3 3 3 3	3 3 2+ 2+ 3	3 3 3 3 3 2 1	3 3+ 2+
25	120	SI	7.3		3 3 3+ 3+ 2 2+ 2+ 3+	3+ 3 2+ 3+ 3 3 2+	3 3 3 3 3	3 3 2+ 3 3+ 2 1	3 3+ 2+
PRELIMINARY MAJOR IMPACTS					* * * * *	* * *	* *	* *	* *

* SI – Spray Irrigation
OF – Overland Flow
RI – Rapid Infiltration

Phase 2

Using the results of the Phase 1 analysis, the environmental planner can now itemize potential land treatment sites for detailed study in Phase 2 as well as the areas in which maximum study efforts should be focused. Specific information on each potential area of study may not require all the tasks indicated in Table 5 and conversely other sites within a given study area may require additional study efforts. Also, intermediate levels of effort between the Phase 1 and 2 categories may further optimize the planning process. For example deep soil borings or a preliminary cost effectiveness analysis can be performed prior to a full commitment to a Phase 2 detailed site study. In any case the environmental planner must include sufficient levels of effort in evaluating the selected land application alternatives in Phase 2 to adequately define both the technical and environmental feasibility of the land treatment alternative under study as well as its viability as a practical wastewater treatment method.

Credit: Much of the material in this paper was developed by Anderson-Nichols & Co. under contract (68-01-4617) with Region I of the Environmental Protection Agency for the preparation of an Environmental Assessment Manual designed to improve the assessment procedures being used in wastewater facility planning.

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Table 5. Consultant and Subcontractor Manpower Commitment for the Evaluation of the Land Application Alternative.

ITEM	FACILITIES PLANNING BUDGET	ENVIRONMENTAL ASSESSMENT BUDGET
Phase 1, Preliminary Site Screening and Development	10-20 man days	5-10 man days
*Phase 2, Detailed Site Evaluations		
Surveys		
Borings		
Cadastral		
Quantify Wastewater Characteristics		
Soils Mapping	230-1400 man days	30-110 man days
Groundwater Mapping		
Bedrock Mapping		
Soils Renovation Capacity		
Soils Hydraulic Capacity		
Report		
Public Hearing		

*Includes an approximate range of manpower effort to completely evaluate a single site

DESIGN CRITERIA

HISTORICAL PERFORMANCE OF DESIGN DEPARTURES OF MUSKEGON COUNTY WASTEWATER MANAGEMENT SYSTEM NO. 1 AS VIEWED BY THE DESIGN ENGINEER

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ABSTRACT

A number of departures from conventional design were used in the Muskegon County Wastewater Management System No. 1, primarily to save nearly $20 million in construction cost. This paper summarizes several of these design departures:

1. Elimination of grubbing in the areas cleared of trees for agricultural purposes.
2. Elimination of most of the lining of the sand-bottom lagoon in favor of intercepting and returning the leakage.
3. Soil-cement slope paving 8" thick.
4. Prefabricated pumping stations up to 28 MGD capacity.
5. Automatic machines for laying the drainage pipes.
6. Sand ditches for major water conductors.
7. Percolation through soils underlying the storage lagoon as a treatment process.
8. Storage lagoons as biological treatment works.

INTRODUCTION

This project in Muskegon County, Michigan, was designed under the direction of the writer during the period July of 1970 to May of 1971, when bids were taken. Construction began in the fall of 1971; the first wastewater arrived in May of 1973. At the time of this writing, parts of the system had been in operation for about 5 years.

This land treatment system proved to be remarkably low in construction cost. Comparisons are made with estimating curves which have been provided by the U.S. EPA for guidance of consulting engineers. Part of the reason for the large difference in cost lies in the use of departures from conventional design for many of the features of the project, thereby incurring some risk. If the nonconventional design does not work, or works poorly it reflects upon the consulting engineer in a detrimental manner. Because the saving in construction cost does not accrue to the benefit of the consulting engineer, it would appear foolish for him to subject himself to such risks. For better or for worse, however, this was what was done in the Muskegon County project.

For this reason, there is an unusual opportunity to evaluate the performance of design departures which are seldom attempted on the scale at which they were attempted in the Muskegon project. As the person most aware of the risks and benefits which are involved in the use of these design departures, it is appropriate for the design engineer to take a look at the actual performance after the first five years. This paper reports the results of that second look.

DESIGN DEPARTURES

Definitions

The departures from conventional design which were used in the Muskegon project and which are discussed here may not appear to be departures to some design engineers. They may point to other projects in which the same ideas have been used successfully. Nevertheless, the writer believes that the ones he selected for discussion in this paper would be perceived as being departures from the conventional design by most engineers.

List of Departures

The list of departures which have been selected for discussion appears in the Abstract of this paper and will not be repeated here. Suffice it to say that not all of the design departures have been listed, but only those which the writer considered would be of major interest to prospective designers of land treatment systems.

DISCUSSION

Elimination of Grubbing

Risks. Much of the area was covered with trees at the time the project site was selected. Certainly the trees would be cleared off, but was it necessary to clear out the roots as well? It was decided to eliminate grubbing, reasoning that the roots would decay in place as a result of the irrigating and fertilizing. Damage to farming equipment occurred during plowing or discing, but such difficulties were small compared to the cost saving.

Damage to underground electrical cables also occurred. Proper procedure calls for an initial run of the plow without the electrical cable to discover and clear away underground obstacles. In a second run, the plow is drawn along the same course, this time laying the cable. Unfortunately, the electrical contractor elected to attempt the installation with only one run of the plow per course.

Cost Savings. The actual cost of the clearing of 3850 acres of trees was bid at $1.5 million if burning of the cleared trees would be permitted, and at $1.9 million if burning would not be permitted and the trees had to be stacked in the "corners" between irrigation circles. The final cost was about $1.8 million. This included some grubbing in critical areas - such as the sites of the dikes for the storage lagoon - but did not include grubbing of roots throughout the major portion of the area. The specifications provided for pushing the trees over so that the roots would be pried out with the trunk. This was done in most instances. It was not done in the case of the smaller trees, which did not have enough stiffness and strength in the trunk to pry out the roots. The actual cost of tree clearing averaged $467 per acre.

By comparison, the costs for clearing of 3850 acres of land as read from the estimating charts provided by the U.S. EPA are as follows:

	$ millions, EPA	
	Chart	May, 1971
Heavily Wooded	6.0	5.3
Brush, Trees	1.4	1.2

The Muskegon site would call for taking the average of the two figures. The cost from the estimating charts is seen to be on the order of $3.2 mil. This may be compared to the actual cost of $1.8 million. The largest factor in accounting for this difference was the elimination of the grubbing requirement. The saving of $1.4 million was far greater than the additional expenses incurred in the repair of the agricultural discs caused by collision with roots and other obstacles encountered in cultivation.

Elimination of Lagoon Lining

Risks. The large storage lagoons of the Muskegon project were constructed entirely on sand and the dikes were comprised entirely of sand. The sand had been tested for its thickness and permeability by means of 4 different well pumping tests. It was apparent that considerable leakage would occur unless steps were taken to curtail it. Large leakage could also threaten the safety of the dikes, which might be destroyed by piping of water through or beneath them. For both reasons, it was necessary to control the leakage.

Conventional Approach. The conventional approach would be to line the entire bottom of the lagoon. This approach is indicated in the US EPA estimating charts, which show the cost of such lining for lagoons with 5000 MG of storage (Muskegon size) to be $15 million. Now the total cost of the Muskegon project leakage control system, was less than $1 million. Why the large difference?

Design Departure. The design departure used in this instance was to control the leakage to a tolerable amount, then intercept it and return it to the reservoir. The amount of the leakage was limited by the use of a clay lining 400 ft. in width around the perimeter of the lagoon, the lining thus covering perhaps 20% of the total area. The total bid cost for 1.74 million sq. yds. of clay blanket was $834,000. This is a far cry from the $15 million one would read from the US EPA cost estimating chart. Even if one were to use the fact that only 20% of the bottom would be covered with clay, the US EPA chart would lead to an estimate of $3 million, still several times the actual cost. Two reasons are given for the difference:

a. The Muskegon approach involved the lining of only 20% of the total lagoon bottom.

b. The unit costs used in the US EPA publication were for a lining material much more expensive than clay.

Even when one adds the cost of the perimeter ditch around the lagoons for the intercepting of the leakage and the cost of the two pumping stations for the return of the intercepted flow to the lagoons, the result is to add only about $100,000 more to the cost. Thus, the total cost for 100% management of leakage through the sand bottom of the 1700 acres of storage lagoon was less than $1 million. As compared to the use of a 100% lining using expensive materials - which the US EPA estimating chart says would cost $15 million - this is a substantial saving. Furthermore, there is no guarantee that the 100% lining would actually control all of the leakage, whereas one can be reasonably confident that all of the leakage is con trolled by the interception and recycling system actually used.

Soil Cement Slope Protection

A thin, 8" thick, paving of soil cement was used on the 4:1 interior slope of the sand dikes to serve as wave protection and as a means of limiting seepage. This was a deliberate departure from the conventional type of soil-cement wave protection such as that developed by the Bureau of Reclamation, which uses stair-stepped layers of soil cement, resulting in an effective thickness of 2 or 3 feet. The 8" thickness was selected strictly as a means of reducing the cost of the 635,000 square yards of such protection required.

The cost saving - as compared to the use of say 24" of soil cement or the use of Portland cement concrete paving - was on the order of $6 per square yard, or about $3.8 million. Some ice damage during the first winter of operation, when the water was unusually high, produced some anxiety on the part of many that the choice was a good one. Subsequent experience has shown the cost of maintenance to be relatively small compared to the saving in original construction cost. Actual expenditures to date - for the years 1974, 1975, 1976 and 1977 have not exceeded an average of $50,000 per year. This is about 3% of the original cost of the installation. The interest on the difference in construction cost would have been about $200,000 per year, let alone the saving in the original investment itself, plus the saving in the cost of maintaining the alternative system.

Prefabricated Pumping Stations

Seven prefabricated pumping stations of the subterranean type were used in the Muskegon County Project. The largest of these had a capacity of 28 MGD, and the smallest a capacity of about 1.5 MGD. These stations were part of the sewage collection system. The combined cost of these seven stations installed including wet wells and electrical power substations was $1.3 million. The largest one (28 MGD) was bid at $255,000. The next largest (20 MGD) was bid at $225,000. The smallest (1.5 MGD) was bid at $81,000. The equation $C = 80,00 + 6250\ (Q)$, where C = cost in dollars and Q = station discharge capacity in MGD, would appear to fit the data, (May 1971 price levels).

On the basis of this equation, the main pumping (P.S. "C") with capacity of 80 MGD would have cost $580,000 as a prefabricated subterranean station. Its actual cost - constructed as an above-ground conventional reinforced-concrete-and-masonry structure with office and workshop - was $1.7 million. Even when allowances were made for such features as the office, the workshop and the surge tank, the cost was at least twice what would be expected for a prefabricated pumping station.

Thus one could conclude that the $1.3 million construction contract for the 7 smaller pumping stations would have been at least $2.6 million if constructed-in-place station had been used.

To-date, the principal problem of the 28 MGD prefabricated subterranean station has been the dissipation of the heat generated by the electric motors. An air conditioner was provided in the station, but proved to be inadequate and it was augmented by the addition of another unit.

Automatic Drainage Pipe Laying

The deep permeable sands of the project site permitted spacings of drains up to 500 ft. Such large spacing was a direct result of calculations based upon measured transmissibility of the sand formation, the hydraulic loading on the drainage system and the hydraulic capacity of the 6" diameter corrugated plastic drainage pipes.

Furthermore, the drains were originally bid at $2.87 million, compared to an Engineer's Estimate of $0.6 million. Through field demonstrations arranged by the design engineers, a contractor was pursuaded to do the same job for less than $0.8 million. The cost saving can be attributed to the use of automatic, laser guided, computer-controlled machines which accurately placed 6" drain pipe at depths between 5 and 9 feet at speeds of 1500 feet per hour.

Drainage problems of the site during the first 5 years of operation are confined to the areas of muck soils, which are found in the south 1/4 of the irrigated area of the main project site. The design here was predicated on the assumption that the drain pipes would be laid below the muck soils in the underlying sand, and that the percolating water would move down through the muck and then horizontally through the sand to the drains. Evidently this mode of operation did not develop as planned and the operators are coping with the problem by using less irrigation water and by discing to air dry for planting.

Deep Ditches vs. Large Conduits

The final product of the system is delivered to the receiving streams via large and deep ditches excavated in sand. In order to minimize the number of pumping stations, these ditches were sometimes made very deep to pass through some high ground. One might raise the question of why not concrete conduits? These could be passed through high ground in deep trenches and covered up to create a more attractive landscape. Furthermore, the quality of the underdrain water - free from all suspended solids and bacteria - could be maintained in closed conduits. In the open ditches, this water is reinfected with bacteria from the atmosphere and surrounding ground and also picks up a new load of suspended solids and nutrients. (Published statistics on the final effluent quality show the 4 or 5 mg/l of suspended solids picked up in the flow through the sand ditches and the corresponding amounts of nutrients and bacteria. The water coming out of the underdrains is essentially free from such contaminants.)

There are several other types of large ditches which are also used throughout the site. The following tabulation indicates what was involved:

Type of Ditch	Length
Outlet, Mosquito Creek	7 miles
Outlet, Black Creek	1 mile
Lagoons, ditches	7 miles
Bypass ditches	5 miles
Total	20 miles

If these ditches had been replaced by concrete pipes, for example, the cost increase would have been extremely large. On the flat grades available, it would have been necessary to use very small velocities and hence large pipes.

With a typical slope of 1 foot in 10,000 feet, the capacity of a 48" diameter pipe would have been only 13 cfs. A 96" diameter pipe would have a capacity of 90 cfs, approximately what would be required for the main outlet to Mosquito Creek. Taking an average size of 72" for this pipe, the required 7 miles of it would have cost about $2.5 million. By contrast, the total cost for 13 miles of ditches, including incidental concrete structures and associated pumping stations, was $1.1 million. The saving in cost through the use of ditches was probably at least $250,000 per mile, or a total of say $5 million for the project as a whole.

What was lost by the use of ditches rather than concrete pipes? One could list the following:

1. Large areas of land were taken up for ditches. If pipes had been used, this land surface would have been available for other purposes.

2. The final effluent of the underdrain system - free from suspended solids- bacteria and nutrients, was "recontaminated" as it passed through the open ditches on its way to the receiving stream. (The "recontamination" was the result of "natural" causes, and hence would not by some be classed as "contamination". Nevertheless, it showed up as increased suspended solids, bacteria and phosphorus.)

It was judged by the writer at the time of the design and it is still his opinion, that the added advantages were not worth the several millions of dollars of additional cost.

Percolation Through Lagoon Bottoms as Treatment

At the time of the design of the Muskegon County system it was recognized that the controlled leaking through the bottom of the lagoon could constitute a treatment process. The distance of percolation through the sand to the intercepting ditches was a minimum of 500 feet, 400 feet of this produced by the clay blanket around the perimeter of the dikes.

Nevertheless, the Michigan authorities would not permit any credit to be taken in the design of the project for the effect of this percolation. It was to be measured, however, so that future designers might be able to take advantage of it.

Some rough calculations were made of the amount of iron in the mass of sand which the percolating water would encounter, in an attempt to make a crude estimate of how long the phosphorus precipitation might continue. The lagoons began the leaking process in 1973, and so there has elapsed a period of nearly 5 years during which the quality of the intercepted water has been sampled and analyzed. What have been the results?

The first water to be intercepted was of course the ground water under the lagoon site. This water was forced out by the leakage water entering the sand mass under the lagoons. With a typical percolation velocity of 5 feet per day under the 0.1 hydraulic gradient, and with a 10% available volume for movement, the actual progress made was about 50 feet per day. The 500 feet distance would then be tranversed in about 10 days. Thus, very soon after the leakage began, the first percolated water would be showing up in the intercepting ditches. After the first month, it could be reasoned that most of the intercepted water had leaked from the lagoon.

Yet for nearly five years (May 1973 to April 1978) the effect of this anaerobic percolation through iron-bearing sands has been to reduce the total P from about 2 to about 0.02 mg/l, a 99% removal. In the ditch, the P content goes up to 0.1 mg/l because of the biological community in the ditch. Even so, the performance has been remarkably consistent and reliable.

Biological Treatment in Storage Lagoons

The original design of the Muskegon County system assumed a BOD loading on the storage lagoons of 20 pounds of oxygen demand per acre per day. For the 1700 acres, this amounted to 34,000 pounds of oxygen demand to be satisfied each day by the natural reaeration which would occur in these large lagoons.

The entire system was designed for 42 MGD of water with an assumed incoming BOD of 250 mg/l, which amounted to a design BOD load of 88,000 pounds per day. Of this total, 54,000 was to be satisfied by the mechanically aerated lagoons and the remaining 34,000 by the naturally aerated storage lagoons.

To date, the system has not been fully loaded. Currently, the load is about 28 MGD with a BOD of typically 250 mg/l. This amounts to about 58,000 pounds of oxygen demand per day. Assigning 34,000 to the storage lagoons leaves 24,000 for the aerated lagoons. Thus, these aerated lagoons would be loaded to about 24,000 ÷ 54,000 = 44% of their design capacity. This light loading was not realized by those who first operated the system, and they used all of the mechanical aeration capacity even though it was not needed. When it was realized that this practice was wasteful of electric energy, it was stopped and a fraction of the equipment was used. At present, about 1/3 of the capacity is being utilized, with the remainder of the BOD being satisfied by the storage lagoons.

To date, there has been a negligible accumulation of solids on the bottom of the storage lagoons. (The deposition in the aerated lagoons is also very small.)

While the use of naturally aerated lagoons to satisfy the BOD of sewage is certainly nothing new, the use of lagoons of 850 acres in extent for this purpose was new. To achieve a satisfaction of 34,000 pounds of oxygen demand per day without using one kilowatt hour of electricity is an important achievement in this day of energy conservation. It deserves wider publicity.

One could conceive of large sewage treatment system which would provide sufficient area in the storage lagoons so that 100% of the biological treatment would be achieved without the use of electrical energy. In the case of the Muskegon lagoons, it is apparent that a flow of about 16 MGD could be handled without any electric energy. This is a large flow. The investment in the storage lagoons themselves is about $8 million. The leakage out the bottom is on the order of 16 to 20 MGD, and this is water filtered through a minimum of 500 feet of sand. It thus appears that the $8 million storage lagoon could provide for 16 MGD with the following treatment.

(a) Flow regulation; the ability to receive shock loads up to many times 16 MGD without adverse impact.

(b) Biological treatment; the natural aeration processes which would occur without the use of electrical energy would be sufficient in themselves to satisfy the oxygen demand.

(c) Final filtration through a minimum of 500 feet of sand, removing suspended solids and bacteria.

(d) Possibly phosphorus removal for a considerable period of time, by virtue of the ability of the chemicals in the underlying sand to tie up the phosphorus in the percolating water.

(e) A system without sludge problems.

Examination of the high quality of the water in the intercepting ditch around the storage lagoons is evidence of the performance of a system of the type outlined above. One wonders whether this type of treatment might indeed be an economical solution in many locations where soils of the same type would exist, these being mainly dune sand.

There are other locations in the world where similar types of sewage treatment are utilized. One of these is in Holland, where again the dune sands are used. Another is in Phoenix, Arizona, in the experimental work of Herman Bouwer. The soils there are in two parts, the upper layer of 5 feet or so being a sandy loam, and the lower one of 200 feet or so being a gravel.

Data from the performance of the Muskegon County System should be made available through the auspices the State of Michigan Department of Natural Resources, so that future economies in construction and operation of sewage treatment plants can be achieved.

The effect of the inherent treatment ability of the storage lagoons is to increase the total capacity of the Muskegon County System by at least 16 MGD. This brings the capacity of the main site to about 58 MGD, which may be compared to the present load of 28 MGD. There is evidently considerable ability for expansion of industry and related development in the Muskegon County area without increasing the need for additional sewage treatment capacity.

SUMMARY

Savings in construction cost of the Muskegon County project which may be attributed to departures from conventional design are as follows:

No grubbing	$3.2 million
Limited lining	$4.0 million
Thin soil cement	$3.8 million
Prefab Pump Station	$1.3 million
Drain Pipe Laying	$2.0 million
Deep Outlet Ditches	$5.0 million
Total	$19.3 million

This saving in construction cost largely accounts for the difference between an original estimate by an independent consultant of $70 million and the actual cost of about $45 million, including land and interest during construction. (The remaining $5 million saving results from the absence of a general contractor; Bauer Engineering coordinated the several independent contracts.)

In addition, the data provided by the Muskegon project on the use of large lagoons on sand as treatment works should save future construction and operating dollars for projects which would be constructed in similar geological conditions. Whether this saving will be achieved depends upon the initiative of designers in taking adventage of such opportunities.

DEVELOPMENT OF LAND APPLICATION DESIGN CRITERIA FROM FIELD INVESTIGATIONS

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ABSTRACT

A recently completed feasibility study developed land application design criteria for Fort Polk, Louisiana. The initial site assessment provided information about general site conditions, groundwater, soils, climate, and special conditions. Detailed site investigations were conducted to provide specific soil, geologic, and hydraulic conductivity information. Application rates and storage requirements were developed for rapid infiltration, slow rate, and overland flow systems. The best apparent alternative was found to be a rapid infiltration system.

INTRODUCTION

The engineering design of a land treatment system relies upon the application of a general procedure to site-specific conditions. Fort Polk, Louisiana, is presented as an example of those methods.

EXISTING SITE CONDITIONS

General. Fort Polk is located in the west central portion of Louisiana near the communities of Leesville and DeRidder. The approximate distances to principal cities in Louisiana are Shreveport, 177 km; Alexandria, 97 km; and Lake Charles, 145 km.

The existing land use is an impact area for the small arms firing ranges. Nearly mature longleaf pine provide the predominant vegetative cover. The rolling hills vary in elevation from 85 m to 110 m.

Groundwater. Groundwater occurs regionally in both artesian and perched water table aquifers. In the shallow water table aquifers, groundwater recharge is from precipitation that enters the ground and percolates downward through sandy beds. In the artesian aquifers, groundwater movement is generally downgradient within sandy strata between impermeable beds of clay. Discharge areas include streams, lakes, swamps, and the Gulf of Mexico.

Soils. The Ruston-Malbis-Lucy Association is predominant within the potential areas for land application. Soil associations are only general classifications, so additional information was determined by site investigations.

The soils series were related to the topographic relief. The more permeable sandy soils (Eustis, Lucy, and Ruston) occur in the upper elevations, while the clays and less permeable soils (Malbis, Beauregard, and Susquehanna) predominate in the lower areas and drainage channels.

In the Zion Hills No. 1 area, sandy and sandy loam soils up to a depth of 15 m overlay a continuous clay strata. A perched water table occurs above the clay strata, which appears at the soil surface as seepage in areas adjacent to Drake's Creek.

Major differences were noted in the Rosepine area. Soils were generally moderate to somewhat poorly drained.

The entire area is underlain by a strata of clay and silty clay occurring within 1.8 to 2.1 m of the surface.

Climate. The climate of Fort Polk is classified as humid subtropical with mild winters and hot, wet summers. Climatic data were collected from 1951 to 1973 at the U.S. Weather Station in Leesville, Louisiana. The temperature of the area ranges from an average high of 36.7°C in the summer months to a low of -3.9°C in the winter months. The mean rainfall per year is in excess of 135 cm and the maximum annual rainfall occurred in 1923 with 224 cm. Snow is infrequent and occurs about once a year.

The number of operating days per year for land treatment alternatives can be obtained from basic data, such as length and probability of occurrence of freeze-free periods, and minimum temperatures. A 90% probability exists that freeze-free periods will be 206 days using 0°C, 264 days at -4.4°C, and 340 days at -8.9°C.

A 50% probability exists that monthly precipitations will vary from 5.8 to 14.1 cm. A 5% probability exists for monthly precipitation occurrence between 21 and 33.5 cm.

Land Use. All the land at Fort Polk is utilized for the training mission at the Fort. The additional use of land to receive and treat wastewater will require a dual use within an area.

Forest Management. Harvesting and reforestation of woodlands on Fort Polk are under the management of the U.S. Forest Service, Vernon Division. The management cycle from planting to harvest for longleaf pine trees is usually about 40 years. The age of the trees in the areas considered for a land treatment system varies from 30 to greater than 50 years. The trees would require cutting and reforestation during the design period, if a forest irrigation system were to be considered.

Agricultural Management. Although natural fertility levels in the soil are low, climatic conditions at Fort Polk are favorable for forage production. The growing season extends from April to November, with ample distribution of rainfall typically occurring.

As an example, the potential forage production using coastal Bermuda hay with nitrogen fertilization of 336 kg/ha will be given. An established stand would yield 4 to 6 cuttings, with a yield of 17 metric tons/ha per cutting. An annual yield of 68 to 100 metric tons/ha can be estimated.

The management of pastureland can include year-round pasture with a forage such as Bermuda grass, or a combination of Bermuda grass with a winter planting of rye for grazing. The year-round pasture with an established forage appears to involve less farm operation and better agricultural production.

Special Conditions. A protective program has been instituted specifically for the endangered species Dendrocopos borealis, common name is the red-cockaded woodpecker. A program of surveying, mapping, and marking the location of the red-cockaded woodpecker colonies, den trees, nesting trees, and support stands on the military post is a continuing project.

Unusable areas occurring in the study area include the stream beds and adjacent seepage area, the nesting trees and 152 m surrounding area of the endangered woodpecker species, and some slopes of 20 to 30%. The location of these areas is shown in Figure 1.

Water Quality. All wastewater treatment alternatives at Fort Polk would involve discharges to streams of the Upper Calcasieu River Basin (either directly or through seepage above streams). Analysis of the water quality of the streams was provided in sampling programs in 1974 and 1975. The changes in water quality from nonpoint and point sources are most notable for increases in COD, suspended solids, and ammonium nitrogen.

Wastewater Treatment

Existing Treatment Facilities. Although there are two wastewater treatment facilities on Fort Polk, serving North and South Fort Polk, the feasibility study for the land treatment is limited to South Fort Polk. The South Fort Polk wastewater treatment plant is being expanded by the addition of essentially duplicate units that added 44 L/s in capacity to the existing facilities. This addition gives the facility a total design capacity of 110 L/s. The existing trickling filter will provide secondary level preapplication treatment prior to land application.

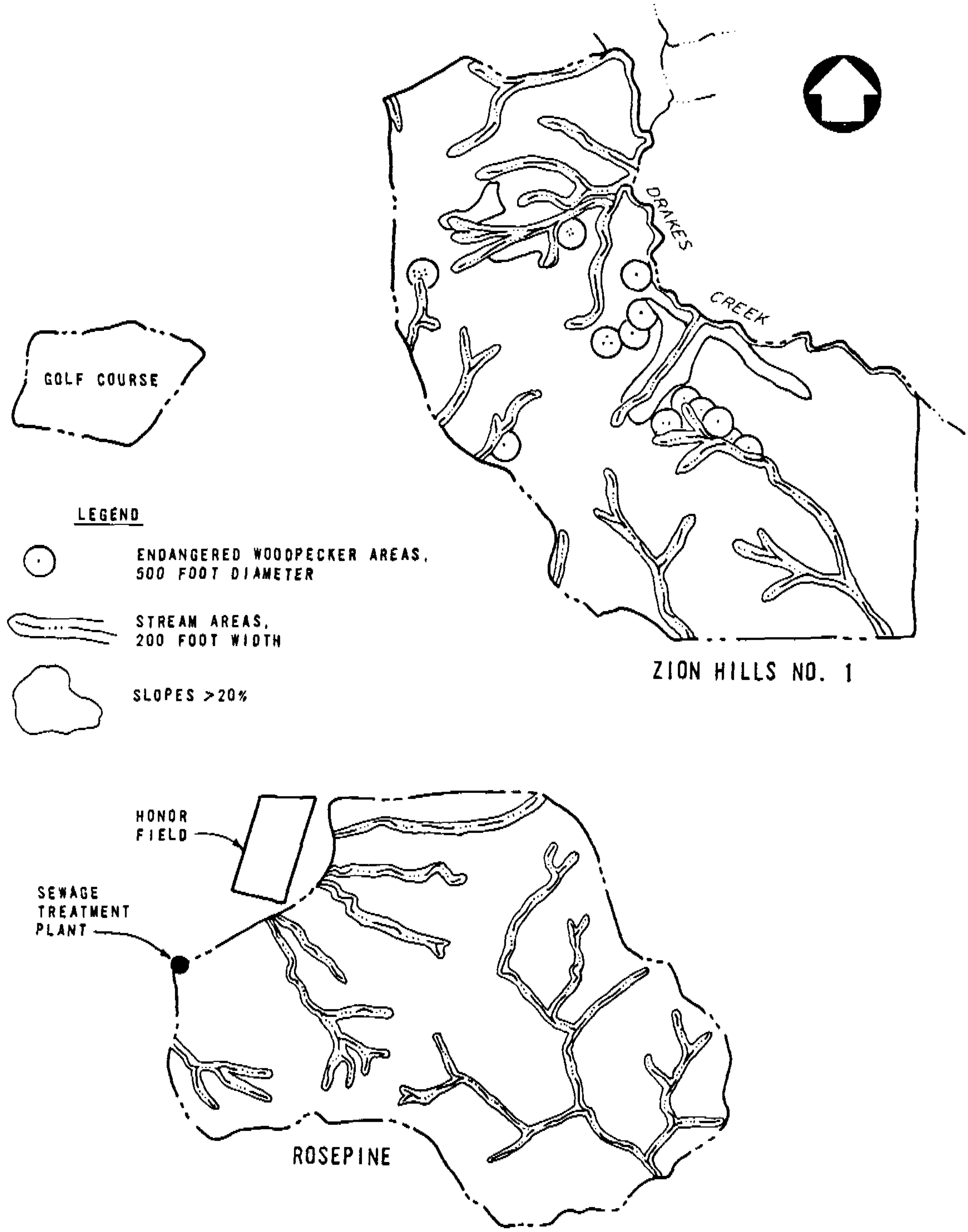

Figure 1. Unusable Land Application Areas

Wastewater Characteristics. The wastewater effluent was characterized by the USAEHA during the period of 4-15 November 1974. The results of the analyses show typical values for a secondary effluent with no excessive concentrations of major or minor constituents.

Flows. The design flow is 166 L/s. The present flow at the South Fort Polk treatment plant is 66±17.5 L/s.

NPDES Permit Requirements. The South Fort Polk treatment plant expansion satisfies the present and proposed interim NPDES permit requirements. Long-term discharge requirements will apparently be based on oxygen demand criteria, which require high removals of BOD and nitrification of the effluent.

Nonpoint discharges of wastewater do not require permits for discharge. Discharge to permanent groundwater would be required to meet EPA drinking water standards.

Land Treatment Criteria. The state discharge criteria report that "the following should be taken into consideration in the design of such [land treatment] facilities. (1) The equivalent of secondary treatment of sewage is mandatory prior to land treatment. (2) Disinfection is required only if there is potential human exposure either directly or by spray. (3) Land treatment with sewage is not acceptable for dairy pastures or for crops normally eaten uncooked" [1].

FIELD INVESTIGATION

Soils

Initial Soil Reconnaissance. Zion Hills No. 1 and the Rosepine areas were selectively surveyed within the areas that appeared suitable for one or more of the land application alternatives. Areas were delineated after field reconnaissance with regard to soil type, drainage, and slope. Approximately 1,000 ha in Zion Hills No. 1 and 773 ha in the Rosepine areas were selected to be intensively mapped.

Detailed Soil Investigation. Areas selected during the reconnaissance survey were mapped in detail according to standard procedures of the SCS. Pits were dug to 3.0 m by backhoe in the major soil types and profile descriptions were made. Soils were sampled for chemical and mechanical analysis by soil horizons. The Eustis, Lucy, and Ruston soils in the Zion Hills No. 1 area are well drained soils with fine sand or sandy loam surface horizons. The Malbis and Susquehanna soils, occurring in the Rosepine area, are moderately well drained to somewhat poorly drained acid soils. Surface horizons range from loamy sand to sandy loams and are usually thinner than 51 cm. The Eustis, Lucy, and Ruston soils contained an argillic horizon at a depth of 61 to 122 cm to maximum clay accumulation.

Geologic Borings

Borings were made in the Zion Hills No. 1 area to very firm clay or sandy clay in the bottom of each hole. The contour map of the top of this impervious strata indicated a 1.5% dip to the east in the northern section and less to the south. The clay strata were apparently continuous beneath Zion Hills No. 1. Evidence for this conclusion was that similar material was recovered in every boring and the color and texture were consistent in all borings. Boring logs showed that these strata were at least 4.6 m thick under most of the area, which was also an indication of the continuous nature of the horizon.

Soil Hydraulic Conductivity Testing

The hydraulic capacity of the Eustis, Lucy, and Ruston soils was determined by field investigations. The 5.5 m diameter basins were constructed on the site and operated for three or four times.

The locations for the basins were determined after soil mapping. Each basin was sited on a representative soil profile for the soil series and adjacent to a test pit.

The three basins were flooded rapidly (less than 15 minutes) to a depth of about 15 cm by pumping from a 3,800 L tank truck. Time intervals and water height were recorded until most of the water infiltrated.

The test results for the Eustis soils are shown in Figure 2. The approximate steady-state rate was chosen as the hydraulic conductivity. The values of hydraulic conductivity

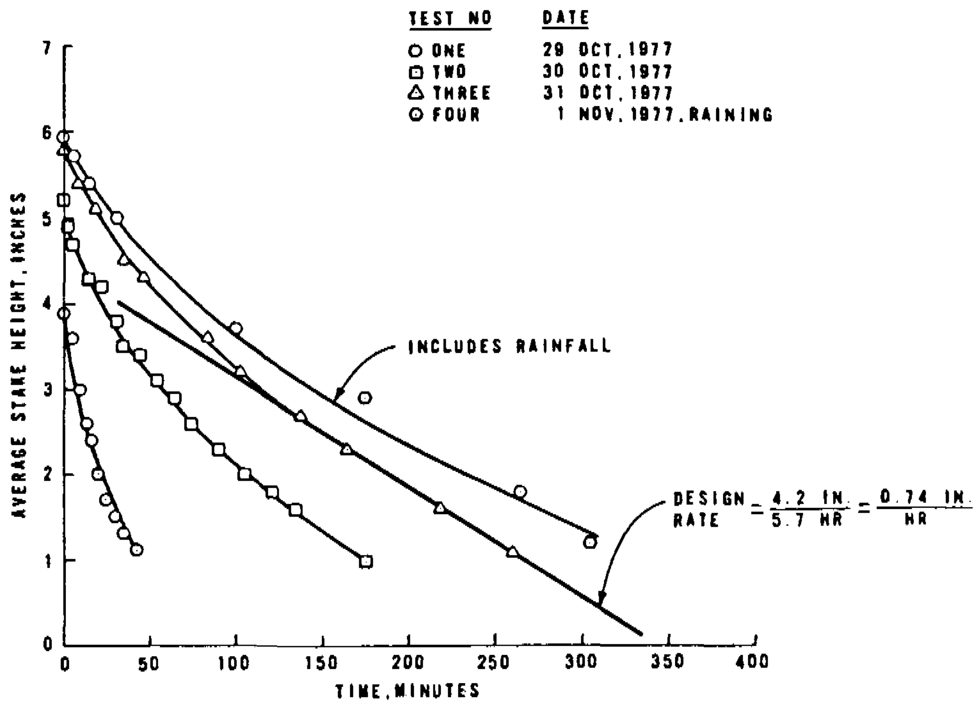

Figure 2. Hydraulic Conductivity, Eustis Soil Test Basin

are 45.2 cm/d for the Eustis soil; 21.8 cm/d for the Lucy soil; and 26.2 cm/d for the Ruston soil. The values represent vertical movement since no significant lateral movement was indicated by the soil moisture cell readings.

CRITERIA DEVELOPMENT

Application Rate Criteria

Application rate criteria are determined by individual site conditions and are specific for the type of system and the treatment requirements. The application rate criteria are determined for a rapid infiltration system by the site-specific hydraulic conductivity. Further limitations on a rapid infiltration system may come from requirements for short-term nitrogen removal or long-term phosphorus removal. For slow rate systems, application rate criteria are determined from water balance, nitrogen balance, or agronomic considerations. Hydraulic conductivity of the soil should be considered, but may not be a major limitation. For overland flow systems, application rates are based on climate, length of terrace, and slope of terrace.

Rapid Infiltration. The development of criteria for rapid infiltration basin design considered the hydraulic conductivity of "clean" water, the level of preapplication treatment, climate, and amount of bed surface maintenance. In considering that all of these factors are highly favorable, the design rates for the soil types are: Eustis, 46 cm/wk; Lucy, 23 cm/wk; and Ruston 23 cm/wk [2].

The operation cycle of the basins encompasses 2 weeks flow of wastewater applied during 2 days of continuous flooding. For example, the Eustis soils basins would receive about 91.4 cm of wastewater during a 48-day period, with the remaining 12 days of the cycle to allow for complete infiltration (estimated 1 to 3 days) and reaeration (about 11 to 9 days) of the bed profile. A complete application cycle would encompass 2 weeks and require seven sets of basins.

The subsoil water movement was calculated from a measured infiltration rate, a known depth to impervious clay strata, and particle size classification of materials from the soil surface to the clay strata. The limiting soil horizon was the argillic horizon or depth to maximum clay concentration. The particle size classification below the 3 m depth of soil survey test pits was obtained by split-spoon samples during geologic boring. No increase was noted in clay content of the samples that would indicate a restrictive layer to water movement.

The measured soil infiltration and permeability of the surface horizon was found to be from 1.9 cm/h to 0.91 cm/h. Conservative estimates of horizontal and vertical permeability were made at 0.25 cm/h and 0.025 cm/h, respectively. The velocity of flow away from the basin was computed using an observed clay strata gradient of 1.5%. This minimum gradient of the clay would produce a flow velocity of 0.2 m/d assuming a specific yield of 30%. Wastewater application mounding would increase the gradient, so a greater velocity would result from groundwater mounding.

The transmissivity of the subsoil would move a minimum volume of 61 $m^3/m^2 \cdot d$ at a saturated depth of 4.6 m above the clay. A basin width of 61 m receiving a surface application of 0.9 m/d would not exceed subsoil transmissivity under these conditions. Since this value exceeds the maximum application and provides adequate subsoil flow using conservative criteria, no underdrains or recovery wells are required to assist subsoil water movement or prevent excessive groundwater mounding.

Slow Rate. Application rates for a slow rate system are determined by nitrogen limitations, agronomic management, or hydraulic conductivity. The soil types considered for the slow rate system were shown to have adequate hydraulic conductivity of 45.7 cm/d, so hydraulic limitations should not be a concern. Agronomic considerations should not pose limitations at the site. The mean number of days when the minimum temperature drops below 0°C is 40, occurring between November and March. The January monthly mean for the daily minimum temperature is 3.3°C.

A nitrogen and water balance was used to calculate acceptable applications that satisfy a potable water nitrate-nitrogen limit. The following assumptions were used:

Item	Design assumptions
Total nitrogen (applied), mg/L	20
Crop nitrogen uptake, kg/ha	336
Annual evapotranspiration (ET), cm	96.5
Annual precipitation minus ET, cm	39.4
Allowable leaching concentration, mg/L	10
Denitrification, % of total nitrogen application	15

The nitrogen balance was computed using design assumptions and a water balance for the average year. The soil conditions are favorable for microbial denitrification with a high annual temperature and moist soil conditions. The lack of organic matter may have a limiting effect, so a denitrification rate of 15% of the total applied nitrogen was utilized [2]. The overall results are summarized in Table 1.

Table 1. Annual Summary of Nitrogen and Water Design Balances

Applied wastewater, liquid, m	5.3
Applied wastewater nitrogen, kg/ha	1,053
Denitrification loss, kg/ha	157
Crop uptake nitrogen, kg/ha	336
Leaching nitrogen, kg/ha	560
Average leaching concentration, mg/L	9.8
Leaching volume, precipitation and wastewater, m	5.7

Woodland Irrigation. Application rates are determined in a similar manner for woodland irrigation and slow rate systems. Suitable data for the nitrogen uptake and evapotranspiration of tree species are substituted for the agronomic crops that are normally considered in a slow rate system. The nitrogen uptake of young evergreen trees was assumed to be 67.2 kg/ha. The annual evapotranspiration was 38 cm/yr for pine trees. The annual distribution of nitrogen uptake by trees is not as well defined as with agronomic crops.

Consequently, an annual nitrogen balance was calculated rather than a monthly balance.

An application rate of 2.5 cm/wk was assumed to occur during a 13-week period from December to February. A rate of 5.0 cm/wk was assumed to occur during a 39-week period from March to November. The application rate produces an annual average nitrate concentration of 9.9 mg/L [2].

Overland Flow. Application rate criteria for overland flow systems have been determined empirically from pilot, research, and demonstration studies. The variable design factors are terrace gradient, terrace length, and to a lesser degree, preapplication treatment.

For lagoon or secondary effluent, loadings of 15 to 40 cm/wk are considered. Lower values of 17.5 to 25 cm/wk should be considered (1) for slopes greater than 6%, (2) for terraces less than 45 m, or (3) because of reduced biological activity during very cold weather.

A hydraulic application rate of 15 cm/wk was used for overland flow system feasibility at Fort Polk. The application rate represents an average value, because consideration must be given to nonapplication times due to climate and agricultural management.

Storage Criteria

Storage of wastewater is required for most land treatment systems, because of daily and seasonal imbalances between wastewater supply and allowable applications. Storage requirements can be controlled by either climatic constraints or agricultural management practices. Overland flow systems and many irrigation systems will be affected primarily by climatic constraints. Agricultural management practices, on the other hand, will usually control storage requirements in those irrigation systems where crop production is important. Rapid infiltration systems operate year-round under most conditions, and consequently have minimal or no storage requirements.

Rapid Infiltration. Storage requirements for a rapid infiltration system are determined by maintenance or emergency considerations, rather than climatic conditions. Precipitation on the infiltration basins will be retained by the 1.2 m high containment berm around the basins. This provides sufficient freeboard to contain the maximum monthly precipitation at the 5% probability level. All wastewater applied and precipitation falling into the infiltration basins will receive treatment by passage through the soil column. The only requirement for a separate storage basin would be to provide emergency wastewater retention.

Slow Rate. The storage requirement of a slow rate (irrigation) system can be estimated either from a climatic or a water balance evaluation. The climatic evaluation is based on weather records. Seasonal variations in wastewater generation are not considered, nor are the hydraulic capabilities of the soil or the consumptive water use by plants. In contrast, the water balance evaluation is a monthly accounting method that balances the quantities of wastewater generated and wastewater applied. The applied wastewater volumes can be computed from a nitrogen balance, plant consumptive water use, or a maximum hydraulic conductivity. Several computer programs are available to determine storage requirements [2].

The EPA-1 program tabulates the number of days of nonapplication based on daily climatic records. The decision on whether or not wastewater application occurs is based on assigned threshold values for temperature (0°C), precipitation (1.3 cm/d), and snowfall (2.5 cm of snowcover). The EPA-1 program produces the following results:

- EPA-1 estimated maximum storage - 6 days
- Length of maximum freeze period - 5 days

The EPA-2 storage program was developed to evaluate periods of extensive and continuous precipitation. Precipitation may limit application if surface runoff must be controlled or if continued saturated soil conditions need to be avoided. The results of the EPA-2 program based on an estimated available water capacity of 15 cm and reduced applications during periods of saturation or near saturation in the soil profile are given below:

Recurrence interval, yr	Slow rate system, EPA-2 maximum storage, d
Maximum	35
20	31
10	16
4	8
2	6

Water balance storage requirements were computed to be 74 days from wastewater application volumes that varied from 20 cm/month to 79 cm/month.

An engineering decision was necessary to determine an optimum storage requirement for least cost. Variable factors influenced the overall cost as outlined in Table 2.

Table 2. System Design Changes Based On Variability of Application Schedules

Application schedule	Storage volume	Distribution pumping, power requirement	Storage cost	Application area	Field components, cost
Uniform (up to 10 cm/wk)	Decrease	Decrease	Decrease	Increase	Increase
Variable (5 cm/wk to 20 cm/wk)	Increase	Increase	Increase	Decrease	Decrease

The two extremes in application scheduling were a near uniform application rate and a variable rate based on nitrogen utilization by plants. The storage requirement based on a near constant application schedule was 40 days. The storage requirement provides ample storage to satisfy additional operational or existing climatic requirements.

Overland Flow. Storage requirements for an overland flow system are determined by daily operating considerations, since application rates are near constant throughout the year. Nonapplication periods occur when daily precipitation is greater than 1.3 cm; air temperature drops below -3.9°C; and greater than 2.5 cm of snowcover exists. These criteria are computed in the EPA-2 computer program [2] and yield the same storage requirement as a slow rate system.

Woodlands Irrigation Systems. The storage requirements for a woodlands irrigation system were calculated by a monthly water balance. The maximum storage requirement of 38 days exceeds the climatic criteria generated by the EPA-2 program.

SUMMARY

Criteria for system feasibility and design were developed for Fort Polk, Louisiana. The basic site conditions were described. Field testing procedures and surveys provided site specific criteria. Application rate criteria were determined from hydraulic, climatic, and agronomic considerations. Storage requirements were computed by system need, water balances, or climatic constraints.

The selection of the best land application alternative required an in-depth engineering analysis of cost and location factors as well as the design criteria. This presentation was limited to design criteria. A cost-effectiveness analysis determined that rapid infiltration was the best apparent alternative.

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DESIGN CRITERIA

PRETREATMENT TECHNIQUES AND DESIGN MODIFICATIONS FOR RAPID INFILTRATION LAND TREATMENT SYSTEMS

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Pretreatment techniques to optimize infiltration rates and the hydraulic capacity of rapid infiltration systems were investigated utilizing laboratory soil columns and field installations. Domestic wastewater was used to compare the effect of: aerobic versus anaerobic pretreatment, ozonated effluent, applied wastewater BOD_5, TSS, and algal cell concentration as well as varying temperatures on the infiltration rates of three major soil types. A cube root relationship has been developed which can be

$$\sqrt[3]{\frac{(BOD + SS)_2}{(BOD + SS)_1}}$$

used to estimate the reduction in application area requirements attributable to increased removals of BOD_5 and TSS. An optimum pretreatment selection procedure is proposed which minimizes the total present worth cost of both the pretreatment and application area portions of a rapid infiltration system.

Laboratory scale rapid infiltration basins incorporating a coarse sand filter nitrification step followed by a fixed film denitrification step have been developed utilizing methanol and greywater as the carbon source. Optimum hydraulic and organic loading rates for the nitrification-denitrification process were determined. A pilot scale field installation was constructed in October 1977 using domestic waste from a single family residence. A conceptual design for a full scale rapid infiltration-nitrification/denitrification basin is proposed for land treatment sites requiring a high degree of nitrogen removal prior to groundwater recharge.

Ongoing laboratory studies have investigated using expandable montmorillonite clay structures in aqueous suspensions for removal of soluble, refractory or toxic organics prior to rapid infiltration land treatment. The objective being incorporation of chemically active soil types, which are characteristically impervious, in a land treatment system utilizing less active, pervious soils. Commercial and laboratory grades of expandable Wyoming bentonite and non-expandable kaolinite were contacted in batch experiments to determine wastewater total organic carbon (TOC) adsorption rates and capacities. The results of the batch experiments indicate that adsorption or ion-exchange (organic cations) of wastewater TOC on the internal structure of expandable clays is a function of both the level of wastewater treatment and the TOC/clay concentration rate.

Optimum clay adsorption concentrations and uptake rates for the wastewater tested justify their use in a pretreatment step prior to application on rapid infiltration basins.

OPTIMUM BOD_5 AND S.S. PRETREATMENT REQUIREMENTS

Pretreatment for Reducing Infiltration Area

Dunbar(1) reported that city sewage was first treated in Bunzlau, Germany in 1559. Raw sewage was put on land. Enthusiasm with wastewater farming grew. In England Sewage

Utilization Acts were passed in 1864-67. Urban population increased along with industrialization. As a result land disposal areas became overloaded and research was initiated to decrease land area/wastewater ratio. Sir Edward Frankland developed a sand filter which could be loaded about ten times faster 0.03 $m^3/m^2/d$ than an infiltration land area at a load of 30 $m^3/ha/d$. In 1860's Austin recommended that suspended solids should be removed before land application. Tatton's work(1) supported the concept of pretratement (sand filtration) and he suggested the following loads in $m^3/ha/d$:

SOIL DESCRIPTION	WITH PRETREATMENT	RAW SEWAGE
sandy loam over clay	45	23
turf over sand	90 - 135	33
sand over gravel	213	45

A Royal Commission (1898) (1) recommended the following loads in $m^3/ha/d$ for soil infiltration surfaces for pretreated effluent:
sandy loam over clay - 12 to 25 $m^3/ha/d$
sand over gravel - 115 - 230 $m^3/ha/d$

It was recognized that pretreatment enhances infiltration and properly loaded infiltration fields lasted 30 years without losing their performance.

The first laboratory study on the effect of pretreatment on soil clogging was conducted by Winneberger, J.H. et al. (2) using 6 Oakley sand columns and 2 Columbia sand columns. Half of the columns were intermittently dosed with septic tank effluent and the remainder with an extended air plant effluent. The column experiment was terminated after 5 weeks, before clogging rates reached equilibrium or zero rate. During the dosing of the columns the package plant effluent quality improved. The mean rate of clogging was greater with the septic tank effluent (1.5 to 2 fold). Because the extended air reactor was not operating properly and the duration of testing was short much of the data (severity of clogging) remained within the experimental error range.

University of Wisconsin investigators (3) tested Almena silt load with distilled water, septic tank effluent and aerobic unit effluent. Two separate studies were conducted. The first study involved 6 columns with soil moisture pressure profiles and quantity of infiltrated liquid measurements. The study showed that the aerobic unit effluent infiltrated more liquid but the soil columns showed greater moisture pressure drops. The study did not include enough samples and showed that the columns were not identical. A second study was conducted with more soil columns, applying effluents that did not differ in quality BOD_5 + SS = 75 mg/l for septic tank effluent and BOD_5 + SS = 88 mg/l for aerobic unit effluent. Slightly higher ponding times were found for the aerobic unit effluent as should be expected. The conclusion (3) that pretreatment was not found to affect the degree of clogging in fine textured soils could not have been reached from their experimental data. However, when two different effluents with a BOD_5 + SS ratio of 0.43 were applied to sands a substantial difference in clogging was found (3).

A laboratory study by Laak (4) used 30 soil columns and three soil types loaded with various load rates for up to 160 days. Two different effluents with a BOD_5 + SS ratio of 0.45 was used. The results showed that the clogging progressed according to the cumulated total mass or load applied (the sum of BOD_5 and SS in kg) regardless if the effluent applied was pretreated aerobically or anaerobically. The effect of improved pretreatment was also verified in the field where three extended aeration units were put in line, to units in soil areas where septic tank systems had failed. A later study (5) conducted at a field research station confirmed that a BOD_5 + SS ratio of 0.7 lagoon/septic tank effluent over a 2 year period clogged horizontal 2 m^2 soil surfaces (2 different soils) to a lesser degree. A laboratory study (4) showed that the clogging layer behaved similarly at 20°C as well as at 8.5°C. A later laboratory (6) study showed that the clogging layer reacts to a temperature rise of 15 degrees centigrade by becoming temporarily more permeable. A brief study (9) showed that ozonation reduces primary effluent color suspended solids and 56% of the BOD_5. When the ozonated effluent was infiltrated into matured clogged soil surfaces, the soil surface biocrust microorganism population did not change.

The laboratory study (4) using columns yielded a relationship which could be expressed by the following equation:

$$A_x = A_o \sqrt[3]{\frac{(BOD + SS)_x}{(BOD + SS)_o}} \qquad (1)$$

where BOD_5 and SS are expressed mg/l; subscripts o - known; x - to be selected; A - infiltration area, m^2.

Equation 1 can be used to compute the area of infiltration or clogging surface required if the hydraulic load and soil permeability is kept constant but the concentration of BOD_5 + SS is changed by pretreatment. Area A_o can be computed for a $(BOD + SS)_o$ value of 150 to 250 mg/l (representing a septic tank effluent) from Equation 2 which was derived from plotted data (7,8) gathered by different investigators.

$$A_o = \frac{Q \times 8.64 \times 10^6}{40\ k - \frac{5.6}{\log_{10} k}} \qquad (2)$$

where k = saturated permeability in cm/s; Q = flow in m^3/s; and A_o - infiltration area in m^2. Kropf's (4) laboratory study, using a single type of effluent (septic tank effluent) showed that by dosing and up to 3 days resting a smaller volume of liquid is passed in 19 weeks than through continuously loaded soil column surface (3 different soils) at equilibrium or zero clogging rates which are reached in 3 to 6 months. A total of 72 soil columns were tested. The experiment showed that to maintain higher infiltration rates of equilibrium or mean zero clogging rates the clogging surface needs to be 'starved' by more pretreatment or by applying loads less than the maximum.

Table I shows the reduction of infiltration area at different BOD_5 + SS ratios. Equation 1 was used to compute the reduced infiltration surface when pretreatment is increased.

Optimizing the Cost of Pretreatment

An optimum level of pretreatment for rapid infiltration systems can be determined as a function of cost using the results of the previous section. In order for higher levels of BOD_5 and SS removal above a given base level, to be economically justified, the increase in total life cycle cost of the additional pretreatment must be at least equal to the reduction in total life cycle costs attributable to the reduction in application area. One method which can be used to relate total life cycle costs of any given wastewater treatment alternative to another is the Present Worth Cost (PWC) analysis. Using the PWC procedure, all capital and operational and maintenance costs as well as salvage values at the end of a predetermined study period are converted to an equivalent present worth value at the beginning of the study period for each wastewater treatment alternative under consideration. The alternative with the lowest PWC procedure can be found in texts on engineering economics; however, a basic approach to optimizing rapid infiltration pretreatment cost would then be to minimize the PWC of the entire system; i.e., pretreatment plus application area PWC according to the following relation.

$+\Delta$PWC of Increasing Pretreatment Above a Base Level $\leq -\Delta$PWC of Decreasing the Application Area Required (3)

Inspection of Eqs. 1 and 2 indicates that the results of such an economic comparison will be a function of: wastewater flow, the base level of pretreatment, the saturated in-situ permeability of potential application sites and the (BOD_5 + SS) removal efficiency of the pretreatment system as well as local costs for the construction and maintenance of the rapid infilatration systems under consideration. The analysis also assumes that there are no constraints on the availability of land. In order to simplify the analysis a "unit application area PWC breakeven point" can be utilized which is defined as the incremental increase in PWC to go from the base level of pretreatment to a higher level of pretreatment divided by the area saved in using the higher level of pretreatment. The general "unit application area PWC breakeven point" equation is given below:

Unit application area PWC breakeven point (for the ith level of pretreatment) $= \frac{\text{PWC } i^{th} - \text{PWC base}}{\text{Area}_{base} - \text{Area}_{i^{th}}}$ (4)

The unit application area PWC breakeven

TABLE I

Degree of Pretreatment vs. Clogging Surface Required

$\frac{(BOD + SS)_x}{(BOD + SS)_o}$ ratio	$\frac{A_x}{A_o}$	Percent Reduction in Infiltration Surface
1:1	1	0
1:4	1.6	38
1:8	2	50
1:16	2.5	60

$(BOD + SS)_o$ is for septic tank effluent and has a value of 150 to 250 mg/l. A_o can be computed from Eq. 2.

point so derived is an economic indication of the total life cycle costs (PWC) which can be expended per unit of area in the design, construction and maintenance of rapid infiltration application basins using the base level of pretreatment before increased pretreatment and removal of BOD_5 or SS is justified. It should be emphasized that this conclusion is based primarily on maintaining the long term hydraulic functioning of the rapid infiltration system and that there may be other reasons for increasing pretreatment above a base level including local regulations, codes or special conditions of the application sites under study (i.e., odors, aesthetic factors etc.).

As an example of the optimization procedure and as a guide to the environmental planner, a pretreatment optimization curve for rapid infiltration systems was developed using average but current (1978 basis) capital costs, operational and maintenance expenditures and salvage values for three levels of pretreatment and a range of wastewater flow rates. See Figure 1. The in-situ saturated permeability of the application site was assumed to be K-0.05 cm/s, the base level of pretreatment, level 1, was taken as primary sedimentation and all levels of pretreatment were defined in terms of $(BOD_5 + SS)$ in accordance with Eq. 1. A 20 year planning period was used with an interest rate of 6-5/8 percent. A summary of the data used in constructing Figure 1 is given in Table II.

Inspection of Figure 1 indicates that the unit application area PWC breakeven point generally decreases with increasing flow rates thus emphasizing the increasing economic feasibility of increased pretreatment at the higher flow rates. Generally Level 2 (BOD + SS = 60 mg/l) is the more economical pretreatment step above the base level 1 (BOD + SS = 250 mg/l) if the unit application

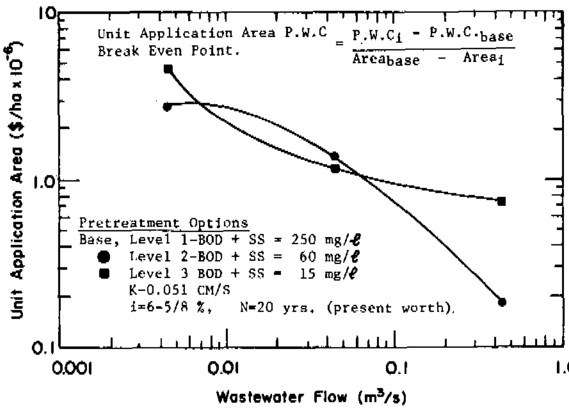

Figure 1. Rapid infiltration pretreatment cost optimization curve.

area PWC breakeven point is exceeded. However, Figure 1 also indicates that there is a range of design flows from 0.007 m^3/s to 0.065 m^3/s where level 3 $(BOD_5 + SS = 15$ mg/l) would be a more cost effective pretreatment step if the breakeven point for level 1 is exceeded.

The environmental engineer and planner should feel free to modify the pretreatment optimization procedure and mathematical relationships as necessary to more accurately reflect actual conditions in the study area so that the basic elements of the procedure can serve as an accurate guide in the selection of the most cost-effective rapid infiltration system possible.

OPTIMIZING NITROGEN REMOVAL WITH A COMBINATION RAPID INFILTRATION-NITRIFICA_ TION?DENITRIFICATION BASIN

A laboratory study investigated biological denitrification efficiency using methanol, greywater and settled

Table II. 1978 Pretreatment Costs

Flow m^3/s	Level 1 PWC US \$ x 10^6	Level 1 area ha	Level 2 PWC US \$ x 10^6	Level 2 area ha	Level 3 PWC US \$ x 10^6	Level 3 area ha
0.00438	1.02	0.59	1.64	0.37	2.7	0.23
0.0438	2.47	5.9	5.53	3.7	6.6	2.3
0.438	13.36	59	31.6	37.0	39.7	23.0

sewage as supplemental carbon. The experimental model consisted of six coarse sand filters providing the nitrifying step followed by six fixed-film anaerobic denitrifying reactors.

The nitrifying reactors had 0.61 m of sand with a uniformity coefficient of 4.0 and an effective size of 0.16 mm. The upflow denitrifying reactors consisted of columns filled to a depth of 0.61 m with 0.05 m to 0.1 m diameter stones.

Septic tank effluent taken from the University of Connecticut Wastewater Experiment Station and primary effluent from the University of Connecticut sewage treatment plant was dosed twice daily into the nitrifying reactors at a rate of 0.068 m^3/m^2/day. The three supplemental carbon sources were added (C:N of 3:1) prior to the denitrification step. All six reactors were operated in parallel under the same hydraulic mode of operation.

Analyses were performed on samples for the concentration of nitrates, nitrites and ammonia nitrogen using a Technicon Auto-Analyzer. BOD_5, suspended solids, and dissolved oxygen were determined as per Standard Methods (12).

Results

Denitrification columns receiving greywater as a carbon source removed 71 percent of the available nitrate while methanol removed 83 percent and settled sewage removed 4.3 percent. The BOD_5, ammonia-nitrogen and suspended solids concentration in the final effluent of the denitrifying reactors increased insignificantly after using greywater or methanol as carbon sources. No significant difference in nitrogen or denitrification efficiency was observed by loading with septic tank effluent or primary effluent.

Using the results of the laboratory nitrification-denitrification studies a conceptual design for a combination rapid infiltration nitrification/denitrification basin is under development at the University of Connecticut. The key design elements of the basin are shown in Figure 2 and include the controlled application of wastewater on a coarse sand filter sized for complete nitrification of the reduced wastewater nitrogen. Once the nitrified effluent has passed through the sand filter, it would then enter a subsurface fixed film denitrification reactor consisting of 0.05 m to 0.1 m diameter stones. Supportive material in the form of a graded mineral aggregate filter will be necessary to prevent breakthrough of nitrifying sand filter while still maintaining hydraulic flow. At this point, a supplemental carbon source, such as methanol or greywater, would be added to the fixed film reactor using perforated diffusers. The sizes of the denitrification chamber would be sufficient to provide a hydraulic residence time of 4 to 5 days to complete the biological denitrification reactions. The nitrogen gas end product would be vented via a continuation of the rock filter to the atmosphere. As a means of ensuring the hydraulic residence time necessary to complete the denitrification step, a Controlled Perviousness Layer (CPL) would be provided in accordance with Eq. 1 and 2

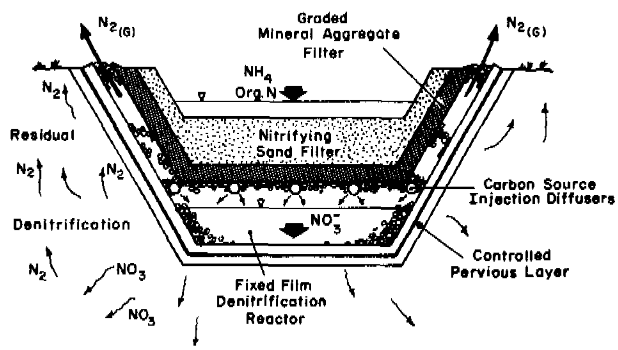

Figure 2. Conceptual design of a combination rapid infiltration nitrification/denitrification basin.

to insure flooding of denitrification reactor between the dosing periods of the basin. As an additional feature of the basin, the final leachate percolating through the soil mantle would contain sufficient soluble carbon and acclimated denitrifying bacteria to provide a "residual" denitrification effect as the leachate moves away from the application area. A pilot scale field installation of the nitrification/denitrification basin was constructed in October 1977 using domestic waste from a single family residence.

The results of laboratory scale studies and the ongoing field scale pilot operation indicate that the wastewater application rates as well as dosing and resting periods for the combination rapid infiltration-nitrification/denitrification basin are approximately the same as rapid infiltration system treating primary effluent (13). These results further emphasize the feasibility of incorporating the rapid infiltration-nitrification/denitrification basin in land treatment applications where a high degree of renovation of wastewater nitrogen is required prior to groundwater recharge of the effluent.

OPTIMUM RENOVATION OF WASTEWATER ORGANICS USING SELECTED CLAY STRUCTURES IN RAPID INFILTRATION SYSTEMS

The basic wastewater renovation mechanisms available to land treatment systems can approximately be divided into three categories: physical, chemical and biological. Rapid infiltration land treatment systems utilize filtration and biological degradation within the first meter of soil from the ground surface to remove wastewater organics and solids. Once past this biologically active zone, however, the renovation of soluble, refractory and toxic organic species in the wastewater is highly dependent on physical-chemical mechanisms such as adsorption, ion-exchange and precipitation.

A major characteristic of rapid infiltration systems, however, is the use of application sites composed of pervious soil types such as sands, gravels and loamy sands (13). It has often been observed that the renovation capacities of the purely physical-chemical mechanisms of the soil environment are inversely proportional to hydraulic capacity. Coarse sands for instance have typical specific surface areas of 0.03 m^2/g and cation exchange capacities of 1.0 meq/100 g while certain expandable clay structures have theoretical specific surface areas 800 m^2/g and cation exchange capacities 100 meq/100 g (13,15). Land treatment systems composed primarily of the more active clay fractions, however, would be severely limited by hydraulic constraints which would require excessive land areas for wastewater application. Also important is the fact that within the "clay size category" there is a wide range of physical, chemical and mineralogical properties. Montmorillonite, a layered silicate, for example, is one of the more active clay minerals and exhibits a unique swelling property in the presence of water and other polar molecules allowing movement of these species into the basal space between the silicate sheets (14,16,17). Kaolinite on the other hand, is a non-swelling clay, with a significantly less potential for renovation of wastewater organics.

Previous research has indicated that migration to and adsorption on the basal planes of expandable clays in aqueous suspensions is a function of the specific organic molecule, structure and size of the molecule, charge and Stern-Gouy layer effects, pH and the presence of other chemical species in solution (16).

It was concluded that the adsorption or ion-exchange of soluble wastewater organics, by even the most active clay fractions could exhibit an "optimum adsorption" range within which maximum uptake of organic species would be accomplished.

In order to confirm this effect, a series of batch adsorption tests were performed using primary (sedimentation), secondary (trickling filter) and tertiary (rapid sand filter) effluent from the University of Connecticut Sewage Treatment Plant. Three types of clays were dosed at various concentrations, pH and contact times in 0.45 μ filtered wastewaters. Soluble total organic carbon (TOC) was determined for each clay dosage before and at the end of each contact period. The clays studied included two swelling montmorillonite structures consisting of a commercial grade Wyoming bentonite supplied by ADM Chemical Company and a laboratory grade bentonite supplied by Fisher Scientific Company, and a non-swelling kaolinite structure using Kaolin N/F supplied by Fisher Scientific Company.

In order to determine optimum clay adsorption ranges for each wastewater, at a pH = 7.0, both the mass of TOC removed per gram of clay and specific removal efficiency were plotted against clay dosage.

The results of the batch tests are shown in Figures 3 through 8 and indicate that for the swelling bentonites both specific mass TOC uptake as well as specific removal efficiency are a function of the wastewater tested, the specific type and grade of clay used as well as its concentration in suspension. Bentonite TOC specific mass uptake and removal efficiency were maximized at clay concentrations of less than 50 mg/l. The Kaolinite specific mass TOC uptake and removal efficiency was consistently less than the bentonites while removal efficiency showed an increasing trend with increasing clay concentrations in suspensions. These results were not changed by varying the pH as shown for the primary effluent in Figure 9. The optimum pH for each maximum TOC adsorption range was determined by further batch testing and indicated that the optimum pH for the removal of the wastewater organics studied was 7.0. This result was attributed to competitive ion exchange and precipitation reactions involving wastewater inorganic species at other than neutral pH. Long term batch tests (14 days) were used to determine TOC uptake rate on clay structures. The time for the bentonites to reach equilibrium was shown to be a function of the wastewater used as well as clay concentration but it was essentially complete within 6 hours.

A method of measuring the specific surface area of the expandable bentonites in aqueous suspensions was developed using a monolayer coverage of the basal planes of the clays at the maximum

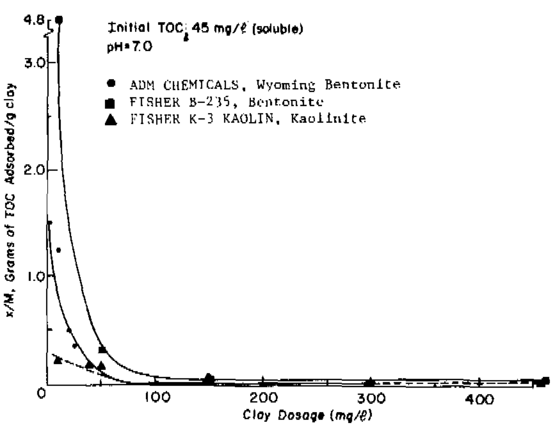

Figure 3. Specific mass uptake of primary sedimentation effluent soluble TOC as a function of clay dosage.

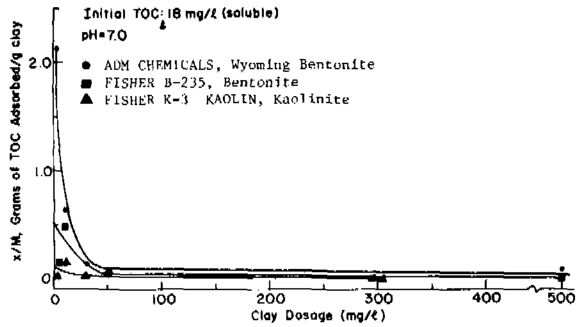

Figure 4. Specific mass uptake of trickling filter effluent soluble TOC as a function of clay dosage.

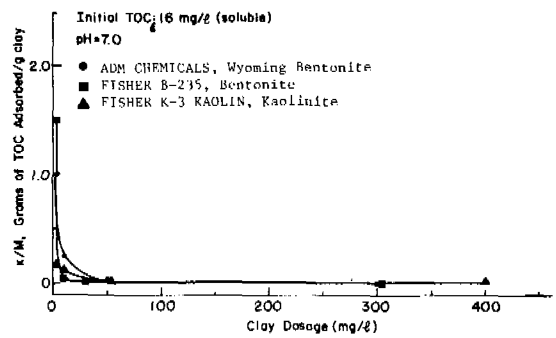

Figure 5. Specific mass uptake of rapid sand filter effluent soluble TOC as a function of clay dosage.

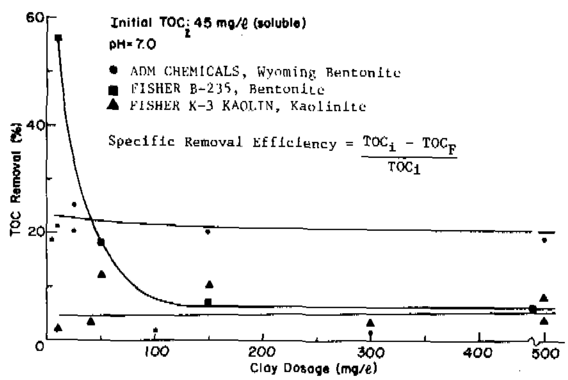

Figure 6. Specific removal efficiency of primary sedimentation effluent soluble TOC as a function of clay dosage.

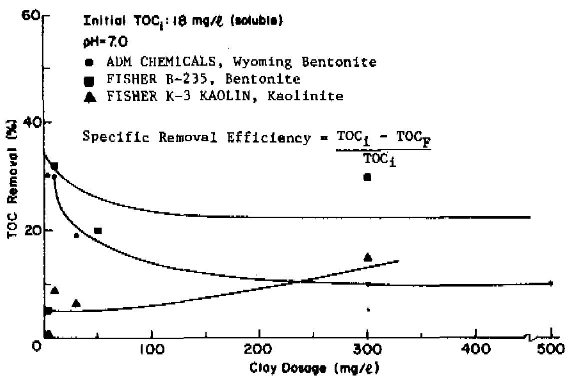

Figure 7. Specific removal efficiency of trickling filter soluble TOC as a function of clay dosage.

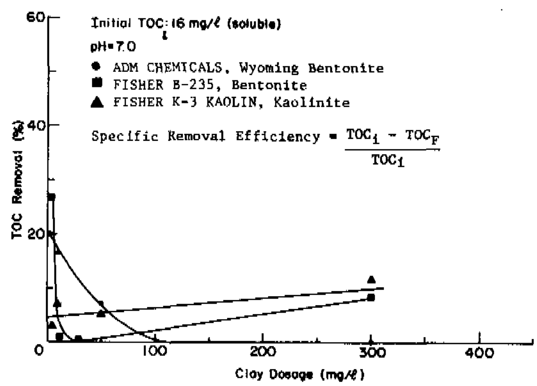

Figure 8. Specific removal efficiency of rapid sand filter effluent soluble TOC as a function of clay dosage.

A method of measuring the specific surface area of the expandable bentonites in aqueous suspensions was developed using a langmuir isotherm approximation of a monolayer coverage of the basal planes of the clays at the maximum adsorption point of methylene blue molecules. Using this technique, a maximum available surface area for the bentonites of 755 to 785 m^2/g was determined.

This procedure also demonstrated that the available specific surface area of expandable clays in aqueous suspensions is a function of clay concentration and the organic species present (i.e., TOC/clay ratio).

The ability of bentonite clay structures to selectively remove specific organics in wastewater suspensions was determined using batch experiments which monitored methylene blue dye uptake in demineralized water as well as the various filtered municipal wastewater. The results are shown in Figure 10 and indicate that the optimum uptake range of methylene blue is dependent on clay dosage as well as the wastewater solution used.

A conceptual design procedure for the optimum use of impervious but active clay structures to remove wastewater TOC in rapid infiltration systems is shown in Figure 11. The optimization procedure would relate the maximum uptake and removal of wastewater TOC by the clay structures to a minimum amount of hydraulic capacity lost in using these impervious structures. It can readily be seen that both these terms are related to the total mass of clay used. Furthermore, based on typical values of permeability for clays and sands as well as the previously determined optimum

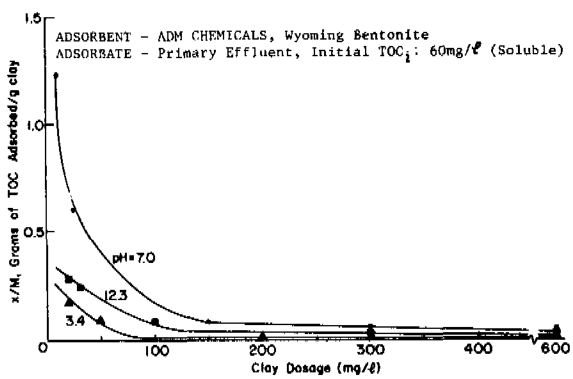

Figure 9. Specific mass uptake of primary sedimentation effluent soluble TOC as a function of pH and clay dosage.

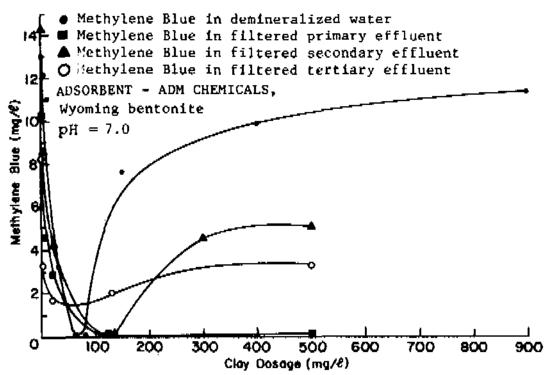

Figure 10. Specific removal of methylene blue dye in suspensions of various wastewaters as a function of clay dosage.

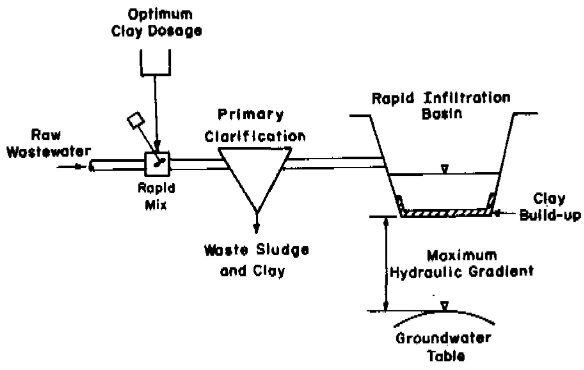

Figure 11. Conceptual design of a rapid infiltration system with optimum utilization of clay structures for removal of wastewater total organic carbon.

clay dosage requirements the hydraulic losses incurred for the wastewaters studied would not significantly affect the normal hydraulic operating range of rapid infiltration systems treating primary effluent.

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DESIGN CRITERIA

PENN STATE'S WASTEWATER IRRIGATION SYSTEMS - PAST AND FUTURE

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ABSTRACT

Three distinct areas are considered--the wastewater renovation and conservation research system used at Penn State for the past 15 years, irrigation system design research performed during these years, and the expanded wastewater irrigation system presently being installed.

The 22 ℓ/s (0.5 mgd) solid-set, above-ground, sprinkler irrigation system was designed in 1962 and installed in 1963. This research system irrigated secondary treated wastewater year-round, including winter periods with temperatures as low as -25° C (-13° F). The 26 ha (64 ac) overall facility included dozens of smaller systems irrigating various crops and woodlands.

System design discussed includes various sprinkler patterns and spacings, commercial and specially fabricated sprinkler heads, and hydrologic effects of operational procedures and spacings. System design aspects presented represent a half-dozen formal theses, as well as extensive informal observations.

Phase I of the expanded system to handle 175 ℓ/s (4 mgd) has been installed. This phase includes a 1893 m^3 (0.5 mg) surge tank, two 0.26 MW (350 hp) pumping plants, and a 4.8 km (3 mi) long 45 cm (18 in) diameter ductile iron main. Design of Phase II, encompassing the remainder of the project, is approximately 85 percent complete. This phase involves all field irrigation system aspects and monitoring. Design in this phase includes modifications from the original system stemming from past research and experience.

RESEARCH SYSTEM - PAST

The pumping plant which supplied secondary treated and chlorinated effluent to the irrigation sites was located on the 60 cm gravity outfall line that by-passed the Duck Pond (Figure 1). A sump which functioned as a pump pit was installed in the by-pass line and a pump house was constructed over this 3 x 3 x 2.5 m sump. The pump house contained two pumping plants which were alternated manually, one serving as a standby. Each pumping plant included a vertical centrifugal pump, designed for an output of 22 ℓ/s at 160 m total head, and a 44.5 kW, 3-phase electric motor. The pump house also contained an in-line meter, which measured the amount of effluent pumped, and a pressure recording instrument equipped with high- and low-pressure automatic signaling devices to indicate possible failures in the distribution or irrigation systems.

A 15 cm buried main line conveyed the effluent to the Astronomy and Gamelands Sites. The main line was buried at least 45 cm; however, since this depth was not adequate to prevent freezing, the wastewater had to be pumped continuously during cold weather. Appropriate vents and drains permitted air

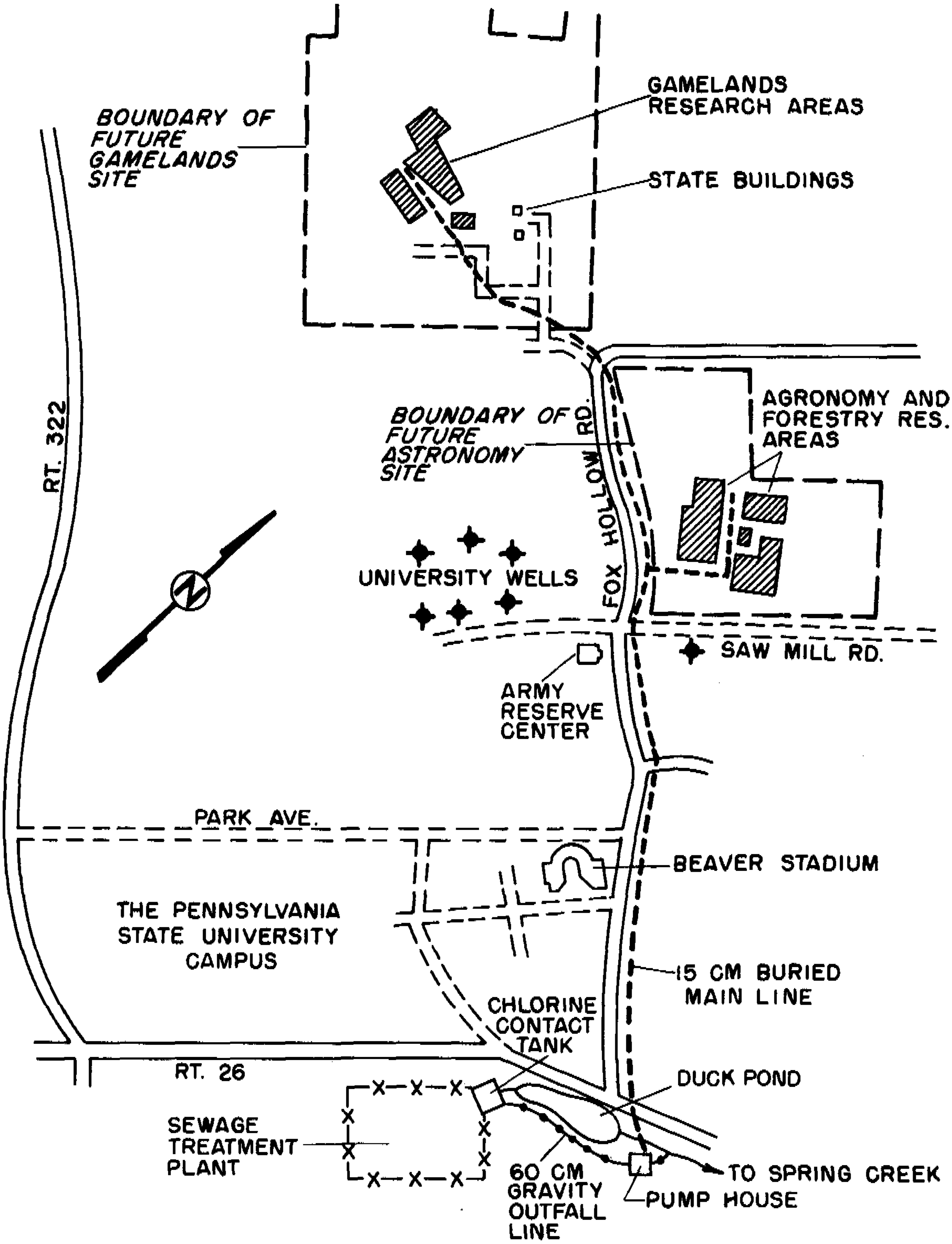


Figure 1. Location of Penn State's Wastewater Research Facility

removal and effluent drainage from the main line when required.

One main branch line terminated at the Astronomy Site approximately 4 km from and 53 m higher than the pumping plant; the second branch line terminated at the Gamelands Site approximately 6.4 km from and 85 m higher than the pumping plant. The pump supplied approximately 22 ℓ/s at 482 kPa to the Astronomy Site or about 19 ℓ/s at 310 kPa to the high point in the Gamelands Site. The rate and pressure desired were regulated in each area by branch line valves which were manually operated.

All pipe beyond the buried distribution system was above ground and comprised the irrigation application systems. These surface systems were composed primarily of 10 and 12.5 cm aluminum main lines and 5 and 7.5 cm aluminum lateral lines, which distributed the effluent to regularly spaced sprinklers on 2.5 cm diameter risers. Various small experimental areas utilized galvanized steel and plastic pipe for comparative purposes.

By appropriate use of specific sprinklers, spacing of the sprinklers along the laterals, spacing of the laterals, and adjustment of pressure at each lateral, application rates of 0.42 or 0.63 cm per hour were provided at individual locations. For an application depth of 5 cm per week only 1.2 ha need to be irrigated each 8-hour period or 1.8 ha each 12-hour period to use 22 ℓ/s continuously. Each arrangement irrigated 3.6 ha each day and thus, the system covers 26 ha in a week. Most of the areas received 5 cm of effluent per week; however, various small experimental areas received 2.5, 10, or 15 cm per week.

A solid-set irrigation system was used in all areas. That is, main and lateral lines were not moved from position to position, as is frequently done in agricultural crop irrigation. This permanent arrangement was preferred since it was difficult to move pipe among the trees; it was nearly impossible to move pipe when it was covered with ice; and the laterals used during winter needed to be on a continuous grade to provide rapid draining during freezing weather.

Sprinkler heads in the agronomic crop areas were staggered, with a distance of 24 m between sprinklers along the laterals and 24 m between laterals. Each sprinkler emitted 1 ℓ/s for an application rate of 0.63 cm per hour. A portion of the reed canarygrass area had sprinklers placed on 24 x 30 m spacings and applied the effluent at 0.42 cm per hour. For open areas the wider spacing is preferred since it permits lower original system costs and more convenient farming operations.

Spacings of the sprinkler heads in wooded areas varied from 24 x 12 m to 24 x 24 m. The variations were due to physical area restraints, research needs, or merely to provide flexibility in the total system. The most desirable spacing for wooded areas seems to be 18 x 24 m, which gives a good compromise between uniformity of distribution and cost of original equipment.

SYSTEM DESIGN RESEARCH

The sprinkler development and hydrologic research summarized in this section is from six Department of Agricultural Engineering theses and is discussed more fully in Reference 1.

Successful operation of sprinkler irrigation systems at below freezing temperatures where the distributed material is high in pollution potential requires equipment that provides uniform application across the area and reliable performance to prevent freeze-ups. Parmele's efforts were directed at distribution techniques involving no mechanical action except for tests conducted with commercial sprinklers. The perforated pipe, trickle from the open end of vertical pipes, and surface infiltration trench systems were deemed unacceptable due to poor distribution. Ice buildup on the turning mechanisms of the two-nozzle commercial sprinklers tested made their operation unsatisfactory at below freezing temperatures.

Givens developed an inverted cone stationary-deflector sprinkler which was nearly 100 percent reliable in operation. However, poor distribution made it unacceptable since most of the discharge landed in a narrow width circle concentric with the sprinkler location. Tests with modified and standard commercially available sprinklers produced results similar to those of Parmele.

Brown modified the stationary-deflector sprinkler developed by Givens by adding curved grooves to induce spinning of the inverted cone. Frequent failure of the cones to spin plus nonuniform distribution led to their abandonment.

Bodman's approach was directed at

achieving uniform distribution over a series of several successive operation cycles. Sprinklers with indexed-positioning mechanisms were developed in an effort to improve upon: reliability, penetration of a frozen surface (Figure 2), and distribution as compared to previously tested equipment. Throughout the development process, emphasis was placed on achieving the desired performance while minimizing damage to trees associated with ice buildup. While each of the sprinklers tested satisfied one or more of the operation parameters, none of them satisfied all of the requirements simultaneously (Reference 3).

All units tested were checked for uniformity of distribution by operating them in a prepared test site equipped with a grid of collection containers. Data collected were subjected to analysis by computer to yield a Christiansen Uniformity Coefficient (CUC) and a Coefficient of Variation (CV). Of the five sprinkler designs tested, four provided CUC and CV values of 85.5 or higher. A value of 75 was considered acceptable for both indices.

Ice buildup on trees was estimated for each type sprinkler following each run. Maximum and average heights of ice were estimated with values ranging up to 5 and 4 m, respectively. These observations led to the conclusion that within wooded areas, sprinklers with low trajectory angles are preferred due to less ice buildup in the trees.

Research to date indicates that equipment is not available to provide reliable operation for averting freeze-ups, to give uniform distribution which helps reduce the potential buildup of nutrients to toxic levels and enhances renovation, and to provide good penetration which helps minimize runoff. A compromise among the three factors is necessary when selecting from presently available equipment. Research, however, does indicate that single nozzle sprinklers are preferred over multi-nozzle units since less ice buildup occurs on the sprinklers in winter and fewer plugging problems are encountered throughout the year because of the larger single orifice.

Two hydrologic investigations were conducted--one by Rebuck on a Morrison sandy loam soil at the Gamelands Site and the other by Jarrett on a Hublersburg clay loam area at the Astronomy Site. The Gamelands Site study evaluated hydrologic aspects under a sequence of irrigation loading depths during spring, summer, and fall climatic conditions. This study verified that under Pennsylvania's stratified soils a large portion of runoff induced by irrigation could more properly be termed interflow. This is advantageous in wastewater irrigation since phosphorus concentrations in interflow are substantially lower than in direct surface runoff.

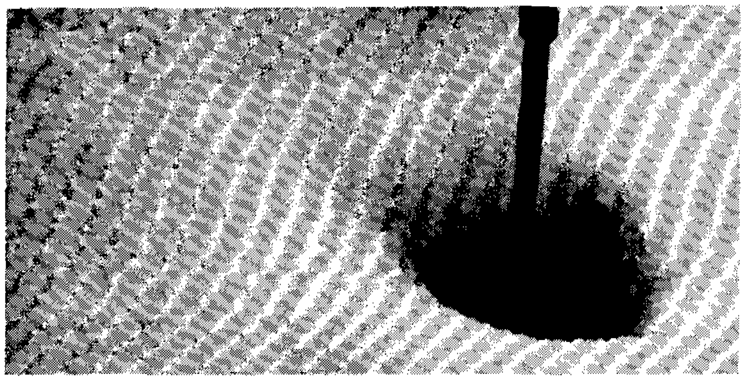
The study at the Astronomy Site was conducted to determine the difference between the watershed's responses with respect to runoff peak rates, total runoff volumes, and chemical quality of the runoff under two irrigation procedures. The one procedure applied effluent to the entire watershed simultaneously to maximize its hydrologic response, while the second procedure applied effluent during a sequence of times to minimize its hydrologic responses. Results indicated that during periods when infiltration and percolation rates were not decreased by soil frost, sequencing the application of irrigation was advantageous. During periods when soil frost essentially prevented infiltration and the surface was coated with ice, however, sequencing had no beneficial effect on total runoff but did result in a generally lower content of phosphorus and nitrogen in the runoff.

EXPANDED SYSTEM - FUTURE

Penn State's proposed wastewater renovation facility is being designed to handle 175 ℓ/s. Phase I of this project has already been installed and includes a 1893 m^3 surge tank, two 0.26 MW pumping plants, and a 4.8 km long 45 cm diameter ductile iron main. The pumping plants are housed in a new structure located adjacent to the present research pumphouse and the 45 cm main, running only to the edge of the Gamelands Site, parallels the 15 cm research main, Figure 1.

The design of wastewater irrigation systems is still somewhat of an art rather than an exact science. One can begin with a 600-page book and by an inductive process develop a system, or he can start with a six-page leaflet and deductively design a system from five major variables which are adjusted by experience. Penn State's expanded wastewater system (Figures 3 & 4) is being developed from these variables with adjustments based on experience gained in the design of over two dozen wastewater systems in a half-dozen

a. Little Penetration (Note smooth ice layer)



b. Moderate Penetration (Note honeycombed effect)

c. Severe Penetration (Note complete melt-through)

Figure 2. Surface Appearance with Relative Degrees of Penetration

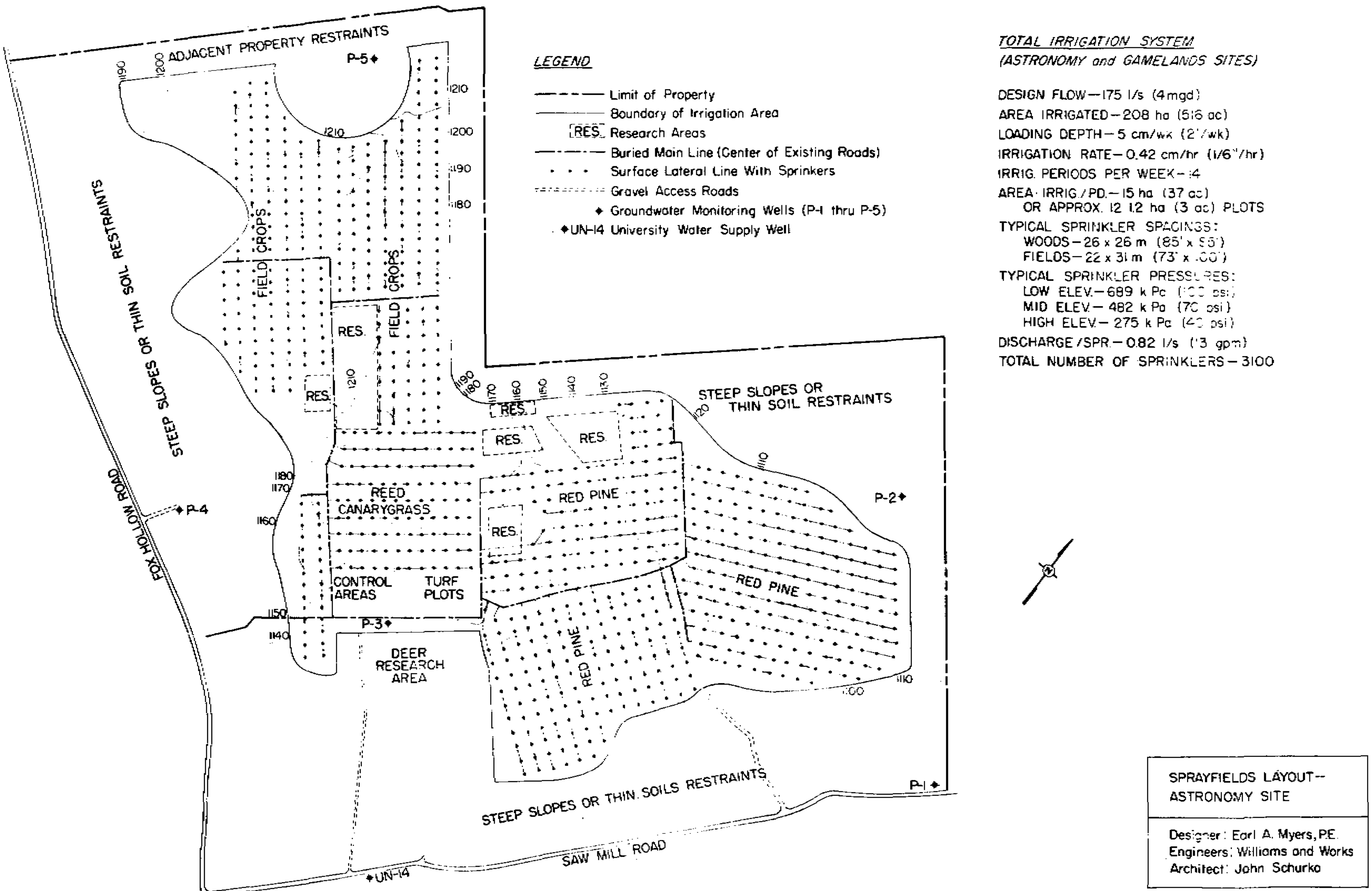


Figure 3. Astronomy Site--Future Irrigation System

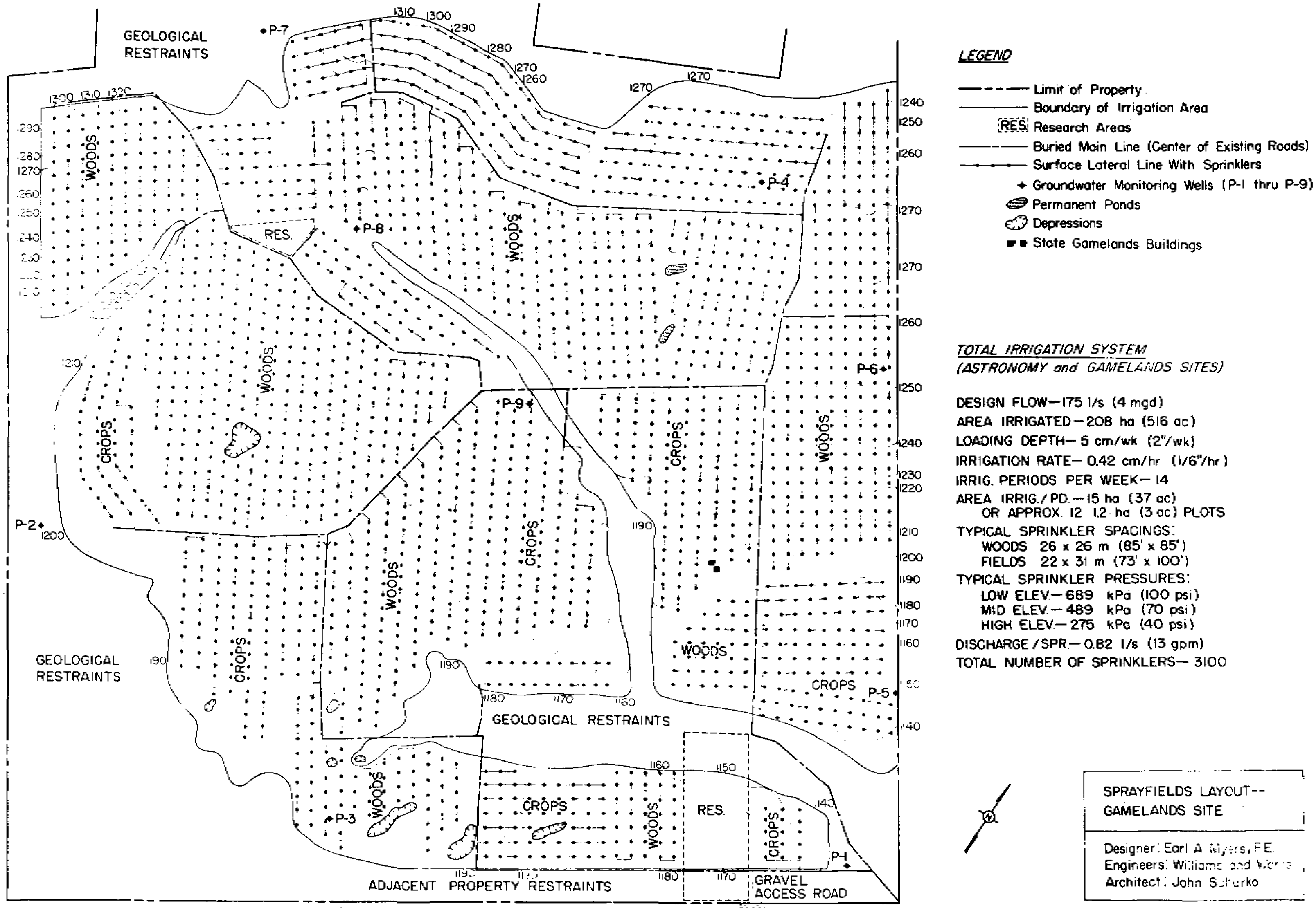


Figure 4. Gamelands Site--Future Irrigation System

states (References 2 and 4). Daily flow from the treatment plant in conjunction with the weekly loading depth determines the amount of irrigation area required. The application rate in conjunction with the loading depth per week establishes the number of separate fields that can be irrigated each week. Sprinkler spacing in conjunction with the rate of application establishes the required discharge rate of each sprinkler, whereas the sprinkler spacing and the area per field establishes the number of sprinklers which should operate at any one time. The fifth variable is the operating pressure at the sprinkler nozzle. This pressure in conjunction with the required discharge rate of the sprinkler determines the diameter of nozzle hole that is required.

Effluent flow rate from the University's wastewater treatment plant varies from approximately 39 ℓ/s to 175 ℓ/s but flows are mostly in the 86 to 163 ℓ/s range; thus, 175 ℓ/s is being used as the basis for design. With a 5 cm weekly loading depth, 208 ha of land are required. A 5 cm per week loading depth is compatible with the renovation and hydrologic capabilities of the site relative to soil type and depth, depth to the groundwater table, and desire for recharge.

Two different fields per day or 14 fields per week can be irrigated when using an application rate of 0.42 cm/hr and a 5 cm loading depth per week. Thus 15 ha need to be under irrigation at all times to handle 175 ℓ/s. This 15 ha requirement is being divided into 1.2 ha plots to obtain hydraulic and hydrologic balance, to permit use of smaller pipe, and to provide a system with nearly uniform pressure requirements among the 14 time periods per week.

Typical layout of the 1.2 ha plots involves 18 sprinklers spaced 26 x 26 m in wooded areas and 22 x 31 m in open areas. The wider spacing in the open area provides lower original costs and easier farming conditions, while the arrangement in the woods provides maximum spacing for which adequate distribution is still attainable. A sprinkler discharge of 0.82 ℓ/s gives an application rate of 0.42 cm/hr on each area, since the areas are equal in size. Also, each area receives its 5 cm of wastewater in 12 hours, permitting a uniform time procedure each week for changing valves manually.

Average operating pressures are to be 275, 482, and 689 kPa at high, medium, and low elevations, respectively. To emit 0.82 ℓ/s from each sprinkler the nozzle diameters are reduced for increasing pressures to 0.68, 0.60, and 0.52 cm, respectively. Using this range of pressure minimizes the need for reducing valves and also reduces the energy requirement of the pumps by one stage or 37 kW. At the cost of 1.5 cents/kWh, this represents an energy savings of approximately $5,000 per year of use for each pump.

Additional design factors relate to winter irrigation, monitoring, and special system arrangements for ease of year-round operation and management. Special deflectors and shields will be used to prevent icing of valves and lateral drains. All-terrain vehicles will be needed in winter to move easily and safely over ice covered areas, to reduce walking distances and time required for lateral draining and maintenance, and to provide ready access to any part of the system without excessive road maintenance or snow removal. A network of wells, some of which are shown on Figures 3 and 4, will be used to monitor subsurface water movement and quality in each dominent direction of water egress. Dedicated management, however, will still be required to operate effectively a system of this magnitude on a year-round basis.

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DESIGN CRITERIA

WETLANDS WASTEWATER TREATMENT SYSTEMS

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ABSTRACT

On the basis of contaminant attenuations in artificial wetlands when used in reclaiming water from various types of sewage, it is desirable to examine these systems in engineering terms and to consider their use in the reclamation of wastewaters in general.

Judging from the demonstrated performance of two wetlands semi-works at Brookhaven National Laboratory, it appears that these systems are technically superior to conventional secondary treatment plants and can be the equivalent of AWT plants. It is estimated that both construction and operating costs of artificial wetlands are less than these costs for conventional treatment plants in the 38 to 3800 m^3/d flow range. Energy consumed in artificial wetlands construction is estimated to be a small fraction of the total energy required to build a modern conventional plant.

Design constraints which are necessary to gain the technical, dollar and energy advantages of artificial wetlands for sewage reclamation are presented. Extensions of the prototype designs to systems for the reclamation of other point and non-point source wastewaters are discussed.

DESCRIPTION

Two artificial wetlands sewage reclamation semi-works are in operation at Brookhaven National Laboratory, Upton, L.I., N.Y. (BNL). Their design and history have been described previously [1-3] in detail but, in summary, they are as follows. The Meadow/Marsh/Pond (M/M/P) has two Reed Canary Grass meadows of .08ha each, followed by a .08ha *Typha* marsh followed by a .08ha pond about 1.5m deep. The Marsh/Pond (M/P) is similar but omits the meadows. Both are underlain by a barrier membrane. Both are fed screened, comminuted, aerated, raw, unsedimented sewage. Both polish their effluent by surface application to a wooded, sandy-soil aquifer recharge area. Schematics of both are shown in Fig. 1.

These wetlands began operation in Spring of 1973. Since Spring 1975,both have been in continuous, on-line operation, reclaiming water from blends of septage and sewage, septage or sewage alone, sludge from septage lagoons and from conventional primary plants and, raw solids from the BNL sewage system clarifier. The wetlands operate throughout the year, without producing odor or vermin, with no regular maintenance, at various application rates between 420 and 835 $m^3/ha\text{-}d$. Both ponds are stocked with fish and freshwater clams. Wildlife is attracted to and live in these natural settings.

No permanent sediment build-up has occurred over the full five year operation in either system despite the fact that no sludge has been removed from the sewage at any point. Hay is harvested from the meadows at the rate of 18Mg/ha-y dry weight. Duckweed is harvested from the M/M/P ponds. Both crops are fed to horses. The M/P is not cropped. The M/M/P requires twice the land needed for the M/P and appears to do a slightly better contaminant removal job. However, the renovative capacity of the M/M/P is

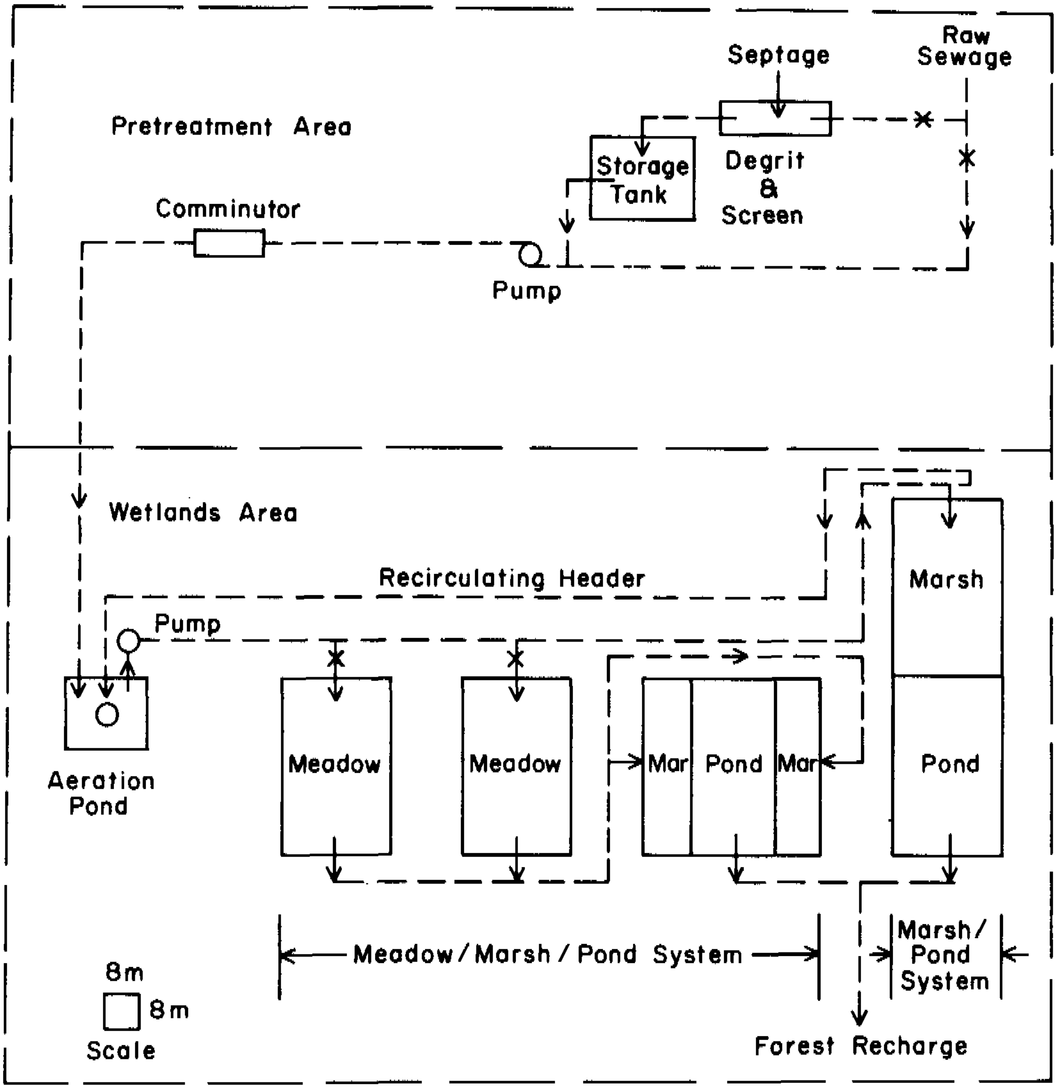

Fig. 1 Brookhaven National Laboratory Schematic Artificial Wetlands Treatment Systems

not enough better to justify its use unless land is not dear and a crop is desired. On an engineering basis, the M/P is more efficient so that where possible it is recommended and has been the model for the studies in this paper.

PERFORMANCE

Two Data Reports covering 13 months operation of both wetlands have been published [4-5]. Subsequent analyses continue to be made for future publication but the data in Table 1 are taken from reports which are now available. Table 1 shows that average effluent concentrations from both ponds are high in Total Suspended Solids and in Total Coliform and, from the M/P, in Turbidity. These contaminant levels are inevitable in discharges from natural ponds and, judging by the low FC/FS ratios found, are due to algae and detritus, not the carryover of sewage forms. Because of these substances, however, the direct reuse of pond effluent is not recommended before they are filtered out. For groundwater recharge at BNL it has been found [2], as reported by others [6], that passage of pond effluent through 2 to 3 m of vegetation covered, sandy soil is sufficient both to filter and to disinfect the percolate to U.S.P.H.S drinking water standards.

Both BNL pond effluents are high in Iron and Manganese. This is due to the content of these elements in the local groundwater. It is common for producing water wells in the BNL area regularly to yield 3 to 4 mg/ℓ of Iron and not uncommon for well water to contain 16-20mg/ℓ.

Table 1. Characteristics of Influent Sewage and Pond Effluents for 13 Month Study Period 8/75 - 8/76 (in mg/l except for pH and as noted)

Contaminant	INFLUENTS		M/P EFFLUENT		M/M/P EFFLUENT	
	Mean	Max.	Mean	Max.	Mean	Max.
Total Solids	562	5,300	206	300	180	365
Total Volatile Solids	335	3,640	102	142	87	241
Total Suspended Solids	353	4,300	43	100	39	104
Total Volatile Suspended Solids	235	3,050	35	76	28	70
Total Dissolved Solids	208	1,000	163	242	140	308
Biochemical Oxygen Demand $(BOD)_5$	170	2,700	19	46	13	62
Chemical Oxygen Demand	495	7,900	58	120	48	120
Total Nitrogen (liquid + solid)	25	91	9.5	18	5.2	15.6
Kjeldahl Nitrogen (liquid + solid)	19.7	88	6.8	14	3.7	12.6
Ammonia Nitrogen (liquid)	8.4	18	3.5	11.5	1.2	6.8
$(NO_2 + NO_3)$-N (liquid)	5.5	17	2.6	6.7	1.5	3.2
Total Phosphorous (liquid + solid)	7.2	27.7	2.1	4	1.6	5.3
Orthophosphate-P (liquid)	4.8	22	1.3	3	1.1	4.6
Total Coliform (#/100ml)*	59.6	20,000	2	234	2.25	127
Fecal Coliform (#/100ml)*	1.6	1,000	.05	10.6	.03	4.5
pH	6.8	8.9	7.4	9.1	6.9	9.2
Turbidity (J.U.)	43	400	8.5	74	4.8	71
Water Temperature (°C)	10	22	11	24	10.6	24
Specific Conduc. (μ mhos)	464	6,600	262	340	224	315
MBAS (ABS)	.3	3	.24	1.4	.3	2.9
Calcium	20	72	14	26	14	47
Chloride	35	110	30	46	29	85
Chromium	.05	.5	.01	.03	.02	.3
Copper	.7	3.2	.03	.14	.04	.2
Fluoride	.5	1	.4	.6	.3	.5
Iron	3.6	20	1.2	5.5	1.5	3.9
Magnesium	4.3	8.5	3.6	6.3	3.3	4.4
Manganese	.14	.75	.1	.3	.1	.5
Potassium	5	11	4	9	3.1	11
Sodium	26	52	25	52	22.8	30
Zinc	1.3	4	.2	.6	.2	.7

*Multiply value by 10^3: Means are geometric.

PLANT COMPARISONS

With the exceptions noted above, both ponds produce an effluent which is generally better than that to be expected as a yearly average from conventional secondary sewage plants [3,7]. With the addition of polishing by vegetated soil filtration a wetlands + recharge sequence can be compared favorably with AWT plant performance. In summary, the recharge quality of BNL wetlands recharge, at the water table, is expected to equal that reported for other land use systems, namely, in mg/ℓ: BOD_5 = 1-2, TSS = 1-2, NH_3-N = .5-1, total N = 2-4 and total P = .1-.5 [8].

A comparison of ΣN removed by an artificial M/P followed by slow rate infiltration [8], with ΣN removal by a contemporary AWT test plant is shown in Fig. 2. The reported [9] performance of the Blue Plains 380 m^3/d, three-stage, suspended growth pilot plant is plotted against ΣN removals measured in BNL models and, against calculated removals to be expected by a 950 m^3/d on-line M/P recharge design commissioned by the Town of Brookhaven [10]. The plots indicate that about 9% of influent ΣN will remain in the AWT effluent after conventional sand filtration and predict 12% in the wetlands effluent after forest filtration. At the given influent ΣN strength of 25 mg/ℓ this means that the AWT process will reduce effluent discharge concentration to 2.25 mg/ℓ and the projected wetlands will reduce it to 3.0 mg/ℓ.

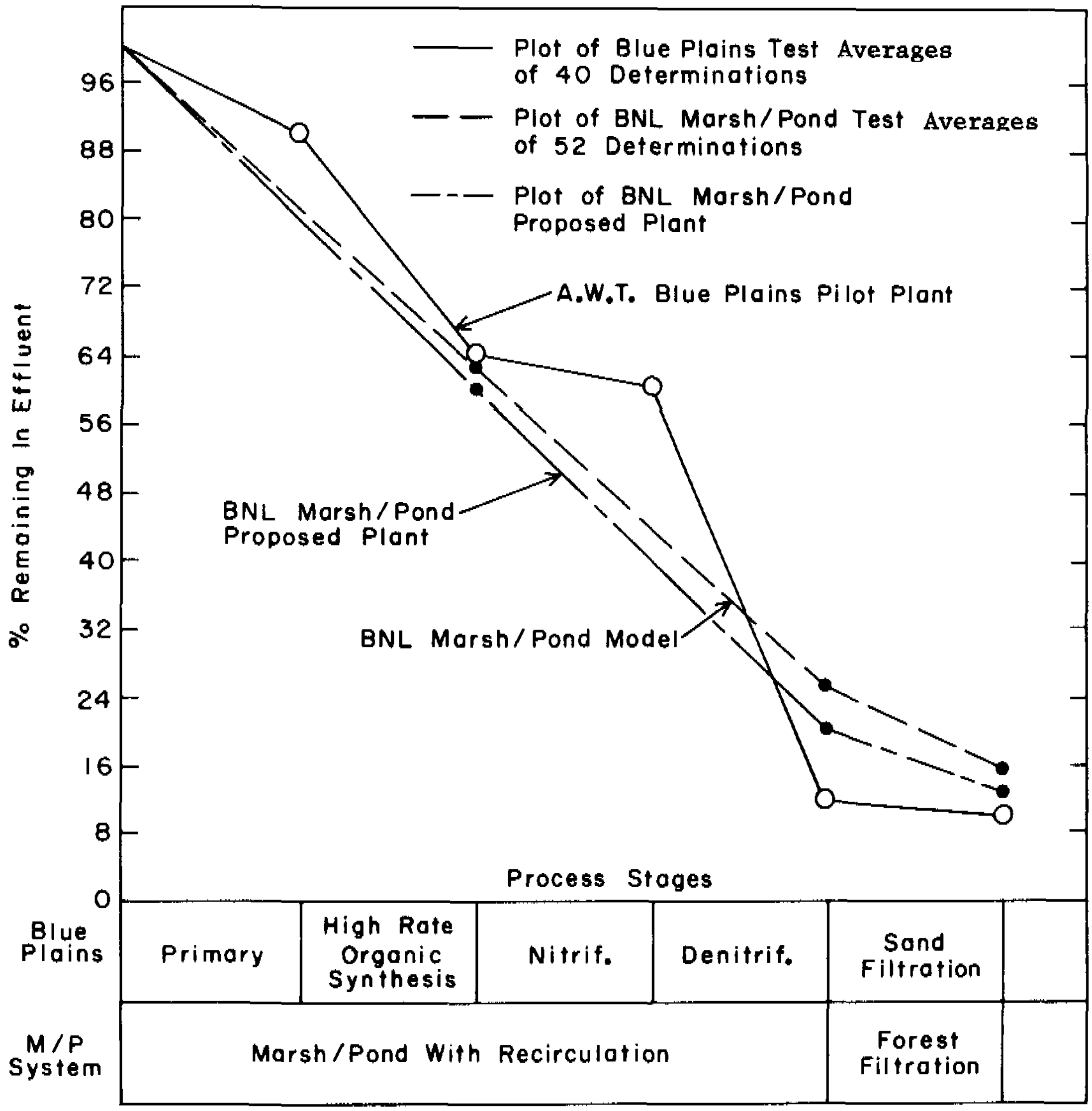

Fig. 2 Comparison of Total Nitrogen Removals

COSTS

Having found that the renovative performance of artificial wetlands followed by vegetated recharge is comparable to that of conventional AWT plants, it is desirable to examine their competitive cost position. It was found that denitrifying artificial wetlands can be built at less cost over their commercial range than so-called package conventional plants providing nitrification only. Fig. 3 shows the relationships.

This plot is the result of a study [11] of the construction cost of artificial wetlands as plants for small communities, compared with these costs for two types of package plant [12] and other small community alternatives.

It is interesting to note that the M/P [13] would be less costly than all alternatives considered by one community with an estimated 53 m^3/d load except for individual on-site disposal systems. Where these are acceptable, a M/P cannot compete with their low first cost below 130 m^3/d. However, package extended aeration plants cannot compete with them below 400 m^3/d and rotating biological media plants below 2,000 m^3/d.

Operating and maintenance costs for M/P plants will benefit from lower debt service expense because of higher conventional plant construction costs. The cost for power is the principal operating expense and is estimated at 1 h.p. per 40 m^3 of throughput. Manpower need not average over 1 M.Y. per 4ha of combined M/P which is the size required for

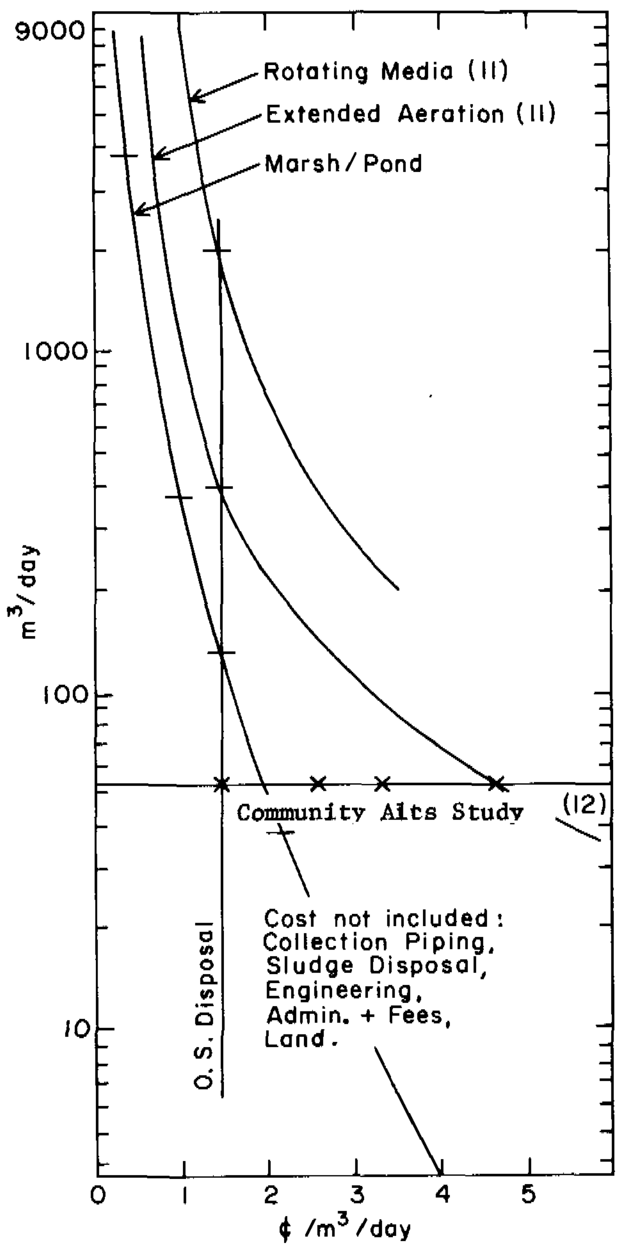

Fig. 3 Plant Construction Cost Comparison

a design throughput of 1900 m^3/d.

ENERGY

In view of the growing need to conserve energy where possible, a study has been made [14] of the total energy required for a conventional AWT plant for use by small communities. An energy comparison was made between a 1900 m^3/d M/P with vegetated soil filtration and the same size conventional activated sludge plant having nutrient removal and sludge dewatering [7]. Both designs are assumed to accept the same influent sewage strength and produce the same effluent concentrations. The characteristics and energy requirements estimated [14] for the two designs are shown in Tables 2 and 3. It is startling to note from these Tables that to manufacture all hardware and to construct the M/P, even with a 100% contingency factor, is estimated to require about 16% of the energy expenditure necessary to manufacture and build the conventional activated sludge plant.

Table 2. System Characteristics

		M/P	AWT
Land Required	ha	5.57	2.63
# Operators	M.Y.	1	4
Retention Time	d	12-25	<1
Annual Chemicals	t	---	222
Annual Sludge	t	---	250
Recharge	m^3/d	1235	1900

Table 3. Energy Requirements

To Construct-Kwhx10^{-3}	864	5310
To Operate-Kwhx10^{-3}/y	788	2666

Operating energy for the M/P will be one third of the energy that it takes to run the activated sludge plant. The M/P cost saving is due to its 50% lower power load demand, no chemical use and no sludge handling, drying or removal effort.

DESIGN

To accomplish these technical, economic and energy conservation advantages over conventional sewage treatment plants, specific design and operating constraints are necessary for both types of wetland. Among these are:

Incoming sewage must be aerated to prevent odor if there is the possibility that it will be septic;

Each marsh, meadow and pond must be sized to prevent flood flows from developing a greater-than-settling velocity and to allow design flows no less than 12-15 days retention in the system;

Artificial wetlands should be stocked with plant and animal populations which are indigenous to the area so that it is certain they do grow there naturally and will not have to be cultured in order to thrive;

The wetlands are marshes, meadows and ponds in configurations and sequences which are designed specifically for the terrain, the climate and the wastewater to be reclaimed;

Each component must be contained by an impermeable barrier to prevent leakage before the influent has run the full treatment course;

Each component must have freeboard so that a flood will be contained or, so ice can rise or snow accumulate above the flow to let the processes continue underneath;

A recharge area must be planned so that natural, vegetated soil will do the final polishing and disinfection without ponding whether the reclaimed water is to go to surface streams or lakes or, is to percolate to groundwater;

Incoming sewage must be screened to remove non-degradables and it must be comminuted if raw fecal matter is to be processed;

Influent sewage annual average contaminant strengths should not exceed those shown in the first column of Table 1 if the average effluent contaminant strengths shown are to be met by either artificial wetlands system.

If these design constraints are met, there appears to be no reason why BNL model wetlands cannot be built elsewhere to reclaim water from any form of sewage.

OTHER APPLICATIONS

In view of the cost and energy effectiveness of artificial wetlands in treating sewage, the BNL models have been investigated as appropriate technology for use in the reclamation of other wastewaters. In addition to authorizing the construction of a M/P to renovate the effluent from their septage, sludge and laundry waste lagoon, the Town of Brookhaven is also considering a BNL proposed [15] recirculation facility to pretreat landfill leachate to a low enough strength for renovation in a M/P wetlands.

In concert with the Town, BNL is developing a wetlands configuration best suited to the removal of contaminants from highway and general runoff ahead of its discharge into a recreational lake. This application is endorsed by the regional 208 study [16] of wastewater management which recommends M/P construction and evaluation with particular emphasis on heavy metal and organic chemical removals. Such a use would be of great value in protecting the Long Island sole source aquifer. Because of the reported [17] high level of both toxic chemicals and heavy metals in area runoff in general and highway runoff in particular, direct use of a Brookhaven model wetlands cannot yet be recommended. Care in the design of the wetlands is necessary to obviate plant and animal toxicity and the initial installation can be considered only experimental since BNL has not worked with high levels of these contaminants.

At the request of The Coordinating Council on the Restoration of the Kissimmee River Valley, BNL recently evaluated the use of wetlands in the control of Florida feed-lot and farm runoff. It was concluded that by converting existing pasture to function as the meadow [18], by diversifying the marshes [19], and by digging shallow ponds M/M/Ps could effectively control surface water contamination by farm runoff. It was felt that pretreatment by regular flushing would be necessary to assure flow and that harvest of aquatic weeds, such as Hyacinth, would be a necessary post-treatment to gain the desired high nutrient removal. A prototype wetland of the Committee's design to control this runoff is now under construction by that group.

Proposals for use elsewhere of engineered artificial wetlands are being considered by several state and local jurisdictions. Others are under discussion with the private sector where it is anticipated these low cost, energy conservative schemes will find acceptance as appropriate technology in the control and reclamation of many wastewaters.

ACKNOWLEDGMENT

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DESIGN CRITERIA

TREATMENT OF WASTE FROM A CONFINED HOG FEEDING UNIT BY USING ARTIFICIAL MARSHES

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ABSTRACT

A 1-year study has been completed to evaluate use of artificial marshes as a treatment system for waste from confined hog feeding operations. A 3 x 2 factorial experimental design was employed to evaluate the performance of marshes planted with one of three emergent species (Sparganium eurycarpum, Phragmites australis, Typha glauca) and receiving three different loadings of hog waste. Each treatment was replicated twice. For each marsh, a complete hydrological and nutrient (N, P) budget was determined. Data on COD and solids were also collected. The growth of the plants, particularly rates of vegetative reproduction, were also monitored regularly throughout the growing season to examine the effect of sludge accumulation on plant vigor. Because the odor associated with present animal waste handling systems has been a major problem, the most important feature of artificial marshes treating hog waste is that they seem not to have developed any odors during their first year of operation. Emergent plants have a system of internal air spaces that allows oxygen from the leaves to diffuse into the rhizomes and roots. Enough oxygen seems to be diffusing out of the roots and into the accumulating sludge to prevent it from becoming completely anaerobic.

During the summer of 1977, the marshes produced a superior effluent to that of an anaerobic lagoon. Up to 70% of the TVS and 50% of the COD was removed by the marshes, largely through mechanical filtration. Approximately 17% of the N and P was removed, mostly by uptake into the plants.

Design criteria for these marshes and suggested modifications are presented.

INTRODUCTION

A form of biological sewage treatment has been developed in Europe (Seidel, 1973; deJong et al, 1977) that combines the action of microorganisms and higher plants. A trench or series of trenches is prepared and filled with a coarse, porous medium, such as crushed rock. These trenches are then planted with selected species of emergent aquatic plants to create an artificial marsh and loaded with raw sewage. According to Seidel (1973), the effluent from such artificial marshes equals or exceeds in quality the effluent from conventional sewage-treatment plants. The major interest in using artificial marshes for treating human sewage has arisen because the systems can be used seasonally (e.g., for summer resorts) and are cheaper and easier to manage than conventional municipal systems.

As far as could be determined, artificial marshes have not been used to treat livestock manures. At present, anaerobic lagoons are the most economical method for processing manure flushed from confined-animal units (Smith and Hazen, 1976). Although lagoons liquefy manure very effectively, they also produce a distinct and

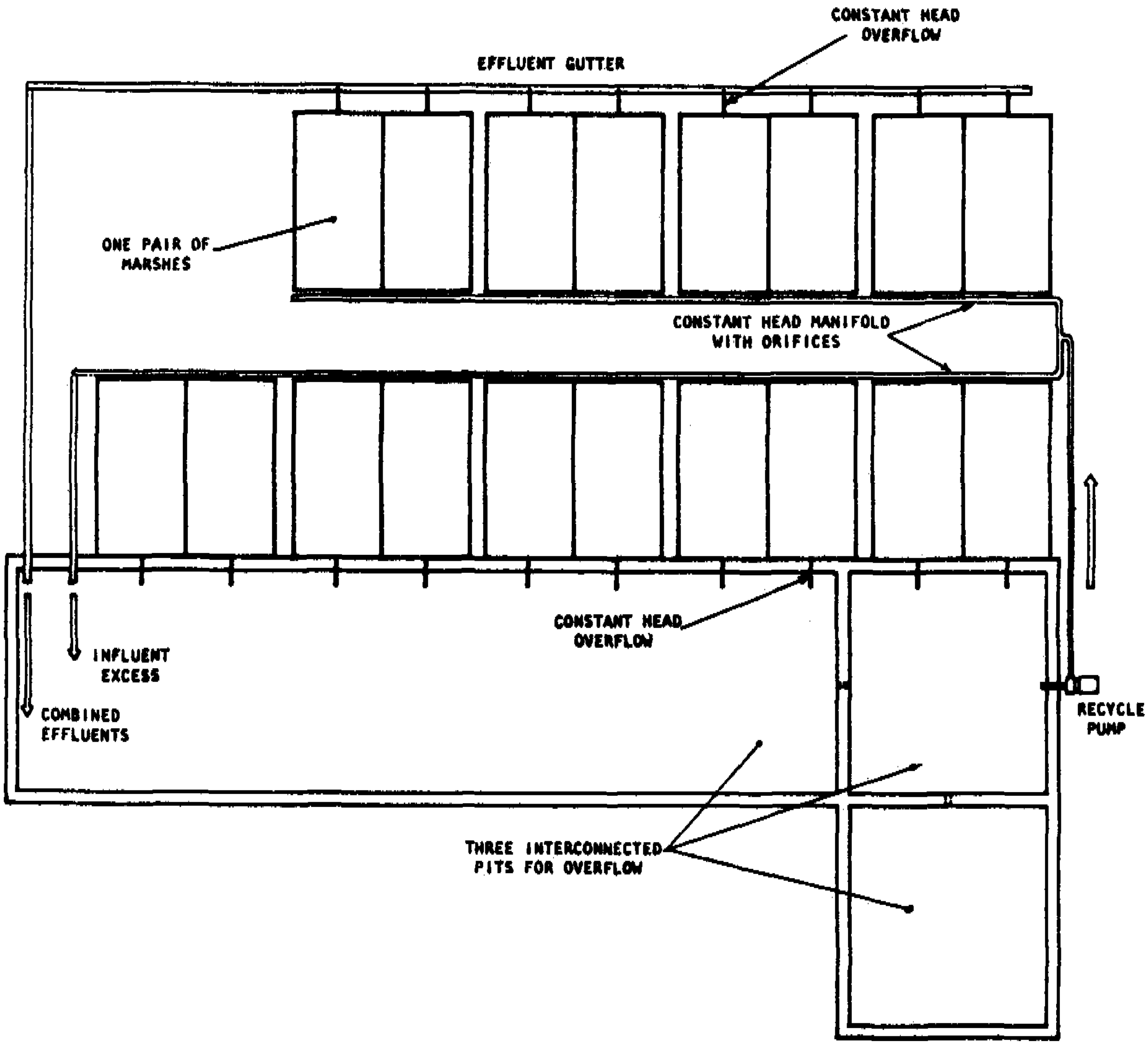

Figure 1. Plan view of 18 small marshes used to treat swine manure at Iowa State University

disagreeable odor. Therefore, we have been examining alternative treatment systems to replace anaerobic lagoons. Marsh systems seem to be an attractive alternative; the gravel matrix would serve as an anaerobic zone, but the plants should keep the upper surface sufficiently aerobic to detoxify obnoxious anaerobic gases rising from the lower levels. Such marsh systems should, therefore, produce an inoffensive effluent suitable for recycle cleaning of the confinement house.

The development of an artificial marsh system that is an alternative to the anaerobic lagoon requires that the following criteria are satisfied:

(1) the vegetative reproduction of plants should not be inhibited.
(2) the quality of the effluent should equal or be superior to that of the anaerobic lagoon;
and (3) no offensive odors should be produced.

The objective of our study was to determine if these criteria could be met by artificial marshes.

MATERIALS AND METHODS

Three species of emergent aquatic plants were chosen for the study: *Phragmites australis* (common reed), *Typha* x *glauca* (common cattail), and *Sparganium eurycarpum* (burr reed). All three are native to Iowa and thrive in submerged, highly organic environments. Rhizomes of the plants, collected from a natural marsh, were planted in early June 1977, in 18 wooden boxes (1.2m x 2.4m) adjacent to a 700-head swine finishing house near Ames, Iowa. Each marsh box was lined with a 0.25mm polyethylene membrane and filled with gravel to a depth of 0.46m.

Three loading rates were used in a factorial design with two replicates. The highest represented the amount of manure produced by a 60kg pig (the average weight in a finishing house) spread over $8.1m^2$ and would be intermediate in area between an anaerobic and aerobic lagoon. Manure was taken from the swine house effluent sewer line and allowed to settle into solid and liquid fractions; the high treatment consisted of 2L of each. The two lower rates were 2L and 0.2L of liquid alone, containing both urine and the flushing medium of recycled lagoon water. The marshes were loaded with a manure treatment daily except Saturday and Sunday.

Each marsh box was fitted with an adjustable overflow pipe; the effluents from each pipe drained into a concrete-lined pit (Figure 1). This pit was initially filled with water from the adjacent anaerobic lagoon presently used to treat the manure from the finishing house. No further additions of water were made to the system except from manure loadings or from precipitation.

Controlled recirculation of the pit contents was achieved by means of orifices on pump-fed manifolds. The flow rate at each orifice was 2.1 L/min. and the recirculation pump operated four times daily for 30 minutes each time. By noting the time between pump start up and the first signs of overflow, the amount of evapotranspiration could be estimated. This estimate was made for the daylight hours and could not account for nighttime losses or gains. This method was not regarded as very accurate so a linear regression of our measurements *vs*. pan-evaporation was calculated. All water-loss data used in developing mass balances were calculated from the regression equation by using pan-evaporation and rainfall data gathered at a University farm 8km to the West.

The plants were examined weekly and new shoots were counted. Shoot heights were recorded at 3-week intervals. Biomass was calculated from a linear regression model, using shoot height, leaf width, and number of leaves, originally formulated from plants collected in a natural marsh. Weekly measurements of the concentrations of total volatile solids (TVS), chemical oxygen demand (COD), Kjedahl nitrogen (total-N), ammonia nitrogen, total phosphate (total-P), ortho phosphate were made of the marsh effluents, the solid and liquid portions of the manure, and the recirculation water.

The system went into operation the week beginning 1 August 1977. Results were collected until 7 November 1977, by which time the aerial portions of the plants had been killed by frost. The boxes were then drained. Because of the small scale of the artificial marshes, no attempt was made to operate during the winter.

Table 1. Total loading of TVS, COD, total-N, and total-P (kg/ha) over the 14 weeks of the study in the three treatments.

Component	TREATMENT High	Medium	Low
TVS	122100.	40367.	37900.
COD	76900.	34967.	29467.
total-N	5770.	4546.	4260.
total-P	2920.	2227.	2107.

Table 2. Above and below ground biomass (kg/ha) of emergent plants in the artificial marshes.

Species	Treatment	Above-ground* Biomass	Below-ground** Biomass
Typha	High	15545.	11885.
	Medium	16100.	12840.
	Low	15930.	18101.
Sparganium	High	14810.	7460.
	Medium	13410.	7330.
	Low	14295.	5580.
Phragmites	High	6162.	6310.
	Medium	7789.	4945.
	Low	7319.	5530.

Note: ANOVA F-test indicated no difference between treatments within species in above- and below-ground biomass at the 0.05 level of significance.

*Measured at peak standing crop; 10 September 1977.

**Seasonal accumulation measured 4 April 1978.

RESULTS

Table 1 presents the characteristics of the three loading rates used. We found that the loadings actually applied were higher than those intended. The recirculating pit contents (initially charged with lagoon liquid) accounted for more than 70% of the total loading for all parameters in both medium and low treatments. As a result, differences between them were obscured. Despite the heavy nutrient loads, the plants grew vigorously and showed no symptoms of stress. Although solids accumulated around the bases of the plants, they did not seem to hinder vegetative reproduction. Biomass (Table 2) indicated no statistical difference between treatments within species (ANOVA, $F_{2,3} = 0.83$; $F_{2,3}$ at $0.05\% = 9.6$). The obvious differences in biomass between species were the result of differences in the morphology of each. In part the low above-ground biomass of Phragmites resulted from the plant initially putting most of its production into below-ground rhizomes. This was supported by the relatively large below-ground accumulation in comparison to the above-ground peak biomass values (Table 2).

Survival of all species was generally high, but some injury to young Sparganium shoots was evident at the highest loading rate. Mechanical injury (crushing) occurred during daily application of the solid manure. Yet since no statistical difference between biomass in the high and low treatments was demonstrated, such injury seems minor. Underground parts of all species survived the winter and were in good condition in late April.

The greatest removal percentages are found in the highest treatments where up to 70% of the TVS and 50% of the COD was removed (Table 3). The significant removal of TVS and COD may be due to both filtration and biological activity. It seems likely that the porous support (gravel) was behaving as an anaerobic filter (see Young and McCarty, 1969) and mechanical filtration is primarily responsible for the reductions of TVS and COD. This is especially true for the highest loading since most of the TVS and COD was in the solid fraction that only the highest treatment received (Table 1). There was some biological activity, however, as evidenced by the removal percentages found in the low loadings of essentially liquid organics.

Removal of nitrogen by the marshes never exceeded 15% (Table 3) and most of this nitrogen was taken up by the plants. For example, 177 grams of nitrogen (12% of the total) was removed by the Typha medium treatment marsh, of which 150 grams can be accounted for in the leaf material alone. The leaf nitrogen concentration of each species was similar in all three treatments, and because the biomass was also similar for a given species (Table 2), the plants should remove a comparable amount of nitrogen from each treatment. Any additional removal such as that found in the highest treatments was the result primarily of mechanical filtration. Negative nitrogen budget values in Table 3 may be the result of errors in estimates used in our mass-balance calculations. They are, however, more likely to be the result of nitrogen fixation since errors in the hydrological budget should also have produced negative phosphorous budgets in the medium and low treatments, but did not (Table 3).

Phosphorous removal was in part through uptake into the plants, but filtration and sedimentation probably contributed significantly to total removal. In the Typha medium treatment, of the 93 grams removed (13% of the total), only approximately 15 grams of phosphorous were accounted for in leaf material.

Because the marsh system was being tested as an alternative to the anaerobic lagoon, a comparison of the effluent quality of both systems was made (Table 4). The existing anaerobic lagoon already produces an effluent of sufficient quality to recycle as a manure transport medium; any effluent produced by the marshes of equal or higher quality would also meet the criterion for recycling. The quality of the marsh effluent continually improved relative to the lagoon during the study. Although there was some dilution water present in the overflow holding pit before the start of recirculation in July, the trend of improvement cannot be explained by dilution alone. There is no doubt that during warm weather the marsh system produced a superior effluent to that produced by the anaerobic lagoon.

Table 3. Percentage reduction of TVS, COD, total-N, and total-P by artificial marshes during the study period.*

Species	Treatment	TVS	COD	total-N	total-P
Typha	High	69.1	48.4	14.8	19.9
	Medium	24.3	30.1	12.0	13.4
	Low	21.7	16.9	6.2	8.5
Sparganium	High	70.5	53.4	11.1	16.0
	Medium	25.1	28.7	-9.7	5.0
	Low	22.9	19.6	-12.3	5.7
Phragmites	High	69.0	48.4	14.9	18.5
	Medium	14.9	28.1	4.0	8.1
	Low	9.9	14.0	-0.9	7.2

*Average of 2 replicates.

Table 4. COD(mg/L) in an anaerobic lagoon and marsh-treated recirculation water.*

Week	Anaerobic Lagoon	Recirculation Water
09	1890.	471.
11	----	330.
13	2160.	241.
15	2330.	207.
17	2100.	159.
19	2040.	151.
21	2030.	159.

*Recirculation water taken initially from the lagoon in June.

DISCUSSION

During the summer, the artificial marshes did not develop an offensive odor. Comments by several visitors tended to reinforce our opinion that even the most heavily loaded marshes were without a pungent swine-manure odor. Although we found these results encouraging, we should stress that the results only cover warm-weather operation.

All three species tested thrived in all three treatments. Typha and Phragmites seemed to be the best species for artificial marshes since their young shoots were not crushed by manure solids. We believe that the marsh plants were responsible for the removal of nitrogen and some phosphorous. The removal of COD and TVS, however, seemed to be the result of mechanical filtration and microbial activity.

Aesthetically, we believe the marsh system has many advantages over the anaerobic lagoon. The question remains, does such a system have a chance of working during the 3-months of very cold weather expected from mid-November to mid-February? We believe that if the manure slurry is piped into the marshes at a depth of 1m and then brought to the surface by riser pipes, the protection afforded by the canopy of dead vegetation would allow flow during the winter. While biological activity will be minimal, the mechanical filtration that would take place should provide an effluent of no worse quality than from an anaerobic lagoon during extreme cold weather.

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AGRICULTURAL AND FOREST MANAGEMENT

THE GROWTH AND NUTRIENT UPTAKE OF FORAGE GRASSES WHEN RECEIVING VARIOUS APPLICATION RATES OF WASTEWATER

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ABSTRACT

This study reports on the growth and nutrient removal of forage grasses receiving three years of wastewater applications. The forages received wastewater at various application rates and schedules and were grown in either a Windsor sandy loam or a Charlton silt loam soil. Plant and soil analyses were performed on representative samples during the study.

Wastewater application rates ranged from 5 to 15 cm/wk which annually supplied 257 to 1119 kg of nitrogen/ha. At these rates the crop removed 214 to 472 kg of nitrogen/ha. As the application rate increased, crop removal for nitrogen increased but at a decreasing rate. The forages yielded from 8.2 to 13.7 t/ha of dry matter.

When 430 kg of nitrogen/ha was applied, the crop removed 329 kg/ha or 77% of the applied nitrogen. At this application rate, 110 kg of phosphorus/ha was applied and crop removal accounted for 32 kg/ha or 29%. Differences in crop uptake were not observed in the application schedules which applied 7.5 cm/wk of wastewater.

Plant and soil analyses indicated that concentrations of potassium were near deficient levels. Potassium deficiencies were related to a greater plant removal rate of this element than applied.

INTRODUCTION

Nitrogen is one of the limiting factors in the design of slow infiltration land treatment systems and in the nitrate form is a highly mobile element in soil solution. Removal of nitrogen in a slow infiltration land treatment system is primarily by plant uptake. Since nitrogen uptake by the plant is not a linear function of the rate at which the element is applied, it is important to obtain nitrogen uptake data at various nitrogen application rates.

Proper agronomic management is also required to obtain and maintain the necessary removal of wastewater constituents at land treatment sites. This will include appropriate crop establishment and soil and crop management techniques. Improper management will result in a reduced crop nutrient removal and therefore a lower application rate and a greater land requirement.

In order for a crop to provide for high nitrogen removal, macro and micro nutrients should be in adequate supply. Also information as to changes in soil pH, soil nutrient availability, and interactions among soil nutrients should be known for long term applications of wastewater. Wastewater application schedules should be reviewed to note differences, if any, in crop removal and to obtain better system management.

OBJECTIVE

The objective of this research was to determine the growth of forage grasses and the removal of nutrients by them during three years of wastewater application. The wastewater was applied at various rates and schedules during the growing season.

MATERIALS AND METHODS

The research was conducted at the U.S. Army Cold Regions Research and Engineering Laboratory (CRREL), Land Treatment Research Facility, in Hanover, New Hampshire, during 1974, 1975 and 1976. One component of this facility is a set of six outdoor land treatment test cells, which have been described in Iskandar et al. (1976). The test cells measure 8.4 m x 8.4 m x 1.5 m and are equipped with water sampling devices.

A Windsor sandy loam soil was used in three of the cells and a Charlton silt loam in the other three. A forage mixture that included reed canarygrass (*Phalaris arundinacea* L.), timothy (*Phleum pratense* L. var. 'Climax') and smooth bromegrass (*Bromus inermis* Leyss. var. 'Lincoln') was seeded at the rate of 12.1, 6.6, 5.5 kg/ha, respectively, to all test cells on 21 May 1973. Domestic wastewater collected from Hanover, N.H. was either primarily or secondarily treated and applied to the test cells by spray irrigation. The wastewater contained an average of 28.0 ppm nitrogen, 6.4 ppm phosphorus, 11.0 ppm potassium, with a pH that was near neutrality. The wastewater was disinfected with ozone prior to application.

The application rates and schedules for each test cell during this study are shown in Table 1. The primary and secondary wastewaters were similar in the concentration of constituents which would directly affect plant growth (Table 2).

In May 1976, cells 2, 3, 4 and 5 were reconditioned. The existing vegetation was killed with the herbicide glyphosate. Potassium chloride at the rate of 300 kg of potassium/ha was applied to the four cells. Dolomitic limestone was applied at the rate of 7308 kg/ha to cells 2, 4 and 5 and 6488 kg/ha to cell 3. The soil surface was then tilled and seeded with orchardgrass (*Dactylis glomerata* L. var. Pennlate) and reed canarygrass.

The forage was cut three times per year at a height of 7.5 cm. The harvest dates were 3 June, 28 July and 17 September in 1974; 12 June, 23 July and 23 September in 1975; and, 8 June, 4 August and 15 September in 1976. Each cutting was collected and weighed to obtain the total fresh weight per cell. Randomized grab samples were dried at 70°C and dry weights per plot were calculated. The grab samples were analyzed for nitrogen, phosphorus and potassium using standard procedures (Black, 1965).

RESULTS AND DISCUSSION

Nitrogen

Crop uptake is the principle mechanism for removal of nitrogen and other wastewater elements in the slow infiltration mode of land treatment. In this study weekly wastewater applications during the growing season (April thru October) ranged from 5.0 to 15.0 cm/wk during 1974, 1975 and 1976. The nitrogen loading rate ranged from 257 to 1119 kg/ha (Table 3). Annual nitrogen removal by the forage during this time ranged from 214 to 472 kg/ha.

Forage yields, on a dry weight basis, resulting from the various wastewater application rates ranged from 8.2 to 13.7 t/ha during 1974 to 1976 (Table 3). Increases in nitrogen removal are closely associated with increases in crop yields. Crop yields of about 10.5 t/ha removed over 300 kg/ha of nitrogen annually.

During 1974 and 1975 when the nitrogen loading rate varied from 427 and 433 kg/ha on five of the test cells, the crop removed from 308 to 341 kg/ha of nitrogen (Table 3). At this application rate the forages were removing an average of 77% of the nitrogen which was applied during the growing season.

In 1975 on test cells 1 and 6 at the 5.0 cm/wk application rate, the crop removed more nitrogen than was applied. This reflects the more efficient use of nitrogen by the plant at lower application rates and the availability of residual nitrogen in the soil from previous years applications. Similar findings on crop removal of unaccounted for nitrogen has also been reported for corn and reed canarygrass treated with wastewater by Sopper and Kardos (1974).

Table 1. Test cell weekly application schedules.

Application Period	Test cell no. 1 (Windsor soil)	2 (Windsor soil)	3 (Windsor soil)	4 (Charlton soil)	5 (Charlton soil)	6 (Charlton soil)
1974-1976	5.0-S* (1d)†	15.0-S (3d)	7.5-P (3d)	7.5-P (3d)	7.5-P (24h)**	5.0-S (1d)

* (5.0-S) 5.0 cm of secondary effluent/week (P - primary effluent).
† (d) is number of daily (8 hours) application periods per week.
**(24h) Application occurred over one 24-hour period.

Table 2. Mean annual water quality analyses of primary and secondary effluent.

Parameter	Primary 1974	1975	1976	Secondary 1974	1975	1976
pH (pH units)	7.4	7.0	7.2	7.5	6.7	7.3
conductivity (μmhos/cm)	394	357	457	402	342	458
total carbon (ppm)	84.1	68.6	-	62.5	35.9	-
total phosphorus (ppm)	7.0	6.1	6.4	7.1	5.8	6.1
total nitrogen (ppm)	26.4	-	31.3	26.9	-	27.5
Kjeldahl-N (ppm)	25.7	24.1	31.2	24.2	12.3	23.2
Nitrate-N (ppm)	0.6	0.4	0.1	2.4	9.0	4.3
Nitrite-N (ppm)	0.0	-	-	0.0	-	-
Ammonium-N (ppm)	22.1	21.8	28.7	21.6	11.3	21.1
Potassium (ppm)	8.3	11.2	13.9	8.8	10.6	13.0
Chloride (ppm)	36.1	29.1	36.3	32.7	32.6	37.7

Table 3. Annual crop yields and nitrogen removal by established forage grasses grown in the experimental test cells during 1974, 1975 and 1976.

Application rate cm/wk	Cell No.	Nitrogen applied kg/ha	Nitrogen in forage kg/ha	Yield t/ha
1974				
5.0	6	427	331	10.5
5.0	1	433	308	9.6
7.5	4	523	361	11.6
7.5	3	612	361	10.5
7.5	5	685	378	12.6
15.0	2	1119	450	13.0
1975				
5.0	1	277	306	11.3
5.0	6	285	299	9.2
7.5	5	428	339	12.1
7.5	4	431	341	10.8
7.5	3	432	326	10.8
15.0	2	867	472	13.7
1976				
5.0	1	257	214	8.2
5.0	6	299	240	11.1

Figure 1 shows the removal of nitrogen by crops at the various application rates of nitrogen over the three year period. Crop removal increased as nitrogen application rate increased. The slope of the curve generally obeys the "law of diminishing returns." Therefore as nitrogen application rates increases, crop uptake also increases but at a decreasing rate for each extra unit of nitrogen applied.

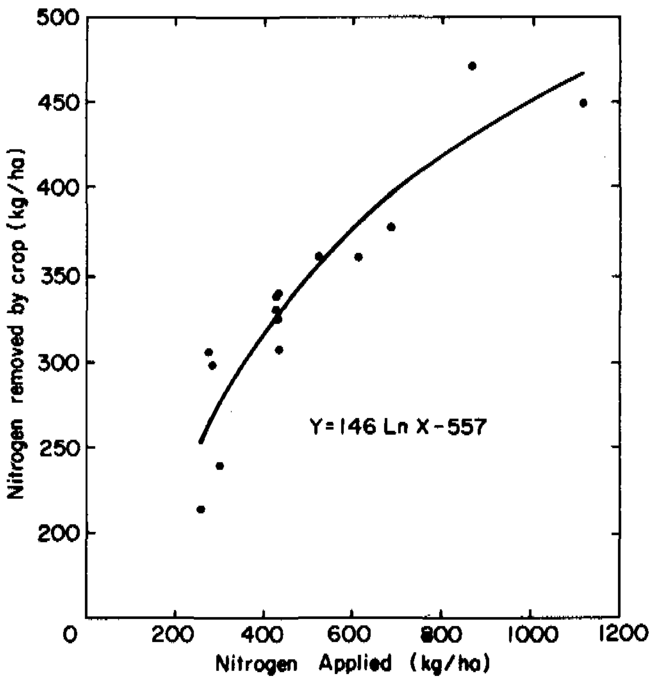

Figure 1. Crop uptake of nitrogen at various application rates over a three year period (1974-1976)

This curvilinear relationship is important when estimating nitrogen balance sheets at land treatment sites. Any adjustments to the application rates will directly influence crop removal. For example, at the 1974 nitrogen application rates of 427, 612 and 1119 kg/ha, the crop removed 331, 361 and 450 kg of nitrogen/ha (Table 3). At these nitrogen application rates the crop removed increasingly smaller percentages of the nitrogen applied during the growing season, i.e. 78, 59, and 40%. The amount of nitrogen not taken up by the crop is potentially available to leach to the groundwater.

Two different application schedules were employed, which supplied 7.5 cm/wk of wastewater during 1974 and 1975 to the Charlton soils. In one treatment the wastewater was applied on three consecutive days (2.5 cm/day) at eight hours per day, with a 16 hour rest period between applications. In the other treatment, wastewater was applied during a single 24-hour period. In both treatments wastewater was applied at the rate of 0.3 cm/hr. As noted in Table 4, when averaging two seasons of data, crop uptake of nitrogen and percolate quality was similar in both the application schedules. These results indicate that varying the application schedule while applying 7.5 cm/wk of wastewater in a slow infiltration system will have little effect on crop uptake and percolate water quality.

Phosphorus

Unlike nitrogen, phosphorus can be retained in the soil and usually under proper management is not a short term problem in land treatment of wastewater. Iskandar et al. (1976) noted that in the CRREL test cells this element was effectively removed by the soil with little or no leaching to the groundwater. The long-term effects of phosphorus are usually associated with determining the life expectancy of a land treatment site through its soil buildup and eventual leaching in soils after many years. Phosphorus removal by plants ranged from 31 to 45 kg/ha when 73 to 222 kg/ha was applied in the CRREL test cells in 1974 and 1975 (Table 5). Removal efficiency was low when compared with nitrogen and ranged from 14 to 42% of the applied element during the two growing seasons (Palazzo, 1976).

In Minnesota (Larson et al., 1975), various forage grass species receiving two rates of wastewater were harvested three times during the year. Crop removal of phosphorus ranged from 23 to 42 kg/ha and 37 to 54 kg/ha at each successive application rate for wastewater. In Pennsylvania, reed canarygrass was irrigated with wastewater over a six year period (1965-1970). Hook et al. (1974) reported that a total of 925 lb/A (1017 kg/ha) of phosphorus was applied and the forage removed 280 lb/A (308 kg/ha) during the six-year period. Therefore, average annual applications and crop removal were 170 and 51 kg/ha, respectively. The authors also noted that, on a properly managed site, most added phosphorus will remain in the soil or be removed by the crop. These data

Table 4. The effects of wastewater application schedules on crop uptake of nitrogen and percolate quality.

Years of application	one 24 hour application: Nitrogen applied	Nitrogen in plant	NO_3-N in percolate	three 8 hour applications: Nitrogen applied	Nitrogen in plant	NO_3-N in percolate
	kg/ha		ppm	kg/ha		ppm
1974-1975	557	354	11.2*	477	341	12.4*

*No statistically significant differences at the 5% level.

Table 5. Amounts of phosphorus applied to test cells and removed by forage during 1974 and 1975.

Application rate	Applied	Removed
cm/wk	------- kg/ha	------
1974		
5.0	116	34
7.5	174	34
15.0	301	45
1975		
5.0	73	31
7.5	108	31
15.0	222	44

indicate that when wastewater annually supplies appoximately 100 kg/ha of phosphorus to a site the forage grass is able to remove about 30 kg/ha or 30% of the applied.

Potassium

Potassium is one of the macro elements, along with nitrogen and phosphorus, required for plant growth. Although contained in wastewater it is not considered a pollutant and therefore receives little attention in land treatment of wastewater. However, potassium is important in maintaining optimum plant growth and, in turn, maximum nitrogen removal from wastewater.

Typical plant contents of potassium average about 90% of that for nitrogen in forage grasses (National Research Council, 1970; Potash Institute of America, 1973). Therefore, if a forage removes 330 kg/ha of nitrogen annually, 300 kg/ha of potassium should be available to the plant.

Although the characteristics of wastewater will vary, it will usually contain 2 to 3 times more nitrogen than potassium. In the design and operation of land treatment systems it is usually considered that the crop will remove large quantities (about 70%) of the applied nitrogen. If the plant removes large quantities of nitrogen, it would also remove greater amounts of potassium than was applied.

In this study more potassium was contained in the crop than was applied in the wastewater (Table 6). The unaccounted for potassium removed by the crop during 1974 and 1975 ranged from 12 to 159 kg/ha at wastewater application rates from 5 to 15 cm/wk. In another wastewater irrigation study at Penn. State greater or nearly equal amounts of potassium were removed by corn silage than what was supplied in the wastewater (Sopper and Kardos, 1974).

The potassium concentrations in the forages and soils taken from the test cells from 1973 to 1976 are shown in Table 7. Concentrations of this element consistently declined in both the soils and plants during this study. Both the concentrations of potassium in the leaf and the soil were low or marginally deficient in terms of optimum growth of forage grasses (Martin and Matocha, 1973; Doll and Lucas, 1973).

Since plant removal of potassium is reported as being about 90% of that for nitrogen, the following practical equation was developed to avoid deficiencies of potassium at land treatment sites:

fertilizer potassium required (kg/ha)

= estimated annual removal of nitrogen by crop (kg/ha) X 0.9

\- potassium supplied by wastewater (kg/ha)

Table 6. Quantitative amounts of potassium applied and removed by crop during 1974 and 1975.

Cell No.	Application rate	Applied	Removed	Unaccounted for
	cm/wk	kg/ha	kg/ha	kg/ha
1974:				
1	5.0	98	257	159
2	15.0	286	350	64
3	7.5	121	244	123
4	7.5	115	291	176
5	7.5	241	302	61
6	5.0	104	241	137
1975:				
1	5.0	152	238	86
2	15.0	233	340	107
3	7.5	223	246	23
4	7.5	216	269	53
5	7.5	233	245	12
6	5.0	153	208	55

Table 7. The concentrations of potassium in plants and soils receiving wastewater.

Cell No.	Application rate cm/wk	Sample*	1973	1974	1975	1976
1	5.0	leaf	2.65	2.67	2.10	2.04
		soil	0.12	0.14	0.06	0.07
2	15.0	leaf	3.00	2.72	2.47	--
		soil	0.12	0.16	0.06	0.08
3	7.5	leaf	2.75	2.37	2.22	--
		soil	0.12	0.14	0.10	0.05
4	7.5	leaf	2.70	2.54	2.36	--
		soil	0.09	0.37	0.11	
5	7.5	leaf	3.05	2.46	2.07	--
		soil	0.09	0.37	0.11	
6	5.0	leaf	2.40	2.31	2.22	1.83
		soil	0.09	0.22	0.07	

* Leaf concentrations in %, soils sampled at the 0 - 15 cm depth and analyses reported in meq/100g.

If the soils are initially low or deficient in this element, then extra potassium fertilizer will be required, depending on soil test results. The reactions of potassium in soils are relatively complex and largely dependent on soil type. Therefore, soils and plants should be checked annually for potassium to avoid excess or deficiency problems. One problem which might develop with excessive applications of potassium is magnesium deficiency. Magnesium is important to both crop growth and animal nutrition. Therefore, a thorough analysis of soils is needed to avoid imbalances of exchangeable cations and other elements in soils.

SUMMARY

The growth and nutrient removal of forage grasses receiving various wastewater application rates and schedules was studied over a three year period from 1974 to 1976. Nitrogen applications, resulting from the wastewater irrigation, ranged from 257 to 1119 kg/ha. As the application rate increased, crop removal increased but at a decreasing rate.

The results of this study show that when 430 kg of nitrogen/ha was applied in the wastewater, the forage removed an average of 329 kg of nitrogen/ha or 77% of the applied. Forage yields ranged

from 8.2 to 13.7 t/ha and were noted as being related to the removal of nitrogen. The wastewater applied also contained about 110 kg of phosphorus/ha and the crop removed 32 kg/ha.

Differences in crop uptake of nitrogen, phosphorus and potassium were not dependent on the schedule at which the wastewater was applied. Therefore, there was no advantage in spraying the wastewater at a rate of 0.3 cm/hr in one continuous 24 hour period or for 8 hours a day over a three day period.

Reductions in the plant and soil concentrations of potassium were related to the N/K ratio contained in the plant as compared to that supplied by the wastewater. The removal rates of potassium by plants, with optimum removal of nitrogen, exceeded the potassium supplied by the wastewater. An equation is presented to assist in avoiding deficiency problems with this element at slow infiltration land treatment sites.

From the study it appears that long term use of typical municipal wastewaters may cause imbalances in soil and plant nutrition. These imbalances could affect plant growth and nutrient removal, and could create dietary problems for animals consuming this forage. Therefore, plant and soil analyses should be continuously performed and local agricultural persons consulted on the results to avoid potential problems.

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AGRICULTURAL AND FOREST MANAGEMENT

USE OF WASTEWATER IN TURF IRRIGATION

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ABSTRACT

The use of wastewater for turf irrigation was investigated at the USAF Academy. From the time of its construction in 1960, the Eisenhower Golf Course has been irrigated with secondary wastewater. Soil sampling began in 1964. Elevated levels of potassium, copper, iron and zinc occurred in soils from the irrigated portions of the golf course. Analysis of soil cores from sites sampled in 7.62 cm (3 inch) increments to a depth of 45.72 cm (18 inches) indicates a buildup in metal concentration which is inversely proportional to depth. A change in the species composition of the turf has occurred; the perennial grasses originally established on the greens and tees (seaside bentgrass, *Agrostis palustris*) and on the fairways (common bluegrass, *Poa pratensis*; redtop, *agrostis alba*; and colonial bentgrass, *agrostis tenuis*) have been replaced by shallow rooted annuals, primarily annual bluegrass (*Poa annua*). Some adverse effects (e.g., yellowing and premature needle drop) have also been observed in Ponderosa Pine (*Pinus ponderosa*) on and adjacent to irrigated areas. This report presents a portion of the data collected over 12 years of soil sampling.

INTRODUCTION

Land application as a method for recycling liquid effluent from municipal and industrial sewage treatment plants is of current interest. This method has been proposed as the final step in wastewater treatment because it is a very effective way to renovate wastewater which can then recharge the ground water aquifer. In addition where irrigation is necessary or desirable sewage effluent can be used as irrigation water thereby reducing the demand on potable water supplies and water treatment facilities. Land application also has potential for reducing commercial fertilizer costs by taking advantage of the nutrients in sewage effluent.

The principal use of municipal sewage effluent irrigation has been in turf grass areas, such as municipal parks, golf courses, cemeteries, and highway medians. Many U.S. municipalities and several Air Force bases use sewage effluent irrigation (18). However in recent years projects have been initiated to use sewage effluent in agricultural and forest irrigation (9, 12, 15). Other studies have been conducted using sewage effluent for reclamation of disturbed land areas such as strip mines and mine spoils. Some of these studies have been designed to determine the effects of overland flow treatment of wastewater (3, 10).

Numerous examples have been reported where wastewater irrigation caused changes in soil composition and in soil build up of the chemical constituents contained in the wastewater (8, 13). Some of these soil changes are acceptable or even desirable; however, in some cases, the build up has resulted in the formation of nonproductive soils (4). In most instances, however, under proper management procedures, land treatment systems can renovate wastewater to drinking water quality in the percolate water and not impare plant growth or physical soil stability (11).

In semiarid or arid environments, even when well or river water has been used for agricultural irrigation, soil build up of various chemicals found in the water can be a problem. In arid agricultural regions high evaporation rates may accelerate surface accumulation of salts, thus requiring soil flushing procedures. However wastewater irrigation practices for municipal turf grass management do not presently include a freshwater flushing procedure. Sewage effluents generally contain a diversity of chemicals at concentrations not found in irrigation water. Some of these chemicals have been demonstrated to be biologically concentrated in plant tissue (17).

The present study was initiated to determine the extent of salt and mineral accumulation in soils and to evaluate the effects that have occurred during 16 years of sewage effluent irrigation on turf grass at the U.S. Air Force Academy. A complete description of all soil data is presented elsewhere (5).

SITE DESCRIPTION

Location and Climate

The Air Force Academy is located 27.3 km (11 miles) north of Colorado Springs in El Paso County, Colorado. It includes 7244.1 ha (17,900 acres) of foothills and plains at an altitude ranging from 1932.5 to 2438.5 m (6,340 to 8,000 feet) on the eastern slope of the Rocky Mountains.

The high mountains to the west of the Academy block atmospheric moisture originating from the Pacific Ocean so that the primary source of moisture is the Gulf of Mexico. The transport of atmospheric water from the Gulf into central Colorado is inconsistent. Consequently, most of the area has relative low humidity and the average annual precipitation is 44.5 cm (17.5 inches) which is slightly more than a semiarid climate. A large portion of the precipitation occurs in short, often destructive, cloud bursts which are generally localized. This rainfall is so rapid that most of the moisture is lost to runoff.

Mean wind speed at the Academy is 9 knots, however, wind speeds and direction can be highly variable.

Geology and Soils

Precambrian Pikes Peak Granite forms the mountains of the Rampart Range at the west edge of the Academy area. The predominant bedrock within the Academy area is Dawson Arkose of Cretaceous and Paleocene age. The Dawson Arkose consists about equally of coarse arkosic sandstone and of interbedded lenticular siltstone and clay. The soil type analyzed has been reported as sandy loam.

A complete description of the geology of the Air Force Academy site has been reported by Varnes et al. (16).

Golf Course and Irrigation System

The Eisenhower Golf Course is located in South Lehman Valley at the Academy. The first course was opened in 1960 with 18 holes and a driving range. In construction of the course, provision was made for sprinkler irrigation with secondary wastewater. Modification in 1975 relocated sprinkler heads in order to avoid irrigation of trees in a number of areas and to more efficiently distribute water in turf areas. Irrigation with wastewater has been continuous on this course since the system was activated in 1960.

Irrigation Water Source

Irrigation Water for the golf course originates from the Air Force Academy Sewage Treatment Plant. The plant includes both full primary and secondary treatment (1). Sewage effluent is pumped sequentially through a series of wastewater lagoons (non-potable reservoirs) (Figure 1). These have been identified as wastewater lagoons (non-potable reservoirs) 1, 2, 3, and 4, as defined by the pumping sequence. Lagoons 1 and 2 are aerated. Water from wastewater lagoon 2 is utilized for irrigation of the golf course.

Plant Species on the Eisenhower Golf Course

The dominant tree species on the golf course is Ponderosa Pine (*Pinus ponderosa*). Fairways have been seeded with common bluegrass (*Poa pratensis*),

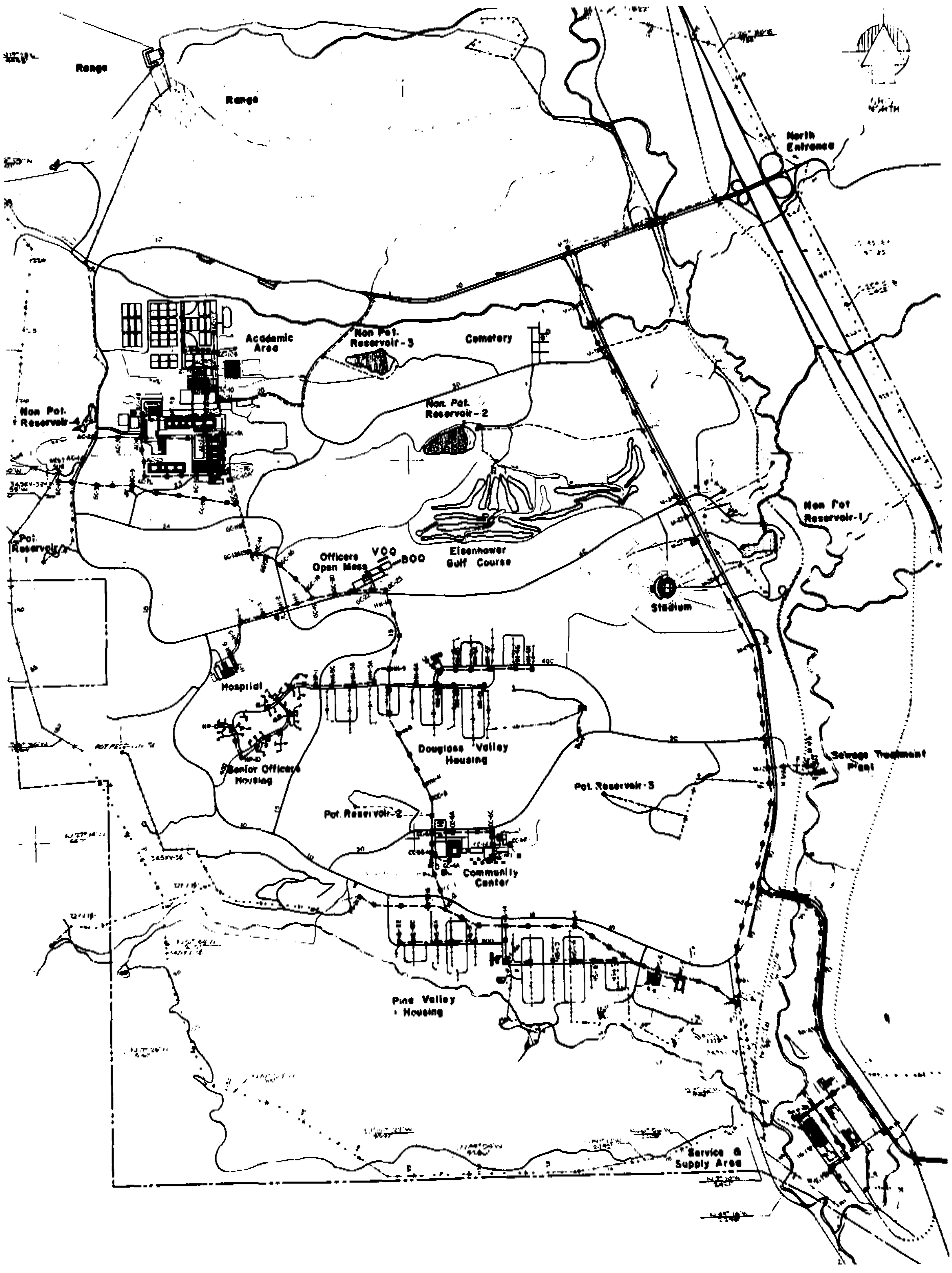

Figure 1. Detailed Map of the U.S. Air Force Academy

redtop (Agrostis alba) and colonial bentgrass (Agrostis tenuis). Greens and tees have been seeded with seaside bentgrass (Agrostis palustris).

Repeated botanical observations have been made by Air Force Academy personnel (unpublished data). These observations have indicated a gradual shift of species density on greens, tees, and fairways from the preferred seeded perennial grasses to the less desirable, shallow rooted annual bluegrass (e.g., Poa annua). In fact, it is estimated that some areas are now more than 90% annual bluegrass.

SAMPLING PROTOCOL AND ANALYTICAL RESULTS

Water Composition

In 1975 a water sampling program was initiated for the Academy Golf Course, and three samples were drawn from the irrigation system during the time period between May and August. These samples are representative of the secondary sewage effluent which is used to irrigate the golf course and were analyzed by the USAF Occupational and Environmental Health Laboratory, Kelly AFB, Texas. The composite results are presented in Table 1. For comparison purposes, typical secondary effluent values (14) are also given in Table 1.

Water Application Rates

Although quantitative application rates for irrigation water are not available, the following estimate has been provided by the Golf Course Superintendent:

a. Approximately 44.5 ha (110 acres) of the golf course are irrigated.
b. There are 753 sprinkler heads used to irrigate the golf course.
c. Each sprinkler head output is .10598 m^3/min (28 gallons per minute).
d. Each sprinkler head runs approximately 15 minutes per night for approximately 180 days per year.
e. The annual irrigation rate estimate is 4841.97 m^3/ha/year (1.58 acre feet per year).[1]

Chemical Constituent Application Rate

The amount of each chemical in the wastewater based on the estimated irrigation rate is given in Table 1.[2] The data are presented to give an "order of magnitude" estimate for the annual rate of application.

Soil Analysis

Soil analyses were performed by the Cooperative Extension Service and Experimental Station Soil Testing Laboratory, Colorado State University, Fort Collins, Colorado. Initial samples were obtained with a 2.54 cm (one inch) soil probe to a depth of 7.62 cm (3 inches). Soil samples from greens, tees, and fairways in 1975 and 1976 were taken with a 7.62 cm (3-inch) diameter soil auger. Soil samples were also taken from adjacent forested areas with the soil auger. Sites were selected from non-irrigated, fresh water irrigated, and wastewater irrigated forested areas.

DISCUSSION

Potassium (K_2O) has increased in the soil surface layer between 1964 and 1972 (Figure 2). This trend seemed to indicate that other chemical constituents of the irrigation water might also be accumulating in the soil surface layer.

Analysis of zinc data obtained from soil core samples in 1975 and 1976 indicated a two-fold effect. First, the zinc concentration found in the irrigated soil is elevated above the concentration found in non-irrigated soil (26.7 ppm irrigated vs. 0.6 ppm non-irrigated). Secondly, there is a pronounced concentration of zinc in the top 7.62 cm (3 inches) of the irrigated soil, but it decreases with depth (Figure 3). This trend is repeated in all soil core samples for iron, copper, potassium and

[1]The calculation for water application rate is as follows: (753 heads) (.10598 m^3/min/head)(15 min/day)(180 days/yr) ÷ (44.5 hectares) = 4841.97 m^3/ha/year.

[2]The calculation for irrigation water chemical content is in grams per hectare per year. Note that they are based on only three water samples and the estimated water application rate.

Table 1. Irrigation water comparison/analysis and chemical constituent application rate.

Parameter	Typical Secondary Effluent[a] (14)	USAFA Irrigation Water Analysis[a]	Chemical Constituent Application Rate[b] gm/ha/year
Ammonia Nitrogen	9.8	12.2	59071.7
Cadmium	0.1	<0.01	48.4
Chemical Oxygen Demand	70	24	Not Applicable
Chlorides	100	27	130732.3
Chromium (Hexavalent)	Not Available	<0.01	48.4
Chromium (Total)	0.2	<0.05	242.1
Color	Not Available	22 units	Not Applicable
Copper	0.1	<0.02	96.8
Cyanides	Not Available	<0.01	48.4
Dissolved Solids	Not Available	250	1210485.0
Fluorides	Not Available	0.07	3389.3
Iron	0.1	0.22	1065.2
Lead	0.1	<0.05	242.1
Manganese	0.2	0.1	484.2
Mercury	5.0 mg/ml	<5.0 mg/ml	24.2
Nitrates	20	17	82313.0
Nitrite Nitrogen	Not Available	0.05	242.1
Oils and Greases	Not Available	.53	2566.2
Phenols	0.3	0.08	382.5
Phosphates	10.0	20.2	97807.2
Silver	Not Available	<0.01	48.4
Sulfates	125.0	34	164626.0
Surfactants	Not Available	0.3	1452.6
Total Organic Carbon	Not Available	9	43577.5
Turbidity	Not Available	3 units	Not Applicable
Zinc	0.2	0.07	338.9

[a]All results in mg/l unless otherwise noted.

[b]To determine the amount of chemicals applied in gm/ha/year the water concentrations obtained by analysis at the Environmental Health Laboratory at Kelly AFB are treated as follows:

(mg of chemical/l)(.001 g/mg)(4841970 l/hectare/year) = gm of chemical/hectare/year.

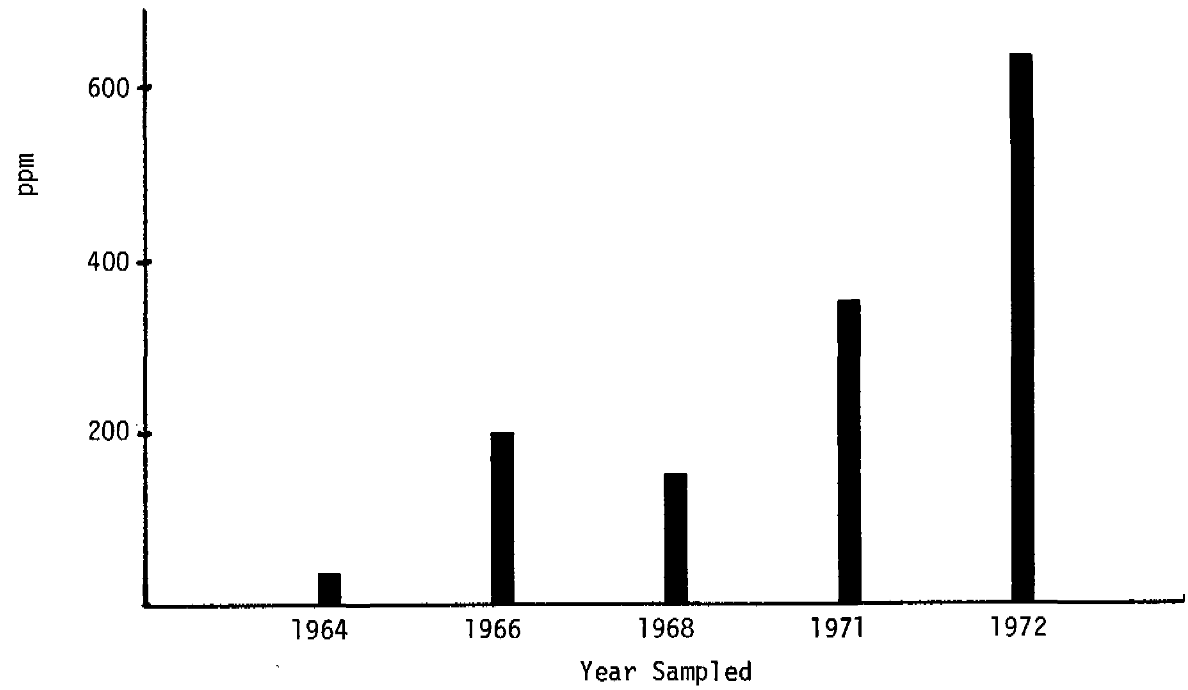

Figure 2. Potassium (K_2O) Concentration vs. Time

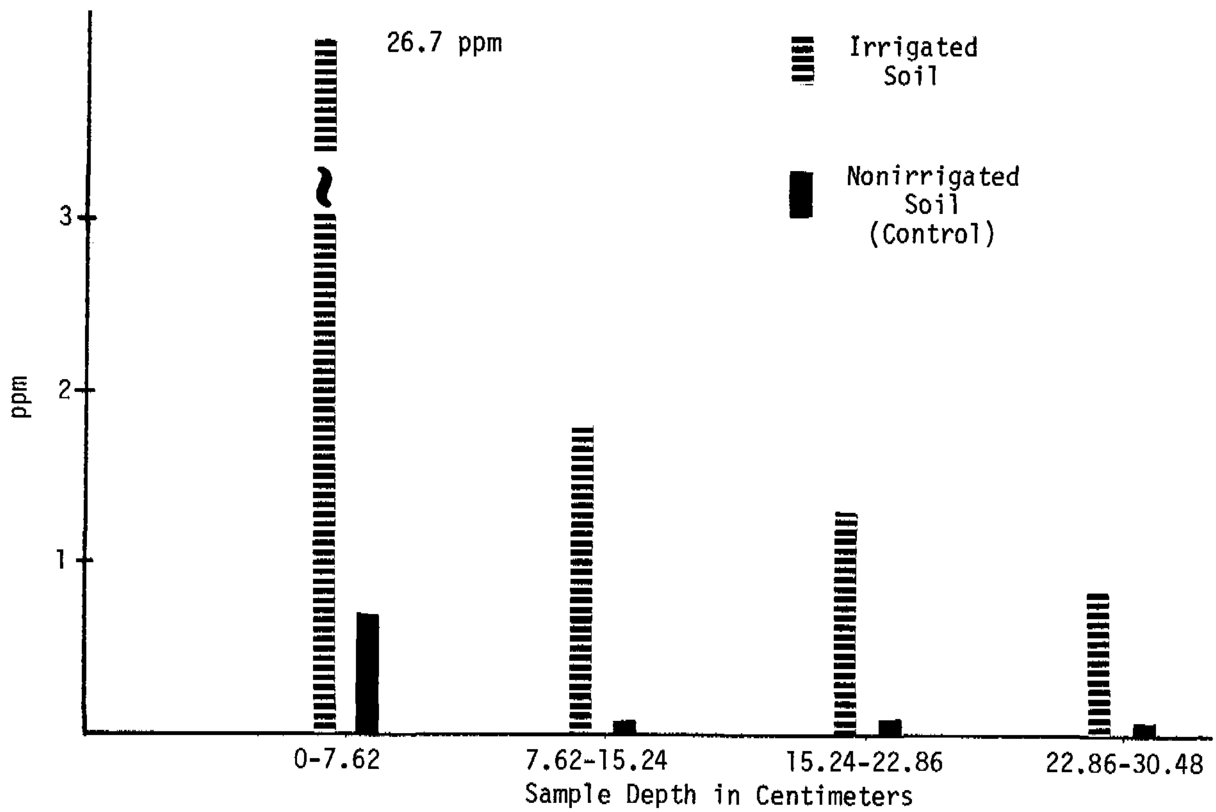

Figure 3. Zinc Distribution in the Soil Profile.

several other minerals (5). Similar distribution in the soil profile has been reported for other areas where sewage effluent irrigation has been used (9).

At the present time it is not known if the soil mineral buildup is responsible for the observed grass species shift and tree distress. The Air Force Academy Cemetery which receives the same water, but in a different application regime, (i.e. infrequent, long, deep penetrating waterings), has over a period of five years successfully transitioned from a condition in which annual bluegrass Poa annua predominated, to the present condition where the more desirable common bluegrass, Poa pratensis is the dominant grass species.

The literature (2) on Poa annua indicated that this shallow rooted species preferred high nutrients, frequent watering and short mowing. This would seem to indicate the species shift to Poa annua observed on the golf course is due to a combination of required management practices (i.e. short mowing coupled with frequent, short, shallow waterings) which reinforced the effects of the wastewater borne nutrients and chemicals.

Therefore the interaction between the high nutrient content of the irrigation water and the management practices necessary to maintain an outstanding golf course in a semiarid environment may have combined to cause the observed spread of Poa annua.

While modification of management practices may allow the reversal of the grass species shift trend and eliminate the tree stress in the short term, the long term salt and metal buildup in the soil may still be a problem. Current work has indicated that trace metals which build up in the surface soil (a) will not migrate through the soil column, and (b) are not considered biologically available once they are incorporated into the soil structure (6, 7).

Much of the work on wastewater irrigation has been done in areas of moderate to high rainfall and low evaporation. However, there is a growing interest in the use of wastewater in areas of low rainfall and high evaporation. Although agriculture has long practiced periodic soil flushing with fresh water, there is presently no similar practice in municipal turf grass wastewater irrigation, and indeed many of our wastewater irrigation systems are dedicated to the use of only wastewater.

At the present time more research is needed to look at chemicals which are deposited with the sewage irrigation water and to determine whether these chemicals remain biologically unavailable once they are incorporated in the soil surface layer. The Air Force Academy in conjunction with the Cold Regions Research and Engineering Laboratory is currently conducting a joint study to determine the effects of wastewater irrigation as it applied to arid high altitude environments. This study will also incorporate analysis of lagoon treatment and further examine the fate of and the soil/vegetation interaction mechanisms of the chemical constituents in wastewater.

CONCLUSION

This study on the Eisenhower Golf Course was initiated because of the observed replacement of the desired perennial grasses with a less desired species of annual bluegrass, Poa annua. In addition, severe yellowing and stress was observed in the native Ponderosa Pine located in areas under irrigation.

Analysis of soil data gathered between 1964 and 1972, indicated a build up of potassium in the soil surface layer. In addition, soil core samples from irrigated sites revealed elevated levels of zinc, copper, iron, and several other chemicals. These chemicals were concentrated in the soil surface layer, and their concentration decreased with depth. It was not determined however, whether the chemicals observed in the soil surface layer were biologically available and thus actually responsible for the observed changes in vegetative growth and development.

Comparison of an area which receives frequent shallow irrigation to an area which receives infrequent deep irrigation indicates that proper management practices can be used, at least in the short term, to mitigate any undesirable effects on grass species distribution which might be contributed to by the use of wastewater irrigation.

ACKNOWLEDGEMENTS

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AGRICULTURAL AND FOREST MANAGEMENT

HEAVY METAL CONTENT AND MINERAL NUTRITION OF CORN AND PERENNIAL GRASSES IRRIGATED WITH MUNICIPAL WASTEWATER

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The renovation of municipal wastewater effluent by land treatment was studied for 4 years (1974-1977). This field study was conducted on a well-drained, silt loam soil (Typic Hapludoll) with the water table at approximately 150-cm depth. The 60 cm of silt loam are underlain by outwash sand and gravel. The effluent, obtained from an activated sludge treatment facility that serves a domestic, suburban population, was applied on corn at rates of ≃ 5 and 10 cm/wk throughout the growing season. Slightly higher rates were applied on perennial grasses. The control treatments for both corn and forage were fertilized with mineral fertilizer and irrigated with ground water in amounts required for optimum dry matter production.

Trace metal concentrations in the effluent were low: 22 μg Zn/l, 11 μg Cu/l, 4 μg Ni/l, 3 μg Pb/l, 4 μg Cr/l, and 0.2 μg Cd/l. Sodium, Ca, Mg, and K concentrations averaged 275, 70, 24, and 12 mg/l, respectively. Only minor changes in trace metal contents of corn and grass tissues were observed.

Copper concentrations in corn and grasses decreased with added effluent, often to the point of plant deficiencies. The K content of corn was reduced to the deficiency level by effluent applications, probably as a result of increased Na uptake. Effluent additions greatly reduced the Fe and Mn contents of all tissue; the reduction may have resulted from i) a higher soil pH on the effluent-treated areas and ii) a very active microbial environment that is continuously binding solution Fe and Mn into an unavailable form.

INTRODUCTION

The renovation of secondary-treated municipal wastewater (hereafter referred to as "effluent") by land treatment has received increasing attention in recent years, with the implementation of stricter laws on the discharge of "pollutants" into surface waters. Besides strong economic and energy-related arguments, land treatment has the favorable attributes of supplying essential plant nutrients, particularly nitrogen (N), and water for agriculture. Bouwer and Chaney (1974) have published a comprehensive review of land treatment of effluent. When effluent loading rates are high, attention must be given to the possible accumulation of toxic trace metals in plant tissue as well as to plant nutrient imbalances that would reduce plant growth.

In this paper, we report the effect of 4 years of high effluent loadings on the trace metal and mineral nutrient composition of corn (*Zea mays* L.) and selected perennial forage grasses. In a companion paper, Marten et al. (1978) discuss the effects of high effluent loadings on feed quality of these crops.

MATERIALS AND METHODS

The soil at the field site was a well-drained, Typic Hapludoll (Waukegan). It is uniform silt loam material overlying neutral to calcareous outwash gravel at about 60 cm with a water table at 140 to 150 cm. Site selection, description of the drainage and

Table 1. Average total elemental composition of wastewater effluent from the Apple Valley, Minnesota, treatment plant (1974-1977).

Element	Concentration		
	Apple Valley+	(Range)	Typical Value‡
	mg/l		
Na	275	(216 - 304)	50
Ca	70	(67 - 76)	24
Mg	24	(24 - 24)	17
K	12	(11 - 14)	14
	μg/l		
Zn	22	(18 - 26)	150
Cu	11	(8 - 20)	100
Pb	3	(2 - 6)	50
Cr	4	(3 - 4)	25
Ni	4	(2 - 7)	20
Cd	0.2	(<0.2 - 0.2)	5

+ Values are averaged over 4 years of annual composite samples.
‡ Values are taken from compilation by Dowdy, Larson, and Epstein (1976).

irrigation systems, and experimental design have been detailed by Clapp et al. (1977).

The effluent used was a secondary-treated wastewater that had passed through a dual-bed polishing filter. Trace metal levels in the effluent (Table 1) were 5 to 10 times lower than "typical" values compiled by Dowdy et al. (1976) but sodium (Na) and calcium (Ca) concentrations were higher than levels often observed for secondary effluent. The pH was ≈ 8.1.

Effluent was sprinkler irrigated on corn and grasses at nominal rates of 5 (low) and 10 (high) cm/wk throughout the growing season on the corn and slightly higher rates on the grasses (Table 2). Control treatments for both corn and grasses were fertilized with mineral fertilizer and irrigated with ground water in amounts required for high dry matter production.

Corn leaf samples opposite and directly below the ear were collected at silking, while the other corn tissues were sampled at physiological maturity. Grass samples were collected from each of three harvests and composited on a weighted average basis. All plant tissues were dried at 70 C, ashed at 450 C, and extracted with 2.0 N HCl. The extract was analyzed by an inductively coupled plasma source emission spectrograph for all elements except cadmium (Cd), nickel (Ni), lead (Pb), and chromium (Cr). These latter trace metals were determined by a direct reading atomic absorption spectrophotometer utilizing a deuterium lamp background corrector where appropriate (Dowdy and Larson, 1975). The elements, zinc (Zn), copper (Cu), Cd, Cr, Ni, and Pb, are collectively referred to as "trace metals" in this paper.

RESULTS AND DISCUSSION

In any discussion of trace metal uptake and accumulation by plants, soil pH must be considered. Initial soil pH was 6.0. By liming and 4 years of effluent (pH ≈ 8.1) application, soil pH ≈ 7.0 on control plots versus 7.4 on the low and high effluent irrigation areas for both corn and grass plots. Soil pH in this range is optimal for minimizing trace metal uptake by growing plants.

One of the major concerns associated with land applications of any waste is the subsequent accumulation of trace metals in crops produced on treated soil, which may ultimately prove detrimental to the food and feed chain. Total metal additions for the 4-year period on corn and grass areas are presented in Tables 3 and 4, respectively. The maximum addition of 1250 ha-cm of effluent (high forage) resulted in the addition of only 0.02 kg Cd/ha. To place this Cd addition into perspective, one should know

Table 2. Annual and total quantities of wastewater effluent applied (1974-1977).

Crop	Treatment	Year 1974	1975	1976	1977	Total
		ha-cm				
Corn	Low	136	109	112	145	502
	High	223	197	218	302	940
Forage	Low	169	137	145	247	698
	High	295	240	268	450	1253

Table 3. Trace-metal concentrations in corn tissues grown following 4 years of irrigation with wastewater effluent.

Treatment	Total Metal Applied	Metal Concentration in Tissue+			
		Leaf	Stover	Grain	Cob
	kg/ha	μg/g			
		Zn			
Control	0	24a	16a	27a	11ab
Low	1.09	23a	21a	26a	14a
High	2.03	25a	12a	22b	8b
		Cu			
Control	0	8.6a	4.9a	0.2a	2.1a
Low	0.56	4.1c	2.6c	0.3a	1.5b
High	1.02	6.1b	3.9b	0.2a	1.9a
		Cd			
Control	0	0.07a	0.12a	<0.11a	0.09a
Low	0.01	0.07a	0.08a	<0.11a	0.08a
High	0.02	0.09a	0.09a	<0.11a	0.08a
		Cr			
Control	0	1.28a	1.26a	0.11a	1.80a
Low	0.20	0.80a	1.27a	0.17a	1.83a
High	0.38	0.90a	1.26a	<0.07a	1.90a
		Ni			
Control	0	0.85a	0.57a	0.24a	1.85a
Low	0.19	0.41b	0.56a	0.20a	1.62a
High	0.37	0.33b	0.68a	0.14a	1.77a
		Pb			
Control	0	0.45a	0.88a	0.47a	0.51a
Low	0.15	<0.40a	0.55a	0.47a	0.52a
High	0.28	<0.40a	0.80a	0.49a	0.53a

+ Means of 4 replications. Values reported on 70 C weight basis. Means within columns within elements followed by different letters are significantly different ($P \leqslant .05$); Duncan's multiple range test.

that EPA's "Proposed Classification Criteria" for Solid Waste Disposal Facilities would allow a maximum cumulative Cd addition of 20 kg Cd/ha (EPA, 1978). If based on this criterion, a site life of 4000 years would be projected. Relatively small quantities of Zn, Cu, Cr, Ni, and Pb were also applied.

Trace metal levels did not increase in the corn tissues studied (Table 3). In fact, Cu concentrations of leaf and stover tissue and Ni content of leaf tissue decreased slightly. The lack of increase in the trace metal concentrations of corn tissues is not surprising in light of the discussions in the previous paragraph on the relatively small quantities of metals added. These findings confirm earlier predictions of Bouwer and Chaney (1974) that "the level of metals in municipal wastewater should soon be sufficiently low that heavy metals will not be a limiting factor in long-term wastewater irrigation practices."

Both Cr and Ni accumulated in the cob tissue of corn, although there was no difference in concentration as a result of effluent applications. Because of this partitioning effect, Cr and Ni uptake would be overlooked if cob tissue were not analyzed. Indeed, we (unpublished data) have observed this for Cr, where sewage sludge was the metal source.

Trace metals did not accumulate in reed canarygrass (_Phalaris arundiancea_ L.), quackgrass (_Agropyron repens_ L.), tall fescue (_Festuca arundiancea_

Table 4. Trace-metal concentrations of selected forage species grown following 4 years of irrigation with wastewater effluent.

Treatment	Total Metal Applied	Metal Concentration in Tissue+ Reed Canarygrass	Quackgrass	Tall Fescue	Orchardgrass
	kg/ha	µg/g			
		Zn			
Control	0	28a	22b	22a	25a
Low	1.52	34a	30a	20a	22a
High	2.67	24a	23b	21a	23a
		Cu			
Control	0	6.3a	6.4a	5.0a	5.9a
Low	0.76	5.1a	5.5a	3.4b	5.1a
High	1.35	3.3b	4.3b	3.6b	4.2b
		Cd			
Control	0	0.07a	0.08a	0.10a	0.07a
Low	0.01	0.10a	0.11a	0.10a	0.09a
High	0.02	0.07a	0.10a	0.07a	0.07a
		Cr			
Control	0	0.74a	0.63a	0.62a	<0.59a
Low	0.28	0.66a	<0.59a	0.60a	<0.59a
High	0.50	0.76a	0.60a	0.66a	0.68a
		Ni			
Control	0	1.96a	1.11a	1.94a	2.44a
Low	0.26	2.15a	1.27a	1.37a	1.55a
High	0.48	1.95a	1.16a	1.57a	1.87a
		Pb			
Control	0	0.64a	0.68a	0.57a	0.80a
Low	0.21	0.58a	0.74a	0.71a	0.65a
High	0.37	0.63a	0.53a	0.46a	0.43a

\+ Means of 4 replications for 3-time cut composite samples (except tall fescue, which was a 4-time cut). Values reported on 70 C weight basis. Means within columns within elements followed by different letters are significantly different ($P \leq .05$); Duncan's multiple range test.

Schreb.), or orchardgrass (*Dactylis glomerata* L.) as a result of effluent applications (Table 4). Copper levels decreased with increasing levels of effluent application, particularly in reed canarygrass and quackgrass. This may be a developing trend, but it did not consistently hold for previous years.

Zinc and Cu are essential plant nutrients. The Zn levels in the treated plants were considered adequate for both corn (Melsted et al., 1969) and forages (Martin and Matocha, 1973). However, the Cu concentration in corn leaf tissue grown on the effluent treatments was close to the critical level of 5 µg/g (Melsted et al., 1969), below which yield reductions can be expected. Our Cu values for grass tissues were well below the average, 11 µg/g, value listed for smooth bromegrass in Wisconsin (Wedin, 1974) and would be considered deficient, 6 µg/g, for alfalfa (Martin and Matocha, 1973). Indeed alfalfa produced in 1976 on our high effluent areas averaged 4.6 µg Cu/g tissue.

All elements listed for corn in Table 5 and grasses in Table 6 are essential for plant growth except sodium (Na). The phosphorus (P), Mg, and boron (B) concentrations in corn were insensitive to effluent applications and are considered adequate for grain production (Larson and Hanway, 1977). Potassium (K) and Ca levels in corn tissue, except grain, decreased with effluent applica-

Table 5. Elemental composition of selected corn tissues grown following 4 years of irrigation with wastewater effluent.

Treatment	Tissue Concentration+ Leaf	Stover	Grain
	---------- % ----------		
		P	
Control	0.34a	0.08b	0.37a
Low	0.28b	0.16a	0.38a
High	0.35a	0.14a	0.37a
		K	
Control	2.22a	2.10a	0.44a
Low	1.47b	1.51c	0.48a
High	1.43b	1.84b	0.50a
		Ca	
Control	0.67a	0.40a	0.00a
Low	0.51b	0.29b	0.00a
High	0.50b	0.31b	0.00a
		Mg	
Control	0.37a	0.30a	0.19a
Low	0.41a	0.23b	0.15a
High	0.31a	0.18b	0.16a
	-------- μg/g --------		
		Na	
Control	1330c	550c	5b
Low	4340b	4050b	31a
High	5670a	6080a	38a
		Fe	
Control	1970a	217a	24a
Low	120b	91b	16b
High	164b	103b	19b
		Mn	
Control	220a	50a	7a
Low	30b	25b	5b
High	30b	25b	5b
		B	
Control	9.8a	7.7a	2.3a
Low	9.8a	8.7a	2.3a
High	9.5a	8.6a	2.7a

+ Means of 4 replications. Values reported on 70 C weight basis. Means within columns within elements followed by different letters are significantly different ($P \leqslant .05$); Duncan's multiple range test.

tions, in spite of supplemental K fertilization. These decreases probably were related to increased Na uptake associated with large Na additions by effluent. In fact, corn was K deficient, which must be a management concern for any wastewater land treatment system.

Major reductions in the iron (Fe) and manganese (Mn) contents of corn tissues resulted from effluent applications, but reductions were the same for both rates of application. Such reductions defy a simple explanation, but may be related to a very active microbiological environment in which new Fe- and Mn-binding sites are being generated continually, so that solution Fe and Mn were bound or complexed in an unavailable form. Also, the higher soil pH of the effluent-treated plots could help explain the lower Fe and Mn levels. However, the Fe and Mn contents of leaf tissue were sufficiently high for normal corn growth and production on all treatments.

The P concentrations in grasses (Table 6) were enhanced by effluent applications, although P content did not differ between high and low treatments. The enhanced P levels may explain the apparent Cu deficiencies noted for grasses grown on effluent treated plots. Wedin (1974) reported that Cu, Fe, and Zn deficiencies have been observed on soils receiving large applications of P.

The K and Ca concentrations in grasses did not change with effluent application, and, along with P and Mg concentrations, were all at sufficient levels for optimum plant growth (Wedin, 1974).

The Na, Fe, and Mn content in grass tissues were very sensitive to effluent applications -- Na increased dramatically while Fe and Mn showed large reductions. These changes are similar to those noted for corn and were present consistently over the 4 years of this study. Unlike data for corn, B levels in the grass tissues were increased by effluent applications.

Table 6. Elemental composition of selected forage species grown following 4 years of irrigation with wastewater effluent.

Treatment	Tissue Concentration+ Reed Canary-grass	Quack-grass	Tall Fescue	Orchard-grass
	---------------	%	---------------	
		P		
Control	0.32b	0.32b	0.32b	0.32b
Low	0.43a	0.38a	0.42a	0.47a
High	0.40a	0.37a	0.42a	0.46a
		K		
Control	2.94a	3.00a	3.20a	3.59a
Low	2.74a	2.70b	2.96a	3.50a
High	2.81a	2.75b	3.14a	3.78a
		Ca		
Control	0.38a	0.38a	0.43a	0.40a
Low	0.36a	0.41a	0.45a	0.42a
High	0.41a	0.42a	0.40a	0.41a
		Mg		
Control	0.35a	0.25a	0.43a	0.39a
Low	0.30b	0.24a	0.34b	0.33b
High	0.26c	0.23a	0.28c	0.28c
	---------------	μg/g	---------------	
		Na		
Control	460c	480c	1280c	1310c
Low	2750b	2460b	3490b	3180b
High	3830a	3510a	5400a	4990a
		Fe		
Control	303a	310a	382a	270a
Low	76b	83b	75b	75b
High	87b	96b	83b	92b
		Mn		
Control	86a	80a	111a	142a
Low	48b	48b	57b	95b
High	36c	38c	48b	86b
		B		
Control	4.0c	4.0c	3.2b	3.2c
Low	8.6a	12.6a	7.4a	9.7a
High	7.3b	8.1b	6.4a	7.7b

+ Means of 4 replication for 3-time cut composite samples, except tall fescue which was a 4-time cut. Values reported on 70 C weight basis. Means within columns within elements followed by different letters are significantly different ($P \leq .05$); Duncan's multiple range test.

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AGRICULTURAL AND FOREST MANAGEMENT

FEED QUALITY OF FORAGES IRRIGATED WITH MUNICIPAL SEWAGE EFFLUENT

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ABSTRACT

We applied municipal sewage effluent to five persistent perennial forage grasses and to corn grown for silage and grain during each of three years. The effluent was applied at ≃ 5 and 10 cm/wk on corn and at higher rates on the perennial grasses throughout the growing season to supply 186 to 800 kg N/ha, depending on the year. A control treatment received 224 to 336 kg N/ha as NH_4NO_3 and ground water irrigation as needed.

Sewage effluent did not consistently influence _in vitro_ digestibility of the perennial grasses or of the corn ears, stover, and fodder. The only consistent influence of effluent application on feed value of the crops was increased concentrations of desirable crude protein when the high effluent rate was used on the perennial grasses. The Ca requirements of ruminants were met by all grass treatments, and the continuing supply of soil P by the effluent insured adequate P concentrations in the grasses to meet ruminant needs.

Long-term effluent applications caused increases in non-nutrient silica concentrations of both perennial grasses and corn stover. However, levels of silica up to 3.5% of forage dry matter did not adversely influence digestibility. This finding refutes the published inference that concentrations of forage silica in excess of 2% of dry matter deter optimum rumen digestion.

We conclude that application of very high rates of sewage effluent to perennial grasses and corn during complete growing seasons will not adversely influence feeding value of these crops for ruminant animals. Forages grown in this manner will have feeding value equivalent to or better than that of forages grown by traditional practices.

INTRODUCTION

Perennial forage grasses have been used to effectively renovate municipal sewage wastes for many years. Forages growing on land treated with nearly all of the wastes produced in Melbourne, Australia are successfully pastured by sheep and cattle; the livestock farm involved has been used in this way since 1897 (1). King and Morris (2) reported that liquid sewage sludge had little effect on the digestibility of Coastal bermudagrass, _Cynodon dactylon_ (L.) Pers., fed to steers when sludge supplied over 900 kg/ha of N and over 200 kg/ha of P; N level of the grass increased due to sludge application, but P and Ca levels did not change. When Day and Kirkpatrick (3) applied treated municipal wastewater to oats (_Avena sativa_ L.) in Arizona, forage and grain protein levels were about equal to those obtained when oats was irrigated with well water and when about 112 kg/ha of N fertilizer were applied.

To our knowledge, no comprehensive analysis has been reported of the feed quality of perennial grasses or of corn (_Zea mays_ L.) grain, stover, or fodder grown in soil receiving high levels of sewage effluent over long periods of time.

Our objective was to determine whether repeated application of high concentrations of municipal sewage effluent to perennial forage grasses and corn intended for fodder would affect the feed quality of these forages for ruminant animals.

MATERIALS AND METHODS

We selected an experimental site adjacent to the Apple Valley Wastewater Treatment Plant of the Twin Cities Metropolitan Waste Control Commission (Minneapolis-St. Paul, Minnesota) that has a Waukegan silt loam (Typic Hapludoll) soil with the water table at about 150 cm deep. This soil overlies a leached outwash gravel at about 60 cm deep. Twelve blocks of land, each about 0.1 hectare, were arranged in a 2 by 3 factorial design with two replications for comparison of perennial grasses and corn grown at each of three sewage effluent rates (based on N application):

Year	kg N/ha/year[1]				
	Control-no effluent (N supplied as NH_4NO_3 to both corn and grasses)	Low effluent		High effluent	
		Corn	Grasses	Corn	Grasses
1	224	369(136)	458(169)	604(223)	800(295)
2	336	186(109)	234(137)	337(197)	410(240)
3	336	204(112)	264(145)	397(218)	488(268)
Mean	299	253	319	446	566

[1]Cm of effluent applied are listed in parentheses.

The phosphorus applied at each effluent rate was:

Year	kg P/ha/year				
	Control-no effluent (P supplied as superphosphate to both corn and grasses)	Low effluent		High effluent	
		Corn	Grasses	Corn	Grasses
1	135	159	198	261	345
2	168	95	119	171	209
3	None	104	135	203	249
Mean	101	119	151	212	268

All plots were K-deficient in year 1. In year 2 and thereafter potassium was supplied to both control and effluent-receiving plots as needed to meet the requirements of high-producing plants (to maintain a soil test of about 300 kg/ha available K).

Two sub-replications were randomly arranged within each forage block, and each corn block had four randomly placed split plots (two representing a tillage variable and two representing a residue return variable). Because these variables did not affect forage quality, they will not be discussed in this paper. Each forage sub-rep was split to represent 2-time, 3-time, and 4-time cutting per season, and further split to include eight species. Because the 3-time cutting was most practical, only it will be included in this paper. Because alfalfa (*Medicago sativa* L.), smooth bromegrass (*Bromus inermis* Leyss.), and timothy (*Phleum pratensis* L.) did not persist on effluent-treated plots into the second and third years, only the five species that persisted well will be discussed in this report (see Table 1, footnote 2 for a listing of these species).

Laboratory tests for nutritive value during each of 3 years included *in vitro* digestible dry matter (IVDDM) via the unpublished 2-stage direct acidification method of the NC-64 Regional Committee, crude protein (CP; Kjeldahl N X 6.25), and the chemical tests of Goering and Van Soest (4). The latter included acid detergent fiber (ADF), acid detergent lignin (ADL), cell wall constituents (CWC), and plant silica. We also measured the concentrations of Ca and P by emission spectrography.

RESULTS AND DISCUSSION

Nutritive Value of Perennial Grasses

The IVDDM of the five persistent perennial grasses that received NH_4NO_3 fertilizer or sewage effluent is presented in Table 1. In the first year, the grasses treated with the high rate (H) of effluent (800 kg N/ha) were more digestible ($P \leq .05$) than were those of the control (C) getting 224 kg N/ha in the form of NH_4NO_3; those treated with the low rate (L) of effluent (458 kg N/ha) had intermediate digestibility. Grass concentrations of the two fiber components that usually are negatively associated with digestibility (ADF and ADL) did not vary among these first-year treatments (Table 2). However, the highly digestible CP fraction was higher ($P \leq .05$) in the H compared to C grasses (Table 3), and the poorly digestible CWC fraction was lower ($P \leq .05$) in the H grasses. Undoubtedly, these composition differences were reflecting the much greater N level that had been applied to the H plots (more than 3-fold greater) compared to the C plots during the first year.

In the second year, when N levels applied to C and H treatments were more comparable (336 kg/ha and 410 kg/ha, respectively), the grasses of treatment C were more digestible than those of treatment H ($P \leq .05$); treatment L grasses (234 kg N/ha) did not differ in digestibility from those of C, even though L grasses had lower CP (Table 3). The C grasses had lower concentrations ($P \leq .05$) of two incompletely digested fiber fractions (ADF, Table 2; CWC, Table 3) compared to the H grasses, which accounted for the digestibility differences.

In year three, effluent treatments did not affect IVDDM, ADF, or CWC concentration of the grasses. However, grasses of the H plots had greater CP and less indigestible ADL ($P \leq .05$) than did those of the C plots. The ADL concentration differences could not be attributed to N applications per se, because the grasses of the L plots also had lower ADL ($P \leq .05$) than did those of the C plots (even though the L grasses received only 264 kg N/ha compared to 336 kg N/ha for the C grasses). Thus, some unknown effluent effect other than N amount was operant in year three.

Examination of the three-year mean nutritive value parameters (Tables 1, 2, and 3) reveals that application of even very high amounts of sewage effluent did not adversely affect feed quality of the perennial grasses. The only consistent influence of effluent application on any of these nutritive value indices was increased concentrations of desirable crude protein when the high effluent rate (high N rate) was used.

Kentucky bluegrass frequently had a lower available energy (lower IVDDM) than the other grasses (Table 1). The highest-yielding grasses (reed canarygrass, tall fescue, and orchardgrass) had similar feeding value, except that reed canarygrass often contained more crude protein (Table 3).

According to Goering and Van Soest (4), concentrations of forage silica in excess of 2% of dry matter can deter optimal digestion by ruminant animals. Silica concentrations of the two effluent-receiving treatments (L and H) exceeded that of treatment C ($P \leq .05$) in the second and third years (Table 4), and they usually exceeded 2% of dry matter. However, comparison of silica values with IVDDM values (Table 1) reveals little relationship between these two variables. Thus, while sewage effluent application definitely caused increased silica concentrations in the grasses after the first year, apparently silica did not reach sufficiently high levels to interfere with digestibility.

Ruminants have maximum dietary requirements of about 0.2% Ca and 0.3% P (5). The Ca needs of ruminants were exceeded by all of the perennial grasses in all of the treatments, and Ca concentrations were not significantly influenced by effluent treatments or by grass species. In years 1 and 3, grasses from effluent-treated plots (L or H) contained more P ($P \leq .05$) than those from the control (Table 4). All the grasses contained sufficient P to meet the needs of high-producing ruminants, except for those of the control treatment in the third year. Fertilizer P had been applied to the control in years 1 and 2 (135 kg/ha and 168 kg/ha, respectively), but not in year 3. Thus, an advantage to use of sewage effluent on forage lands is the continuing supply of adequate P to meet ruminant requirements.

Nutritive Value of Corn

The IVDDM of corn ears, stover (above-ground plant with ears removed), and fodder (complete above-ground plant) is presented in Table 5. Ear digestibility was not influenced by effluent treatment. Control stover digestibility was lower than that of L or H treatments in year 2 and lower than that of the L treatment in year 3. Digestibility of stover from treatment H also was lower than that from treatment L in year 3. These digestibility differences were associated with ADL concentrations (Table 6), in that lower ADL accompanied higher IVDDM. The positive influence of reduced lignin on digestibility of corn stover is well known, and has been the basis for selection of low-lignin (brown-midrib) genotypes. Part of the stover IVDDM difference in year 2 could also be attributed to lower ADF in the L and H treatments (Table 6). Part of the higher stover digestibility of the L treatment in year 3 could be attributed to its lower CWC (Table 7).

The only primary treatment difference in fodder digestibility was the lower IVDDM of the control in year 2, which could be explained by the appreciably lower stover IVDDM of that treatment (Table 5). While some statistically significant ($P \leq .05$) differences appeared in the ADF and ADL of fodder due to treatments (Table 6), these differences were not of sufficient size to affect digestibility. The lower CP of treatment L fodder, compared to C or H fodder, in years 2 and 3 (Table 7) was associated with the considerably lower N application on this treatment (Materials and Methods) in those years.

When sewage effluent application influenced corn crop feed energy value, the effect was positive. As with the perennial grasses, effluent application to corn eventually caused an increase in stover silica (dry matter silica concentrations of 2.6% and 3.4% for control and the mean of L and H, respectively, in the third year). However, again this non-nutrient did not interfere with stover or fodder nutritive value. Ca concentrations were very low in corn ears (<.005%) of all treatments and more than adequate (>.3%) in corn stover of all treatments; fodder Ca was deficient (<.2%) in all treatments. P concentrations were adequate for ruminant rations in corn ears of all treatments (>.3%) and deficient in all stover and fodder (≤.25%).

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Table 1. In vitro digestible dry matter concentration of five persistent perennial forage grasses during three succeeding years of treatment with NH_4NO_3 fertilizer compared to two levels of sewage effluent (Means of three harvests per year and of four replications per harvest).[1]

Year and Species[2]	Soil Treatment			
	NH_4NO_3 (Control)[3]	Low effluent[3]	High effluent[3]	Mean
	% dry wt			
1				
Kentucky bluegrass	65.9	69.5	71.0	68.8a
Quackgrass	67.6	71.8	74.0	71.1a
Orchardgrass	67.7	72.8	74.4	71.6a
Tall fescue	64.0	66.5	68.8	66.4a
Reed canarygrass	68.0	71.2	73.4	70.9a
Mean	66.6b	70.4ab	72.3a	
2				
Kentucky bluegrass	67.6	63.6	61.7	64.3b
Quackgrass	72.5	74.4	69.0	72.0a
Orchardgrass	74.0	75.2	71.6	73.6a
Tall fescue	71.0	71.2	69.6	70.6a
Reed canarygrass	68.6	70.0	69.3	69.3ab
Mean	70.7a	70.9a	68.2b	
3				
Kentucky bluegrass	65.9	62.0	66.0	64.6b
Quackgrass	67.8	68.6	69.3	68.6a
Orchardgrass	68.4	69.8	71.2	69.8a
Tall fescue	68.7	66.8	67.5	67.7ab
Reed canarygrass	67.7	69.2	69.3	68.7a
Mean	67.7a	67.3a	68.7a	
3-year Mean	68.3	69.5	69.7	

[1]Means within rows or columns followed by different letters are different ($P \leq .05$); Duncan's range test.
[2]Latin names in order of listing: *Poa pratensis* L., *Agropyron repens* L., *Dactylis glomerata* L., *Festuca arundinacea* Schreb., and *Phalaris arundinacea* L.
[3]Hereafter designated "C", "L", and "H", respectively.

Table 2. Acid detergent fiber and acid detergent lignin concentrations of five persistent perennial forage grasses during three succeeding years of treatment with NH_4NO_3 fertilizer compared to two levels of sewage effluent (Means of three harvests per year and of four replications per harvest).[1]

Year and Species	Acid detergent fiber				Acid detergent lignin			
	Soil treatment				Soil treatment			
	C	L	H	Mean	C	L	H	Mean
	% dry wt							
1								
Kentucky bluegrass	35.6	35.4	34.6	35.2a	3.9	3.8	3.4	3.7ab
Quackgrass	36.0	33.5	33.9	34.5a	4.3	4.1	4.2	4.2a
Orchardgrass	38.0	36.4	36.2	36.9a	4.0	3.8	3.5	3.8ab
Tall fescue	36.7	37.4	38.3	37.5a	3.4	3.4	3.4	3.4b
Reed canarygrass	35.7	34.0	34.3	34.7a	3.6	3.5	3.2	3.4b
Mean	36.4a	35.3a	35.5a		3.8a	3.7a	3.5a	
2								
Kentucky bluegrass	35.2	37.6	37.4	36.7a	3.8	4.0	3.8	3.9a
Quackgrass	33.6	33.2	35.7	34.1c	4.3	3.7	4.1	4.0a
Orchardgrass	36.4	36.0	37.2	36.5a	3.5	3.3	3.4	3.4b
Tall fescue	33.8	35.9	35.6	35.1bc	2.8	2.7	3.0	2.8c
Reed canarygrass	34.9	36.2	36.5	35.9ab	3.8	3.3	3.1	3.4b
Mean	34.8b	35.8ab	36.5a		3.6a	3.4a	3.5a	
3								
Kentucky bluegrass	36.0	37.7	37.0	36.9b	4.1	3.8	3.4	3.8ab
Quackgrass	36.8	36.0	36.5	36.4b	4.3	3.9	4.2	4.1a
Orchardgrass	38.2	37.6	38.1	38.0a	4.0	3.5	3.3	3.6b
Tall fescue	35.0	37.8	36.7	36.5b	3.5	3.2	3.4	3.4b
Reed canarygrass	37.5	36.9	37.0	37.1ab	3.9	3.4	3.4	3.6b
Mean	36.7a	37.2a	37.1a		4.0a	3.6b	3.5b	
3-year Mean	35.8	36.1	36.4		3.8	3.6	3.5	

[1]Means within rows or within columns followed by different letters are significantly different ($P \leq .05$); Duncan's multiple range test.

Table 3. Crude protein and cell wall constituent concentrations of five persistent perennial forage grasses during three succeeding years of treatment with NH_4NO_3 fertilizer compared to two levels of sewage effluent (Means of three harvests per year and of four replications per harvest).[1]

Year and Species	Crude protein, Soil treatment: C	L	H	Mean	Cell wall constituents, Soil treatment: C	L	H	Mean
	% dry wt							
1								
Kentucky bluegrass	19.1	19.5	21.4	20.0ab	55.8	54.8	52.3	54.3c
Quackgrass	19.2	20.7	23.0	21.0ab	57.0	53.1	52.0	54.0c
Orchardgrass	17.9	18.4	20.4	18.9bc	57.4	56.3	52.4	55.4bc
Tall fescue	16.2	16.7	18.8	17.2c	57.4	58.2	55.6	57.1a
Reed canarygrass	19.8	21.1	23.4	21.4a	57.1	53.6	53.3	54.7c
Mean	18.4b	19.3b	21.4a		56.9a	55.2a	53.1b	
2								
Kentucky bluegrass	16.4	13.0	15.0	14.8b	56.3	59.0	58.9	58.1a
Quackgrass	16.3	15.6	16.6	16.2a	53.7	53.1	55.0	53.9c
Orchardgrass	14.9	12.9	14.6	14.1c	55.4	53.9	56.9	55.4b
Tall fescue	13.4	12.0	17.3	14.2c	54.6	56.0	55.3	55.3b
Reed canarygrass	16.1	14.8	16.9	15.9a	55.6	57.1	56.9	56.5b
Mean	15.4a	13.7b	16.1a		55.1b	55.9ab	56.6a	
3								
Kentucky bluegrass	15.0	14.6	15.2	14.9b	60.0	61.6	59.7	60.4a
Quackgrass	13.9	14.7	16.3	15.0b	61.9	58.7	59.3	60.0a
Orchardgrass	12.8	11.8	14.6	13.1c	60.8	59.0	59.5	59.8a
Tall fescue	13.3	11.8	14.4	13.2c	59.3	61.8	60.0	60.4a
Reed canarygrass	14.9	15.4	17.1	15.8a	61.6	61.6	60.2	61.1a
Mean	14.0b	13.7b	15.5a		60.7a	60.5a	59.7a	
3-year Mean	15.9	15.6	17.7		57.6	57.2	56.5	

[1] Means within rows or within columns followed by different letters are significantly different ($P \leq .05$); Duncan's multiple range test.

Table 4. Silica and phosphorus concentrations of five persistent perennial forage grasses during three succeeding years of treatment with NH_4NO_3 fertilizer compared to two levels of sewage effluent (Means of three harvests per year and of four replications per harvest).[1]

Year and Species	Silica, Soil treatment: C	L	H	Mean	Phosphorus, Soil treatment: C	L	H	Mean
	% dry wt							
1								
Kentucky bluegrass	3.0	3.2	3.0	3.1a	0.39	0.48	0.52	0.46a
Quackgrass	2.1	2.4	2.3	2.3c	0.38	0.51	0.57	0.49a
Orchardgrass	1.8	2.0	2.1	2.0d	0.36	0.51	0.54	0.47a
Tall fescue	2.6	2.7	2.6	2.6b	0.36	0.48	0.53	0.46a
Reed canarygrass	2.3	2.8	2.6	2.6b	0.38	0.51	0.55	0.48a
Mean	2.4a	2.6a	2.5a		0.37b	0.50a	0.54a	
2								
Kentucky bluegrass	2.4	3.1	3.4	3.0a	0.38	0.36	0.39	0.38a
Quackgrass	1.6	2.1	2.6	2.1c	0.46	0.41	0.45	0.44a
Orchardgrass	1.8	2.5	2.9	2.4b	0.41	0.40	0.44	0.42a
Tall fescue	2.2	3.5	3.1	2.9a	0.40	0.40	0.38	0.39a
Reed canarygrass	2.4	3.2	3.4	3.0a	0.42	0.42	0.47	0.44a
Mean	2.1b	2.9a	3.1a		0.41a	0.40a	0.43a	
3								
Kentucky bluegrass	1.9	3.2	2.8	2.6a	0.26	0.31	0.32	0.30a
Quackgrass	1.8	2.1	2.2	2.0b	0.27	0.32	0.35	0.31a
Orchardgrass	1.5	2.3	2.5	2.1b	0.25	0.36	0.36	0.32a
Tall fescue	1.6	2.9	2.5	2.3a	0.26	0.32	0.32	0.30a
Reed canarygrass	1.9	2.7	2.7	2.4a	0.26	0.34	0.35	0.32a
Mean	1.7b	2.6a	2.5a		0.26b	0.33a	0.34a	
3-year Mean	2.1	2.7	2.7		0.35	0.41	0.44	

[1] Means within rows or within columns followed by different letters are significantly different ($P \leq .05$); Duncan's multiple range test.

Table 5. *In vitro* digestible dry matter concentration of corn ears, stover, and fodder during three succeeding years of treatment with NH_4NO_3 fertilizer compared to two levels of sewage effluent (Means of 2 replications, 2 tillage methods, and 2 residue return methods).[1]

Plant part and year	Soil treatment: NH_4NO_3 (Control)	Low effluent	High effluent
Ear	% dry wt		
1	85.1a	83.8a	85.3a
2	93.7a	94.2a	94.0a
3	87.0a	88.0a	88.6a
Mean	88.6	88.7	89.3
Stover			
1	52.8a	53.7a	53.9a
2	58.1b	66.3a	65.3a
3	55.0b	58.5a	56.7b
Mean	55.3	59.5	58.6
Fodder			
1	68.2a	67.5a	67.9a
2	77.6b	80.5a	80.8a
3	74.1a	75.7a	76.0a
Mean	73.3	74.6	74.9

[1] Means within rows within years followed by different letters are significantly different ($P \leq .05$); Duncan's multiple range test.

Table 6. Acid detergent fiber and acid detergent lignin concentrations of corn ears, stover, and fodder during three succeeding years of treatment with NH_4NO_3 fertilizer compared to two levels of sewage effluent (Means of 2 replications, 2 tillage methods, and 2 residue return methods).[1]

Plant part and year	Acid detergent fiber, Soil treatment: C	L	H	Acid detergent lignin, Soil treatment: C	L	H
Ear	% dry wt					
1	11.9a	12.3a	12.0a	2.4a	2.5a	2.4a
2	6.9a	6.9a	7.2a	1.2a	1.2a	1.2a
3	9.9a	9.3a	9.4a	1.8a	1.6c	1.7b
Mean	9.6	9.5	9.5	1.8	1.8	1.8
Stover						
1	45.9a	45.4a	45.6a	5.0a	4.7a	5.0a
2	40.9a	38.1b	35.6c	4.0a	3.0b	3.3ab
3	45.2a	43.7a	45.2a	4.3a	3.5b	4.4a
Mean	44.0	42.4	42.1	4.4	3.7	4.2
Fodder						
1	29.6c	30.2b	30.6a	3.7a	3.7a	3.8a
2	22.2a	22.2a	21.6a	2.5a	2.1a	2.2a
3	24.2a	23.7b	23.1c	2.8a	2.4b	2.7a
Mean	25.3	25.2	25.1	3.0	2.7	2.9

[1] Means within rows within years followed by different letters are significantly different ($P \leq .05$); Duncan's multiple range test.

Table 7. Crude protein and cell wall constituent concentrations of corn ears, stover, and fodder during three succeeding years of treatment with NH_4NO_3 fertilizer compared to two levels of sewage effluent (Means of 2 replications, 2 tillage methods, and 2 residue return methods).[1]

Plant part and year	Crude protein			Cell wall constituents		
	Soil treatment			Soil treatment		
	C	L	H	C	L	H
	% dry wt					
Ear						
1	8.1b	8.4ab	9.0a	31.0a	31.0a	31.0a
2	8.8a	7.7a	8.4a	21.0a	21.0a	22.0a
3	8.5a	7.0b	8.3a	24.2a	22.7a	23.9a
Mean	8.5	7.7	8.6	25.4	24.9	25.6
Stover						
1	7.6a	7.7a	7.8a	72.3a	70.6b	70.2b
2	6.8a	5.3b	6.0b	65.2a	63.9a	63.8a
3	4.5a	3.5a	4.7a	72.8a	69.9b	71.8ab
Mean	6.3	5.5	6.2	70.1	68.1	68.6
Fodder						
1	7.8a	8.0a	8.3a	52.5a	52.4a	52.7a
2	7.9a	6.5b	7.3ab	40.9a	42.1a	41.2a
3	6.9a	5.6b	6.9a	43.8a	42.5a	42.2a
Mean	7.5	6.7	7.5	45.7	45.7	45.4

[1]Means within rows within years followed by different letters are significantly different ($P \leq .05$); Duncan's multiple range test.

AGRICULTURAL AND FOREST MANAGEMENT

THE ROLE OF PONDS IN LAND TREATMENT OF WASTEWATER

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ABSTRACT

Ponds used for storage or pretreatment of wastewater prior to terrestrial application significantly alter wastewater quality. Data from the Michigan State University Water Quality Management Facility (WQMF) indicate that significant permanent phosphorus reduction decreases markedly once benthic sediments are saturated with phosphorus but that excellent nitrogen removal continues.

Phosphorus concentrations within such ponds vary seasonally but significant permanent phosphorus removal should not be anticipated beyond the first two or three years of operation. Harvest of aquatic plants yields only about 10% removal of both phosphorus and nitrogen.

In wastewater ponds, aquatic plant growth leads to accelerated denitrification and increases pH to 10 or more causing large ammonia losses to the air during those periods of the year when algae and aquatic macrophytes are active. While this nitrogen loss to the atmosphere will be controlled by site-specific conditions of light availability and temperature and alkalinity of the wastewater, nitrogen concentration decreases in a logistic fashion as a function of detention time within the WQMF ponds yielding a 95% nitrogen reduction in 120 days. The rate of nitrogen removal, as $gN/M^2/day$, from these ponds is a linear function of nitrogen concentration.

In addition to marked nitrogen losses associated with aquatic plant growth, the ability of these plants to elevate the pH also leads to precipitation of a variety of heavy metals. Such alterations in wastewater should be included in design criteria whenever ponds are incorporated in land treatment of wastewater.

INTRODUCTION

In most systems designed for land treatment of wastewater, wastewater is impounded for some period of time prior to terrestrial application. Such impoundment may range from a series of stabilization ponds for pretreatment of wastewater before it is applied to the land to large storage basins to hold wastewater during those periods of the year when terrestrial application is impractical. Regardless of the form of and reason for impoundment, a variety of changes in wastewater quality occur under the auspices of the aquatic ecosystem within detention basins. These alterations in wastewater quality may limit some potential benefits expected from terrestrial wastewater application but also offer design flexibility if they are recognized and incorporated into the system design.

This discussion deals with both benefits offered and limitations imposed by impoundment of wastewater prior to land application based largely upon data obtained from the Michigan State University Water Quality Management Facility (WQMF).

THE WATER QUALITY MANAGEMENT FACILITY

The WQMF was constructed to allow evaluation and demonstration of the wastewater treatment and nutrient recycle capabilities of aquatic and terrestrial ecosystems both singly and in combination. Secondary wastewater effluent is conveyed at a rate of 1892 m^3/day 7.25 km from the East Lansing, Michigan, activated sludge wastewater treatment plant to the 200 ha WQMF on the southern margin of the Michigan State University campus. There the wastewater flows by gravity through a series of four ponds to a pumphouse from which it is applied as spray irrigation to the 130 ha terrestrial site.

These ponds range from 3.23 to 4.98 ha with a total surface area of 16 ha. Maximum pond depth is 2.4 m at the outlet and mean operating depth of each pond is 1.8 m to place the entire pond bottom within the euphotic zone to encourage growth of aquatic macrophytes. During construction, the ponds were sealed with native clay yielding a low permeability of 0.18 cm/day.

The outlet structure of each pond allows withdrawal of water from the top, bottom, and mid-depth. Mixture of incoming secondary effluent with water from each of three depths from each of the four ponds at the pumphouse allows a wide variety of wastewater quality for terrestrial application.

CARBON DYNAMICS

Ponds charged with wastewater are extremely dynamic systems. The immediate reaction to the addition of nutrient-rich wastewater to ponds is rapid photosynthetic carbon fixation by aquatic plants at rates as high as 24 mg carbon/l/day in ponds receiving raw sewage (1). Such marked carbon uptake rates exceed the rate of recarbonation from the air and significant amounts of carbon dioxide are drawn from the alkalinity system governed by the first and second dissociations of carbonic acid as shown in equations 1, 2, and 3.

$$HCO_3^- + H^+ \rightleftharpoons CO_2 + HOH \qquad (1)$$

$$HCO_3^- \rightleftharpoons CO_3^= + H^+ \qquad (2)$$

$$2\ HCO_3^- \rightleftharpoons CO_2 + CO_3^= + HOH \qquad (3)$$

Such carbon removal results in all three of these reactions moving to the right with an increase in $CO_3^=$, a decrease in HCO_3^- and an increase in pH as shown for the second dissociation in equations 4 and 5.

$$K_2 = \frac{[H^+]\ [CO_3^=]}{[HCO_3^-]} \qquad (4)$$

$$[H^+] = \frac{K_2\ [HCO_3^-]}{[CO_3^=]} \qquad (5)$$

This rise in pH is accompanied by a sharp decrease in free carbon dioxide concentration (1), precipitation of a variety of metal oxy-hydroxides and carbonates (2) and a loss of ammonia to the atmosphere as shown by the equilibrium in equation 6 for which the pK is about 9.2 at summer temperatures (3).

$$NH_4^+ \rightleftharpoons NH_{3\,(g)} + H^+ \qquad (6)$$

Thus, the addition of nutrient-rich wastewater sets in motion a wide variety of changes in water chemistry all mediated by aquatic plant response to such nutrients. However, such plant induced changes in chemical equilibria feed back on the plants to limit both total photosynthetic rate and the type of plants which can exist in the ponds.

Wastewater nutrient content varies considerably, depending on the source. The secondary effluent being discharged from the East Lansing, Michigan, activated sludge system to the WQMF contains about 20 mg nitrogen/l and 5 mg phosphorus/l with an alkalinity of about 3 meq/l. At an atomic ratio for aquatic plants of $C_{100}N_{16}P_1$ each liter of this effluent contains enough phosphorus to allow plant fixation of about 190 mg of carbon, enough nitrogen for about 100 mg of carbon and enough carbon for about 40 mg of carbon fixation. Clearly, carbon is the nutrient in shortest supply relative to photosynthetic activity in ponds receiving such effluent.

Carbon is supplied to the plants from the alkalinity system, from respiratory activities within the pond and from the atmosphere. That supplied by the alkalinity is limited by the alkalinity of the source water (4) while the carbon supplied from the atmosphere is an inverse function of the carbon dioxide concentration of the water (5). Carbon supplied from respiration is determined by the amount of respiratory biomass within the pond but even those ponds receiving raw sewage experience severe carbon shortage during summer months (1)(6)(7).

When wastewater is impounded, plant photosynthesis, unlimited by nitrogen and phosphorus, responds to available sunlight and begins extracting free carbon dioxide (2). As this carbon dioxide is fixed, the alkalinity system adjusts to the new equilibrium as indicated in equations 1, 2, and 3. The pH begins to rise and the free carbon dioxide concentration decreases. The rate of change in this equilibrium is determined largely by the rate of photosynthesis but the extent of change, and thus the maximum pH and minimum free carbon dioxide concentration, is related directly to the time of detention of the wastewater in the pond (2) relative to available sunlight and water temperature.

The specific growth rate of aquatic plants is related in a kinetic fashion to the free carbon dioxide concentration of the water (8)(9) such that as free carbon dioxide decreases plant photosynthetic rate decreases. In addition, the intensity of available light interacts with available free carbon dioxide as a direct multiple to limit photosynthesis by aquatic plants (10)(11). The various aquatic plant species each have a different kinetic response to those interacting limits but with increased detention time and the resulting extremely low free carbon dioxide concentrations the desirable green algae sink faster than they grow (12)(13). At that point the advantage passes to those plants which can maintain their active biomass in the euphotic zone. The plants favored under these conditions are macrophytes and bluegreen algae with their pseudovacueles which allow them to maintain bouyancy. In shallow ponds the factors determining which of these latter plant forms will dominate appear to include detention time, carbon, light and nitrogen availability and the relative aerobic nature of the bottom sediments (2)(14)(15).

Regardless of the photosynthetic dominant, the pH of the water in ponds charged with wastewater increases as a direct function of detention time (2) during that period of the year when available light and existing temperature allow aquatic plant photosynthesis. The resulting elevation of pH causes significant changes in other equilibria which bring about marked alterations in other water quality parameters (2). Of particular interest in cleansing wastewater and recycling wastewater nutrients are the potentials for change in the phosphorus and nitrogen content generated by such elevation in pH.

Within the WQMF, pH routinely exceeds 10 in the three downstream ponds during the summer months and, during the summer of 1976, the macrophyte, Elodea canadensis, maintained a pH well above 10 within the second pond for several weeks with no attempt to manage vegetative growth.

PHOSPHORUS REMOVAL POTENTIAL

Phosphorus is removed from wastewater in ponds by direct sorbtion on to bottom sediments, as metal precipitates and by incorporation into biological tissue.

All bottom sediments have a finite capacity to sorb phosphorus with the amount sorbed being dependent on the composition of the sediments and the water chemistry (16). In ponds receiving domestic wastewater, the initial significant reduction in the phosphorus content of wastewater will decline as the bottom sediments become loaded with phosphorus. An example of this loading is seen in the first of the four WQMF ponds which at times during the second summer of operation had an effluent with a higher phosphorus content than the influent. With continued throughput of wastewater the phosphorus content of the downstream ponds increased. Once the bottom sediments of these ponds become saturated with phosphorus, no further phosphorus removal from the wastewater can be expected from this mechanism. Thus, phosphorus sorption on bottom sediments of ponds cannot be counted on for long-term phosphorus removal from wastewater.

With the high pH levels generated in wastewater ponds by photosynthetic activity, it is tempting to picture significant phosphorus removal as precipitates of various calcium phosphates. However, the elevated pH is accompanied by increased concentrations of carbonate ion which successfully compete with the phosphate for the calcium. The shortage of available cations and the generally more favorable precipitation of carbonate markedly limit precipitation of phosphorus as a calcium salt in wastewater ponds. Nucleation and substitution of the phosphate to form hydroxyapatite appears to require a longer period of time than normally would be considered a reasonable detention period for wastewater (17).

Precipitation of iron and aluminum phosphates undoubtedly occurs, but iron and aluminum concentrations in most wastewater would not allow significant phosphate precipitation. In addition, bacterial respiration of organics at the bottom of such ponds will alter water chemistry such that recycle of the iron bound precipitates will occur (18). Generally, then, precipitation of phosphorus as calcium, iron and aluminum salts cannot be counted on to remove appreciable amounts of phosphorus in wastewater ponds over a sustained period without the addition of significant concentrations of available cations.

Harvest of biologically concentrated phosphorus is the only other mechanism which offers any real potential for removing phosphorus from wastewater ponds. Here the greatest potential lies in the harvest of plant tissue prior to the loss of photosynthetically concentrated phosphorus associated with the use of the plants by other aquatic biota. For all practical purposes such harvest is most efficient if directed at macrophytes and large periphytic algal masses since planktonic algae are difficult and expensive to remove (19).

Removal of phosphorus by harvesting plant biomass presupposes significant phosphorus uptake by the plants which in turn is predicated on rapid plant growth. Within wastewater the growth rate of the aquatic plant is most often limited by the availability of a photosynthetic carbon source (1).

During 1976, 3151 Kg dry weight of floating Cladophora fracta masses were harvested from the first WQMF pond while Elodea canadensis harvest was 9223 Kg dry weight from the second pond and 6554 Kg dry weight from the fourth pond. The phosphorus concentration of the Cladophora, including the appreciable sediment incorporated with it, was 1.60% of dry weight. The phosphorus concentration of Elodea was 0.65% dry weight in pond 2 and 0.33% dry weight in pond 4 with the difference reflecting the much higher phosphorus concentration in the water of the second pond during the summer of 1976. The total amount of phosphorus removed from the WQMF ponds during 1976 by plant harvest was 132 Kg as elemental phosphorus. At a loading rate of 1892 m^3/day for a 365 day period at a phosphorus concentration of 5 mg P/1, the ponds would receive 3545 Kg of phosphorus per year. The phosphorus removal associated with plant harvest from the WQMF ponds amounted to 3.8% of the annual load or 7.6% of the phosphorus entering the lakes during a 180 day summer period.

Population estimates of aquatic plants just before harvest indicated that about 50% of the Elodea population was removed from the second pond and about 25% was harvested from the fourth pond. No population estimates are available from the first pond but the Cladophora ceased appreciable growth when the water temperature reached about 23°C. The harvest from the second pond represented a maximal effort while that from the fourth pond was the result of a lesser effort. The third pond was not harvested.

However, even if all available plants had been removed from all four ponds, the phosphorus removed would have been about 10% of the annual phosphorus loading or 20% of the 180 day summer loading. For the summer period, the 20% projected maximum would equal a removal of one mg P/1.

The absence of significant precipitation of phosphorus and the limited ability of aquatic plants to remove phosphorus from wastewater leave little chance of wastewater ponds meeting phosphorus discharge standards such as those currently in effect in Michigan after the pond bottom sediments become saturated with phosphorus. Detention times required for sufficient phosphorus removal from wastewater far exceed any reasonable period. Thus, after a period of operation sufficient to saturate bottom sediments with phosphorus, storage of wastewater prior to land application will offer little phosphorus reduction.

WATER SOFTENING

The abundance of phosphorus in ponds charged with wastewater insures that phosphorus will not limit biotic activity. Potential limits to plant production in such ponds include carbon, nitrogen, light availability, and water temperature.

The carbon limit is the first to be reached by the plants and with carbon uptake from the alkalinity both pH and carbonate concentration increase. This leads to precipitation of a variety of carbonates and oxy-hydroxides but such precipitation does not represent permanent removal. Periods of respiration which may occur at any time of year but particularly during the winter sharply decrease pH leading to marked dissolution of these precipitates. Thus, reductions in water hardness observed in such ponds during some periods are offset by increases in concentration of those cations which contribute to hardness of water during other periods.

NITROGEN REMOVAL

Significant amounts of nitrogen are lost to the air from wastewater ponds as both ammonia gas and nitrogen gas resulting from denitrification. The amount of each form lost is dependent upon the type of wastewater added, the depth of the pond and the time of year.

During periods of elevated pH, the ammonium ion dissociates to free ammonia gas as was shown in equation 6. Loss of this gas to the atmosphere leads to continued dissociation of ammonium nitrogen and to decreased nitrogen levels in the water. The ability to maintain ammonia nitrogen in wastewater ponds is thus a function of the pH of water shown in Figure 1.

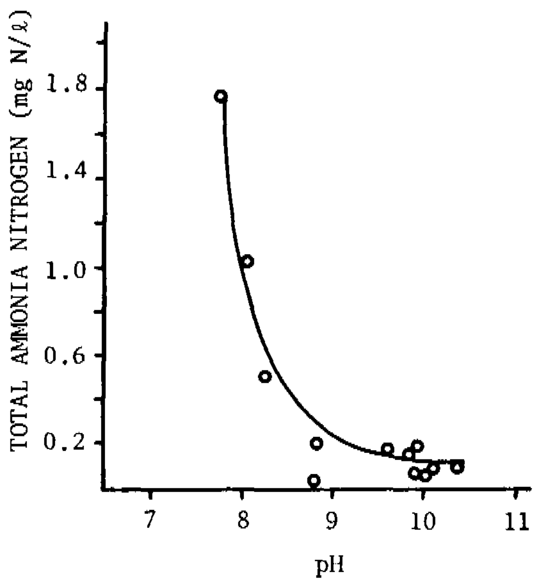

Figure 1. Relationship Between Ammonia Concentration and pH in the Second WQMF Pond During 1976.

Obviously, the rate of nitrogen loss as ammonia is determined by interactions between the degree of wind mixing, the depth of the pond, the water temperature, the rate of nitrification, the rate of ammonia uptake by aquatic plants, and by the rate of ammonia production within the pond. However, much of the nitrogen loss from the WQMF lakes during 1976 shown in Figure 2 must be ascribed to this process since the amount of nitrogen removed cannot be accounted for in either plant tissue or by denitrification.

While it is not possible to obtain accurate estimates of planktonic and filamentous algae which dominated the first pond during 1976, only 4% of the nitrogen removed from the wastewater in this pond was accounted for by harvest

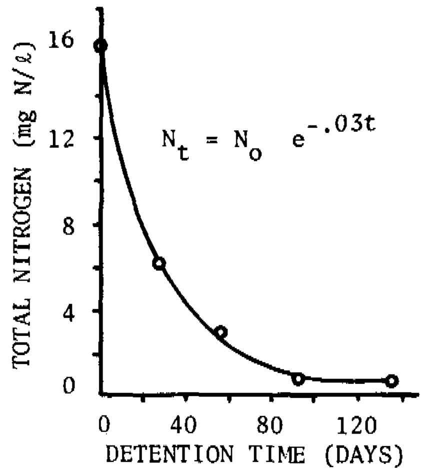

Figure 2. Average Total Nitrogen Concentration in Wastewater as a Function of Detention Time in the WQMF Ponds During 1976.

of Cladophora. Harvest of Elodea accounted for 22% of the nitrogen removed in pond 2 and 90% of the nitrogen removed in pond 4. The third pond was not harvested. Only 9% of the total nitrogen removed during the period from June through October of 1976 can be accounted for in plant harvest from the WQMF ponds.

During the occasional severe respiratory events in the WQMF ponds, denitrification may represent an important nitrogen loss but the supersaturated oxygen levels and the high pH maintained throughout much of the warm season would discourage denitrification. In those ponds charged with raw wastewater, denitrification would play a greater role in nitrogen loss.

The rapid cycling of nitrogen through the plants to the bacteria by both bacterial decay of plant biomass and bacterial use of nitrogenous materials leaked from the plants (20)(21) allows considerable ammonia production. Maintenance of pH values in excess of the pK for the ammonium ion dissociation and the rapid wind mixing of these ponds allows significant amounts of ammonia to be lost to the air. Clearly this loss is directly related to the pH of the water, which in turn is related to the type of waste the pond is charged with and the time the wastewater is retained in the pond.

If the pond is charged with secondary effluent, as is the WQMF, little organic carbon enters the lake and little carbon dioxide is supplied by bacterial respiration of this material. In this case plant photosynthetic carbon supply is met by the carbon dioxide from the alkalinity system and by recarbonation from the atmosphere with the rate of recarbonation being inversely related to the free carbon dioxide concentration in the water. As the plants extract carbon from the alkalinity, the free carbon dioxide level falls and pH rises. The longer the wastewater is retained in the lake, the higher the pH will rise until the free carbon dioxide concentration is decreased to the level where the free carbon dioxide limited photosynthetic carbon uptake rate of the most competitive plant is equal to the atmospheric recarbonation rate. At this point the maximum pH will be reached and, while longer detention times will not yield a higher pH value, they will allow continued loss of ammonia to the air.

The relationship between remaining total nitrogen concentration and detention time is shown in the data in Figure 2 which yields equation 7.

$$Nt = No\ e^{-.03t} \qquad (7)$$

where: Nt = total nitrogen concentration (mg N/1) at time t
No = initial total nitrogen concentration (mg N/1)
t = days

As shown in Figure 3, the rate of nitrogen removal from the WQMF ponds from June through October of 1976 is a linear function of the influent inorganic nitrogen concentration which can be described by equation 8.

$$N_R = -0.017 + 0.038\ N_c \qquad (8)$$

where: N_R = rate of nitrogen removal (g $N/m^2/day$)
N_c = influent inorganic nitrogen concentration (mg N/1)

Combination of equations 7 and 8 to yield equation 9 suggests that the rate of nitrogen removal by shallow ponds charged with secondary effluent is largely a function of detention time when available light and temperature allow aquatic photosynthesis.

$$N_R = -0.017 + 0.038\ (No\ e^{-.03t}) \qquad (9)$$

If the pond receives raw sewage, the considerable carbon dioxide released by bacterial respiration of the organic fraction represents another carbon source to the plants, and extraction of carbon

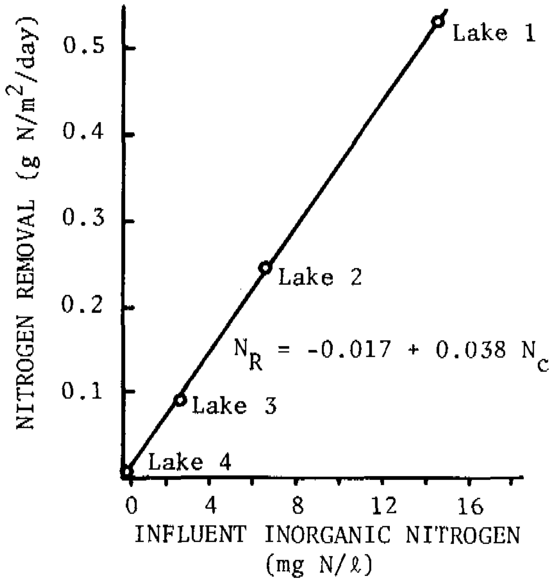

Figure 3. The Rate of Nitrogen Removal in the WQMF Ponds During 1976 as a Function of Incoming Nitrogen Concentrations.

from the alkalinity and thus pH rise will be slowed. Under these conditions a longer detention time will be required to reach the maximum pH level, but the rate of denitrification would be increased.

Given sufficient detention time the biotic communities of ponds charged with wastewater are extremely efficient at stripping nitrogen from wastewater. During the period from June to October, 1976, nitrogen removal within the WQMF ponds amounts to 95% with only 9% of the total nitrogen removal being accounted for by the harvest of aquatic vegetation from the lakes.

DESIGN CONSIDERATIONS

While there is little likelihood that the aquatic ecosystem can reduce the phosphorus concentration of municipal wastewater sufficiently to meet effluent phosphorus standards, it is extremely efficient at stripping nitrogen from wastewater. This nitrogen removal, a direct response to aquatic plant photosynthesis in a carbon limited system, offers real opportunity for the production of a low nitrogen wastewater which can be applied to terrestrial systems when terrestrial plants do not require nitrogen with no fear of contaminating groundwater with nitrate. Since phosphorus is removed by soil sorbtion, this allows significant extension of the irrigation season and a significant decrease in the total acreage required for irrigation and storage of wastewater.

In those systems aimed at maximizing nitrogen recycle, ponding of wastewater should be minimized, particularly during the summer months to prevent the loss of nitrogen to the air prior to terrestrial application. Where wastewater is stored over the winter and applied to the land during the summer, considerable nitrogen losses will occur with the amount of loss increasing as a function of wastewater detention time particularly during the warm months of the year. This nitrogen loss should be considered when such systems are designed and where a series of ponds is used the greatest nitrogen recycle can be accomplished by withdrawing wastewater for land application from as far upstream in the pond series as possible.

ACKNOWLEDGEMENTS

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AGRICULTURAL AND FOREST MANAGEMENT

USE OF NATURAL TERRESTRIAL VEGETATION FOR RENOVATION OF WASTEWATER IN MICHIGAN

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ABSTRACT

Secondary municipal effluent has been applied to both abandoned farm fields and to unmanaged beech-sugar maple forests for the past two years on the Water Quality Management Facility at Michigan State University. Mass balances for phosphorus, nitrogen, and chloride have been constructed. Old-field treatments included unirrigated control areas and applications of 5 and 10 cm/week of spray irrigation. Within each of these three treatments, two replicated plots were unharvested, two were harvested only in June and two others were harvested in both June and September. Mass balance data for the 1976 and 1977 growing seasons indicate that abandoned farm fields will effectively renovate wastewater with proper management. Harvesting extends the active growing season and results in continued N and P uptake through October.

Forest wastewater application studies have included unirrigated control areas and 5 and 10 cm/week application areas. Results from the 1976 and 1977 growing seasons indicated that output leachate concentrations of nitrate-nitrogen often approached or exceeded input concentrations on the 5 cm/week application area. Leachate concentrations on the 10 cm/week application area were much lower, perhaps due to denitrification in these water-logged soils. Runoff from this site, however, often exceeded discharge standards.

Management implications of the 1976 and 1977 data for these forests and old-field wastewater application studies are discussed.

INTRODUCTION

The use of natural vegetation for renovation of secondary municipal effluent has been studied since 1975 as part of the land application studies associated with the Water Quality Management Facility (WQMF) at Michigan State University. The basic premise of these land application studies is that a variety of management alternatives must be developed so that the designer and operator of any particular land treatment facility can select that management alternative that best meets the needs of the local facility. Thus, a variety of crops and cropping practices, a variety of schemes for use of non-cropped areas, plus a variety of spray application rates, harvest schemes, and plant species and varieties have been included in the studies at Michigan State University. Much of these data are in manuscript form and have been or will be submitted for publication in the near future (Brockway et al., 1978; Leland et al., 1978; and several unpublished manuscripts by J. E. Hook and M. B. Tesar, J. E. Hook and T. M. Burton, T. M. Burton and J. E. Hook, and others). The purpose of this paper is to summarize the research on natural or non-agronomic vegetation that has been done on the WQMF at Michigan State University.

MATERIALS AND METHODS

The Water Quality Management Facility (WQMF) consists of a section of the newly constructed East Lansing, Michigan, activated sludge sewage treatment plant that can be operated independently of the main tertiary plant; a 7.25 km, 53 cm transmission line; four 1.8 m deep lakes with a combined surface area of 16 ha; and a 58 ha land irrigation site (130 ha including aerosol buffer zones). While this system was supposedly designed to handle 7,570 m^3/day (2 MGD) of secondary effluent from the East Lansing plant, winter storage requirements limit operation on a year-round basis to 1892 m^3/day (0.5 MGD). The system was operated with 1892 m^3/day of poor quality tertiary effluent from the old East Lansing plant in 1974 and 1975. Changeover to the new East Lansing plant was made during the winter of 1975 forcing a shutdown of the WQMF. It started operating with 1892 m^3/day of secondary wastewater in April 1976, and has been operated at that level since that time. The land research reported here includes results from 1976 and 1977 when the system was operated with secondary, municipal effluent. All land irrigation during this time was with secondary effluent taken directly from the transmission line from East Lansing or a combination of wastewater directly from East Lansing and water that backsiphoned out of the first receiving lake when the irrigation pumping rate exceeded the flow from East Lansing. The effluent was chlorinated before irrigation.

The land irrigation site consists of a 58 ha irrigation site surrounded by a 72 ha aerosol buffer zone. The soils on this site are very diverse and change rapidly both horizontally and vertically. In general, the predominant soils are loams, loamy sands, or sandy loams of the mixed mesic family of Typic Hapludalfs. The Miami, Marlette, Brookston, and Fox loams are the most common soils on the site. The average hydraulic limit for the site is 5 cm/week but varies from 0 to 20 cm/week.

The two primary types of research on non-cropped systems have included various harvest and irrigation schemes for oldfield successional ecosystems and the use of late successional sugar maple-beech forests for renovation of wastewater. The oldfield research has included two sites. One oldfield site was used for investigation of various harvest schemes and irrigation rates on renovation of wastewater during the growing season. A second site was used for studies of the feasibility of winter irrigation in Michigan (Leland _et al._, 1978). A third site was used for the forest studies. Each of these sites is described below.

The oldfield research was conducted on abandoned farm lands that had been in corn cultivation approximately 10-12 years prior to these studies. The volunteer vegetation on the two experimental oldfield sites was dominated by _Solidago_ sp., _Agropyron repens_, _Taraxacum officinale_, _Aster_ sp., and _Poa compressa_ but included a diverse flora of minor species. In general, the grasses, _Agropyron repens_ and _Poa compressa_, and the dandelion, _Taraxacum officinale_, dominated growth early in the season (April to mid-June) with _Solidago_ sp. overlapping them and becoming dominant by early June and with _Aster_ sp. becoming important late in the growing season.

The oldfield site used for harvest and irrigation rate studies was divided into six experimental blocks of six 0.07 ha plots each (36 total plots). Four of these blocks (24 plots) were irrigated with 5 or 10 cm/week of wastewater from mid-April to mid-October in 1976 and 1977 and two of them served as unirrigated controls. Within each block, harvest treatments were assigned randomly; these treatments included no harvest--5 cm/week, no harvest--10 cm/week, one harvest in June--5 cm/week, one harvest in June--10 cm/week, two harvests in June and September--5 cm/week, and two harvests--10 cm/week.

Soil and soil-water were sampled from a small area in the center of each plot. The soil-water was sampled with porous cup vacuum tube type lysimeters placed at the 15 and 120 cm depths and evacuated to 0.8 atm. 48 hours prior to sampling and just before one of the two weekly applications of wastewater. The winter spray site was irrigated year round at 5 cm/week and was sampled at 45, 90, and 150 cm depths at 9 lysimeter sites within the irrigation area and at 4 lysimeter sites from adjacent, non-irrigated control areas.

The forested site was dominated by sugar maple, _Acer saccharum_, and beech, _Fagus grandifolia_ (75 and 11% dominance, respectively, of trees equal to or greater than 10 cm in diameter [Knobloch and Bird, 1978]). This site was divided into three 1.2 ha plots with one plot receiving 5 cm/week of secondary effluent, a second receiving 10 cm/week and a third acting as a non-irrigated control. Soil and soil-water samples were collected from 10 lysimeter sites in each plot. Soil-water samples were collected with tube type porous cup vaccuum lysimeters at 15, 30, 60, 90, 120 and 150 cm depths.

Mass balance studies of nitrogen, phosphorus, and chloride were conducted for all three study sites as follows. Wastewater inputs were calculated from pump records and verified with plastic rain gauges scattered throughout each site. The wastewater was sampled using acid-washed polyethylene funnels placed 1.5 m above ground just before each irrigation period and collected immediately after each irrigation period. Precipitation inputs were monitored with 4 recording rain gauges located at intervals on the WQMF within 1.2 km of any of the individual study areas. Nutrient inputs from precipitation were calculated using literature values of N and P for Michigan (Leland et al., 1978). Evapotransporation from each site was calculated using the technique of Thornthwaite and Mather (1967). Water leaching past the root zone was sampled at the 120 or 150 cm depth as described above and average weekly lysimeter concentrations of N and P plus the amount of wastewater available for recharge according to the water budget method of Thornthwaite and Mather (1967) were used to calculate leaching outputs. When runoff occurred, it was sampled using a combination of individual grab samples and event samples taken with an ISCO sequential sampler. Discharge was calculated using recording hydrographs, V-notch weirs and stage-discharge curves developed for each runoff monitoring station. Removal in harvested vegetation was estimated from tissue analysis and from vegetation yield data determined from subsamples during harvesting or from biomass estimates derived from four randomly collected 0.25 m^2 quadrat samples of above ground biomass from each plot in the harvest plot studies.

Analyses of soil, soil-water, runoff, wastewater, and vegetation followed generally accepted wet chemistry or ion-selective electrode methods and are described in detail by Leland et al. (1978), or will be described in later detailed manuscripts so will not be included here.

RESULTS AND DISCUSSION

Forest Research

The 5 cm/week forested site did not effectively renovate wastewater since inorganic nitrogen leached past the 150 cm depths at concentrations similar to concentrations in applied wastewater. Monthly wastewater input concentrations varied from 12.5 to 20.5 mg N/l in 1976 and from 8.1 to 13.6 mg N/l in 1977. The lower 1977 values were a consequence of irrigation of additional area for other studies in 1977 resulting in greater backsiphoning from the first receiving lake. Nitrogen is rapidly lost from these lakes (see D. L. King, this symposium Volume), so this backsiphoning resulted in lower nitrogen concentrations in the applied wastewater. Weekly average concentrations of wastewater leaching past the 150 cm depth varied from 6 to 16 mg N/l in 1976 and from 7 to 14 mg N/l in 1977 with lower concentrations correlating with lower concentrations in applied wastewater. The mass balance for inorganic N reflects this lack of retention since only 15% of applied N was retained on site (Table 1).

A second forested site was deliberately water-logged by irrigation with 10 cm/week of wastewater to see if denitrification would be promoted. Concentrations in leachate have declined dramatically since the start of the study with concentrations of inorganic N at the 150 cm depth varying from 2 to 4 mg N/l in 1976 and declining to levels consistently less than one mg N/l by July, 1977. Cl/N ratios indicated that wastewater was reaching this depth. Ratios of Cl/N were similar to wastewater at the 150 cm depth on the 5 cm/week plot but increased significantly on the 10 cm/week site. The Cl/N ratio of input wastewater varied between 5 and 10, the ratio for the 150 cm depth leachate varied between 6 and 14 for the 5 cm/week wastewater application site indicating only slight removal of N but increased to 240 for the 10 cm/week application area by the end of the 1977 irrigation season. Chloride values at depth approached calculated values (input corrected for rain dilution and evapotranspirational concentration) by October 1, 1976, and remained at that level for all of the 1977 season. Thus, wastewater was infiltrating to depth, and dilution by the native soil-water pool had become negligible by October, 1976. The lack of infiltration and percolation of inorganic N to the 150 cm depth as indicated by the high Cl/N ratio and low inorganic N concentrations would suggest significant immobilization in the soils, increased uptake by the vegetation, or increased denitrification. Increased denitrification seems

Table 1: Mass Balance for Inorganic N for the October 1, 1976 to September 30, 1977 Water Year.

Treatment	Irrigation Rate (cm/week)	INPUTS (kg/ha) Precipitation	Wastewater Irrigation	Total	OUTPUTS (kg/ha) Vegetation Removal	Runoff	Recharge	Retention
Forest	0	22.25	0	22.25	0	0.04	0.13	22.08
	5	22.25	167.43	189.68	0	0	161.84	27.84
	10	22.25	302.37	324.62	0	71.80	22.89	229.93
Oldfield - No Harvest	0	22.25	0	22.25	0	0	0.25	22.20
	5	22.25	143.02	165.27	0	0	28.84	136.43
	10	22.25	263.12	285.57	0	0	119.20	166.37
Oldfield - One Harvest	0	22.25	0	22.25	37	0	0.14	-14.89
	5	22.25	143.02	165.27	128	0	30.19	7.08
	10	22.25	263.12	285.57	147	0	75.80	62.77
Oldfield - Two Harvests	0	22.25	0	22.25	70	0	0.19	-48.09
	5	22.25	143.02	165.27	156	0	10.17	-0.90
	10	22.25	263.12	285.57	237	0	87.10	-38.53

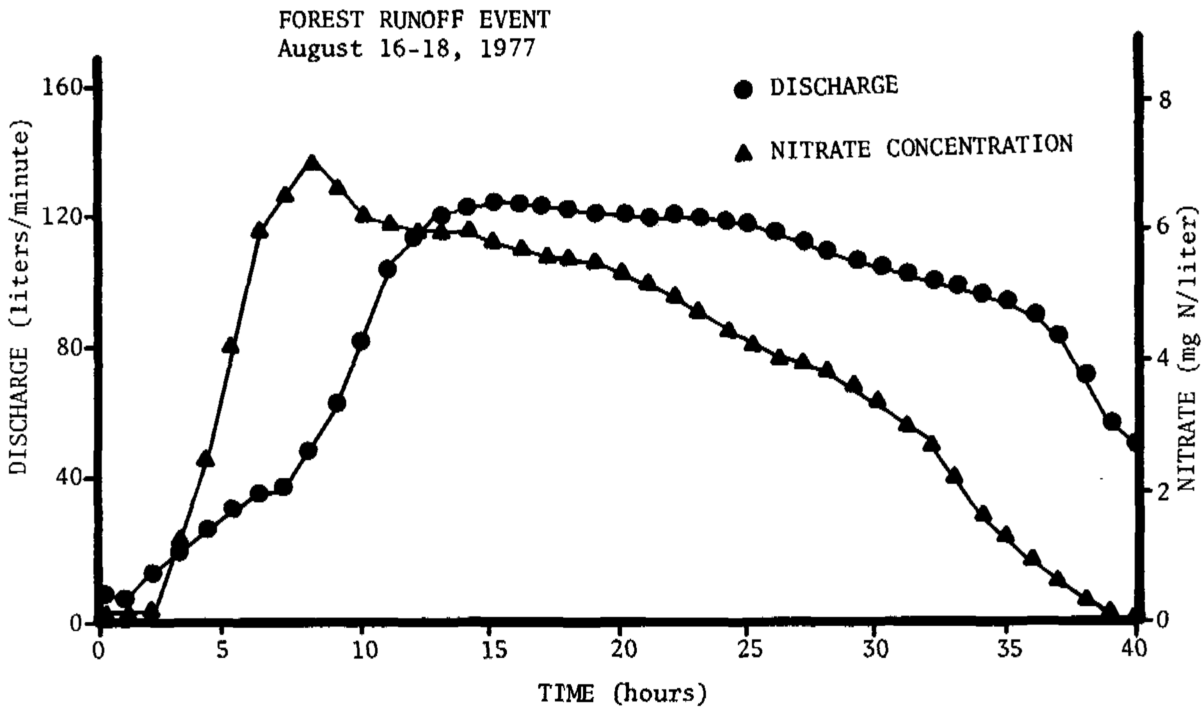

Figure 1: Nitrate Concentration Versus Discharge For a Wastewater Initiated Runoff Event From the 10 cm/week Wastewater Irrigation Site.

the most likely explanation but the others cannot be ruled out.

Even though leachate concentrations of inorganic N was very low for the 10 cm/week wastewater application site, problems with high runoff concentrations still indicated that this site was only marginal at best in terms of nitrogen removal. Peak concentratins of inorganic N during runoff events were often greater than 60% of input inorganic N concentrations. For the depicted example of a typical mid-summer runoff event (Figure 1), input inorganic N concentrations were 10.4 mg N/1, and almost all of this input N was as NO_3-N. Nitrate -N in runoff rose rapidly following start-up of irrigation, peaked at almost 7 mg N/1 several hours before peak discharge, then decreased slowly to very low levels about 30 hours later (Figure 1). On a mass balance basis, this runoff represented only 22% of annual loading (Table 1). Nevertheless, loading with wastewater with higher NO_3-N concentration of 20 mg/1 or more as is typical of effluent from many wastewater plants would probably have resulted in peaks of NO_3-N runoff that exceeded the drinking water standard of 10 mg N/1 for most runoff events. In addition, this runoff also contained peak total P concentrations that exceeded the 1.0 mg P/1 Michigan wastewater standard and resulted in 33% of applied wastewater P being exported in runoff (Table 2).

In conclusion, the forest wastewater application research indicated that little removal of N from wastewater occurred for loadings of 5 cm/week of wastewater. These results are similar to those of Hook and Kardos (1978) for a long-term study at Pennsylvania State University. Deliberate water-logging of the site did reduce leachate to acceptably low levels of NO_3, however, this reduction was accompanied by increased runoff losses of both N and P at levels that were unacceptable or only marginally acceptable. Management of a forested site in such a way that denitrification was promoted but runoff was limited might permit use of wooded sites for wastewater renovation. Such management would require a very careful balancing of wastewater input and infiltration and would probably require more attention than most wastewater land application operators would want to give it. Thus, relatively mature forests may not be very good candidates for land application of wastewaters that contain more than 10 mg N/1 of inorganic N. They do provide

Table 2: Mass Balance for Total P for the October 1, 1976 to September 30, 1977 Water Year.

Treatment	Irrigation Rate (cm/week)	INPUTS (kg/ha) Precipitation	Wastewater Irrigation	Total	OUTPUTS (kg/ha) Vegetation Removal	Runoff	Recharge	Retention
Forest	0	0.17	0	0.17	0	0.1	0.002	0.16
	5	0.17	43.75	43.92	0	0	1.52	42.40
	10	0.17	86.60	86.77	0	29.00	0.64	57.13
Oldfield - No Harvest	0	0.17	0	0.17	0	0	0.01	0.16
	5	0.17	33.30	33.47	0	0	0.40	33.00
	10	0.17	65.80	65.97	0	0	0.55	65.42
Oldfield - One Harvest	0	0.17	0	0.17	9.5	0	0.01	-9.34
	5	0.17	33.30	33.47	14.4	0	0.45	18.62
	10	0.17	65.80	65.97	24.2	0	0.67	41.10
Oldfield - Two Harvests	0	0.17	0	0.17	15.6	0	0.01	-15.44
	5	0.17	33.30	33.47	19.2	0	0.19	14.08
	10	0.17	65.80	65.97	26.9	0	0.51	38.56

excellent removal of P from wastewater. Young rapidly growing forests do appear to be effective at removal of both N and P from wastewater (Breuer et al., 1978; Brockway et al., 1978; Sopper and Kerr, 1978; Urie et al., 1978; and others). Such young forests also represent areas that can be managed for high production of fibers and would seem to be better places for renovation and recycling of wastewater than are more mature forests and woodlots.

Oldfield Research

The oldfield treatment sites were effective at removal of both N and P on an annual basis no matter what harvest regime was used (Tables 1 and 2). However, the harvest of vegetation did remove almost all of the inorganic N added by wastewater irrigation with the two harvest treatment being most effective (Table 1). Furthermore, either of the one or two harvest treatments reduced annual inorganic N leaching by about 35% (Table 1). This reduced annual N leaching was primarily the result of regrowth on the harvested plots. On the unharvested plots, growth slowed and biomass started to decrease in mid-August. At about that same time, leachate NO_3-N concentrations at the 120 cm depth on the unharvested plots began to increase steadily from levels of 1 to 3 mg N/l to concentrations that approached the 10 mg N/l level by late October. Prior to mid-August, there was little difference in N and P removal regardless of treatment. Thus, the effect of harvest was to prolong active growth and uptake of N and P and increase the effective renovation period from mid-August through late October. The harvests also removed significant portions of the added N and P as recyclable biomass (Tables 1 and 2). The June only harvest removed 77% of the N and 43% of P added in the 5 cm/week wastewater application while the June and September harvests removed 94% of added N and 73% of added P. For the 10 cm/week application rate, the June only harvest removed 52% of added N and 29% of added P while the June and September harvest removed 83% of N and 41% of P. Harvesting would, therefore, markedly prolong the sorption capacity of the site for P, reduce dependence on denitrification and immobilization mechanisms for removal of N, and increase the expected longevity of the site as a useful area for renovating and recycling wastewaters. The two harvest strategy would be more effective than only one harvest but either strategy would produce high quality leachate with respect to nitrogen and phosphorus.

Winter Spray Study

Winter irrigation of wastewater is feasible in Michigan (Leland et al., 1978). Results of these studies on the WQMF indicated that the ice build-up on the site insulated the soil and prevented significant frost penetration. As a consequence, infiltration continued throughout the winter. In fact, for the three winters of wastewater application, infiltration of wastewater was 70 to 80% of input with the remainder running off in stream water. Since there are no active uptake mechanisms for NO_3-N in the winter, NO_3-N in leachate did increase throughout the winter period and would exceed the 10 mg N/l through most of the winter if high NO_3-N wastewater were used. Also, phosphorus concentrations during the early spring snow and ice melt period exceeded Michigan's 1.0 mg P/l standard. However, on a mass balance basis, P reduction was excellent with 77% of P added during the period of December 1, 1976 to March 16, 1977 (post-runoff) being retained on site, 13% leaching to groundwater, and only 10% running off the site. Runoff losses could probably be controlled by minimal amounts of diking to prevent runoff during early snowmelt periods. Thus, winter irrigation would be an excellent way to irrigate low NO_3-N wastewater from the lake systems, thereby decreasing storage requirements and reducing construction costs for combined lagoon-land treatment systems.

CONCLUSIONS

Research on non-agronomic areas of the WQMF has demonstrated that (1) oldfields can be used effectively for renovation of up to 10 cm/week of wastewater throughout the growing season; (2) harvesting of oldfields removes most added N and a significant amount of P in plant biomass that can potentially be recycled as green manure, composted, or fed to animals; (3) harvesting of oldfields reduces the annual amount of N leaching to groundwater by about 35% by promoting active plant uptake during the mid-August through October period when growth on unharvested plots is minimal; (4) winter spray irrigation is feasible in Michigan but only if low NO_3-N water from wastewater lagoons is used and if measures such as diking are taken to control runoff from the site during spring snow and ice melt; (5) late successional forests are not effective at wastewater renovation since most added inorganic N leaches through to the groundwater; (6) late successional forests can be water-logged to promote denitrification but runoff from such a hydraulically overloaded site exports substantial amounts of both N and P with peak concentrations of both exceeding the 10 mg N/l and 1.0 mg P/l drinking or wastewater standards, respectively; (7) all areas are effective at P removal if application rates are kept low enough that significant runoff does not occur (between 5 and 10 cm/week for the WQMF).

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AGRICULTURAL AND FOREST MANAGEMENT

IRRIGATION OF FOREST PLANTATIONS WITH SEWAGE LAGOON EFFLUENTS

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ABSTRACT.--Wastewater from facultative sewage lagoons serving small Michigan communities has been used to irrigate both established plantations of *Pinus resinosa* and new plantations of hybrid poplars and Christmas trees. The established plantation has been irrigated six growing seasons at three rates on a one-day per week schedule. The hybrid poplars were irrigated five growing seasons at two rates on a weekly schedule in one location, and at two rates on an irregular schedule for four seasons at a second location. Christmas trees have been irrrigated four seasons on a single irregular schedule. Weed and rodent control has been a serious problem. Most tree species tested showed significant increases in height growth under effluent irrigation. These increases ranged from 28 percent to 181 percent above controls. Soil water at 120 cm depth was maintained at nitrate levels which meet potable water requirements (<10 ppm NO_3-N). Nitrogen and phosphorus loading levels were lower than those reported for most other land recycling research, primarily due to the dilute effluent produced by the 2-cell lagoon systems. "Raverdeaux" poplar was the only species tested which showed a significant increase in foliar nitrogen. A 5-year-old plantation of this species contained 80 percent of the total nitrogen added through effluent irrigation at the 70 mm/week rate. Toxicity symptoms due to excessive boron in leaf tissues were noted in *Pinus resinosa*. Ground cover species form the primary nutrient sink during the early stages of tree plantation development. Complete weed control was shown to cause excessive (>10 ppm) nitrate-N levels in soil water.

LAND TREATMENT FOR LAGOON WASTEWATERS

Stabilization lagoons are widely utilized by small community sewage districts to provide low-cost sewage treatment using minimally skilled operators. Where additional land can be acquired without prohibitive costs land irrigation using the lagoon effluent provides a method for filtering out the excess total dissolved solids which often prohibit direct discharge to surface waters. Land management practices on the irrigated lands often dictate the irrigation schedule. Forest crops, which are compatable with the high-moisture regime and capable of utilizing the excess nutrients, require only minimal cultivation and reduce interruptions for planting and harvesting operations.

Tests of lagoon effluent irrigation were begun by the U.S. Forest Service in 1972 at Middleville, Michigan, on one of the first lagoon sewage treatment systems in the area. The facility included two 4.4 ha (11 acre) ponds serving a largely domestic community of 2,500 population. The effluent was discharged primarily onto agricultural lands. Adjacent plantations of *Pinus resinosa* Ait. and open land were utilized for evaluation of forest cover as treatment sites for effluent.

Three replicate plots of 0.018 ha (0.045 acre) were irrigated by sprinklers at 0, 25, 50, and 88 mm per week one day a week from 1972 through 1976 in the 20-year-old *P. resinosa* plantation. The forest tests were conducted on a Boyer sandy loam soil (*typic hapludalf*). Groundwater levels were 18 to 20 m deep. Evaluations of the chemical characteristics of water passing through the root zone were made from soil-water samples removed from 60 and 120 cm depth using pressure-vacuum lysimeters.

On the open lands two *Populus* hybrids and six other species of trees (see Table 3) were planted in 1972 and irrigated at 0, 30, and 70 mm per week during the 1972-1976 growing seasons. One of the hybrids was also planted in 1973.

Measurements of soil-water quality at 120 cm depths were obtained from pressure-vacuum lysimeters under the hybrid *Populus* plantings during 1975 and 1976.

Two additional studies were installed at Harbor Springs, Michigan, in 1974 on lands owned by the Sewage Treatment Authority. The Harbor Springs lagoon system consists of paired 0.1-ha (0.25-acre) aerated primary lagoons and an 8-ha (20-acre) storage lagoon. The system services a residential-resort community of about 2,000 people. The land disposal area is located on Blue Lake loamy sand (*alfic haplorthod*) with the water table at 12 to 18 m depths. Forestry studies were conducted on the primary effluent disposal areas for this facility, an 8.1-ha (20-acre) area irrigated with a 360° (525 ft.) center pivot irrigator and a 2.4-ha (6-acre) area irrigated at two rates by a fixed set of rotary sprinklers. Four species and four varieties of one species of Christmas tree seedlings (see Table 3) were planted in four replicated blocks under the center pivot irrigator. Two *Populus* hybrids were planted in four blocks, two blocks under each of two rotary sprinklers, delivering effluent at different rates. Unirrigated control tests were established nearby, out of the range of the irrigators.

Irrigation rates and frequencies varied considerably, depending on equipment maintenance and the discharge needs of the facility. The irrigation season usually began in early May and extended until about November 1. Annual dosage levels are listed in Table 1.

Table 1. Mean annual irrigation rate and nutrient loading for forestry tests at Middleville and Harbor Springs

	Depth Irrigated	No. yrs.	N	P	K	B
	mm/yr		----kg/ha/yr------			
Middleville 1972-1977						
Pinus resinosa						
25 mm/wk	453	(6)	36	14	43	3.6
50 mm/wk	906	(6)	73	27	86	7.2
88 mm/wk	1,593	(6)	131	48	151	12.7
Seedling tests 1972-1976						
30 mm/wk	612	(5)	44	16	58	4.9
70 mm/wk	1,291	(5)	103	37	123	10.3
Harbor Springs 1974-1977						
Christmas trees	817	(3)	24	10	64	2.0
Hybrid poplars						
Low Rate	1,998	(3)	79	31	156	5.0
High Rate	4,400	(3)	160	62	343	11.0

Effluents from both lagoon systems were low in both total nitrogen and total phosphorus. The Middleville effluent increased to a relatively stable level of approximately 10 mg/l N and 5 mg/l P during the final 3 years of the test. The Harbor Springs effluent was below 3 mg/l N and 2 mg/l P throughout the test period. Because of the long detention time in the secondary lagoon volatilization of ammonia is expected to keep N levels low, although some increases may occur as the system ages (Meron *et al.* 1965).

QUALITY OF SOIL WATER

Phosphorus was effectively removed under all plantations by soil adsorption or plant uptake (Table 2). Excessive nitrogen moved to groundwater primarily as nitrate-N. There was a lower level of nitrification of organic and ammoniacal nitrogen in the Harbor Springs tests where the effluent was applied much more frequently and at higher intensities. At Middleville after 6 years of irrigation nearly 50 percent of the potassium and essentially all the boron was moving through the soil mantle under the two higher rates of irrigation in the *P. resinosa* plantation. These plots have been the only site where boron toxicity symptoms are evident from sewage irrigation.

Nitrate nitrogen levels at the 120 cm sampling depth have consistently remained within potable limits. Should the nitrogen discharged by the lagoons increase, the higher rates of irrigation would result in excessively high nitrate enrichment of groundwater, such as noted by Sopper and Kerr (1977) in Pennsylvania studies.

SOIL

Effluent irrigation has increased the overall fertility of both the litter-humus layer and the upper soil layer on the Middleville site. In the red pine plantation, litter-humus decomposition increased 50 percent and the pH changed from 4.7 to 6.7 under irrigation. In the 0- to 5-cm soil layer, average organic matter content increased 40 percent and pH changed from 4.8 to 7.2 with irrigation. Total Kjeldahl nitrogen, total phosphorous, cation exchange capacity and base saturation were respectively 50, 65, 65, and 70 percent higher with treatment than controls. Both higher cation exchange capacity and Kjeldahl nitrogen are directly attributable to the increase in organic matter moved into the soil by irrigation. All the phosphorus applied

Table 2. Mean concentrations of various forms of nitrogen, total phosphorus, potassium, and boron in lagoon effluent and in soil water under forest plantations at Middleville and Harbor Springs, 1977.

Study/treatment	N Tot	NO_3-N	TKN	P Tot	K	B
	mg/l					
Middleville Lagoon	11.4	3.2	8.2	2.9	9.3	0.88
P. resinosa						
Control	1.1	0.1	1.0	0.02	-	-
25 mm/wk	2.9	1.5	1.4	0.30	-	-
50 mm/wk	2.8	1.8	1.0	0.06	3.2	1.05
88 mm/wk	6.5	5.4	1.1	0.02	4.2	0.71
Populus hybrids[1]						
Control	2.0	1.1	0.9	0.10	-	-
30 mm/wk	2.5	1.9	0.6	0.05	-	-
70 mm/wk	3.6	2.8	0.8	0.14	-	-
Harbor Springs Lagoon	4.2	0.4	3.8	2.0	7.8	0.25
Christmas trees						
Control	1.0	0.6	0.4	0.03	-	-
Irrigated	2.5	0.2	2.1	0.03	-	-
Populus hybrids	1.2	0.4	0.8	0.02	-	-

[1]*Populus* hybrid data from 1976, last year of study.

through irrigation was accounted for in the upper 10 cm of soil. The fact that no real differences existed between irrigation rates indicated that the upper soil layer had essentially reached equilibrium and additional nutrients were being leached to lower soil layers.

Essentially the same results were obtained in the soils in the hybrid *Populus* plots only lesser in magnitude, partly because of lower irrigation rates and site differences. Uptake by the vegetation was probably different also. Total Kjeldahl nitrogen, total phosphorous, total exchangeable bases, and organic matter increased by 5, 10, 110, and 6 percent respectively. Soil pH changed from 6.0 to 7.2 and humus pH changed from 5.3 to 6.0.

SURVIVAL OF PLANTED TREES

The effect of irrigation on survival of trees varied depending on occurrence of natural precipitation, moisture and nutrient requirements, impact of destructive pests, and characteristics of planting stock.

Rainfall during the 1972 growing season was abundant and well distributed. Irrigation did not significantly affect survival of any selections planted that year at Middleville except the tulip poplar (Table 3), which probably has the highest moisture and nutrient requirement. In 1973, there was an extended dry period in midsummer. Irrigation more than doubled survival of 'Raverdeaux' poplar cuttings, the only selection that was planted that year.

At Harbor Springs, insect and mouse damage was so extensive on irrigated plots that less than 1 percent of the 'Raverdeaux' poplar and only 72 percent of the *Populus canescens* x *P. tremuloides* survived. Survival was better on control plots where moisture stress was the main cause of mortality. This difference between irrigated and control plots is related to the marked difference in growth of nontree vegetation. The standing crop of grass and weeds which provides cover for insects and mice was 90 percent greater on irrigated plots than on control plots.

The effect of planting stock characteristics is most evident in the Christmas tree trials. Austrian Hills Scotch pine stock was exceptionally sturdy. Its shoot/root ratio was only 1.6 compared to 6.7 for French Auvergne Scotch pine. Irrigation increased survival of the Austrian Hills stock by only 1 percent but it increased the survival of French Auvergne stock by 37 percent. The effect of differences in nutrient demand is most evident in a comparison of Austrian Hills Scotch pine and Douglas-fir. These two had nearly the same shoot length and shoot/root ratio. Douglas-fir had slightly smaller stem diameters. Irrigation increased survival of Douglas-fir, the more demanding of the two, by 13 percent compared with 1 percent for Austrian Hills Scotch pine.

GROWTH

Growth was not reduced by irrigation in any of the experimental plantings. At Middleville, average height of 'Raverdeaux' poplar, green ash, and white cedar were significantly ($P \leq 0.05$) greater on irrigated plots. The average heights of all other selections were higher on irrigated plots, but the increase over controls was not significant. There were no significant differences between the two irrigation rates. At Harbor Springs, irrigation nearly doubled the height of *Populus canescens* x *P. tremuloides* after three growing seasons and increased average height of Christmas trees by as much as 58 percent.

NITROGEN CYCLING

'Raverdeaux' poplar is the only selection tested that showed a significant increase in nitrogen concentration in foliage on irrigated plots, and foliage was the only tree component showing an increase. A fully stocked plantation of this hybrid would assimilate, in whole trees including foliage, 120 kg of N per hectare in the first 4 years and an additional 290 kg/ha in the fifth year.

Nitrogen in the 5-year-old plantation would, therefore equal 186 percent of N added in 30 mm of wastewater per week for 5 years and 80 percent of N added in 70 mm per week. The assimilated N would be about equally distributed among the three major components of the trees so harvesting trees without foliage would remove about 2/3 of assimilated N from the site and harvesting main stems only would remove about 1/3.

The pattern of N assimilation by 'Raverdeaux' poplar (Figure 1) is characteristic of tree plantations during the establishment period. In order to control N leaching during this period, some nontree vegetation must be retained as a

Table 3. Survival and height of trees on irrigated and control plots.

Selection	Age	Survival			Height		
		Control	Irrigated	Increase	Control	Irrigated	Increase
	(yrs)	(%)			(cm)		(%)
MIDDLEVILLE							
Populus deltoides x *P. nigra* ('Raverdeaux' poplar)	4	41	87	[1]46	183	492	[1]169
'Raverdeaux'	5	98	99	1	382	629	[1]65[1]
Populus canescens x *P. grandidentata*	5	65	89	24	298	398	34
Fraxinus pennsylvanica var. *lanceolata* (Green ash)	5	88	93	5	163	240	[1]47
Liriodendron tulipifera (Tulip poplar)	5	24	63	[1]39	157	246	57
Larix decidua (European larch)	5	21	26	5	128	225	76
Larix leptolepis (Japanese larch)	5	48	50	2	182	290	59
Thuja occidentalis (Northern white-cedar)	5	37	47	10	61	102	[1]67
Quercus borealis (Northern red oak)	5	53	64	11	113	123	9
HARBOR SPRINGS							
Populus deltoides x *P. nigra*[3]	3	50	<1	50	-	-	-
Populus canescens x *P. tremuloides*[3]	3	89	72	17	61	120	97
Picea glauca (White spruce)	4	80	98	[1]18	44	94	[1]114
Pseudotsuga menziesii (Douglas-fir)	3	69	82	[3]13	21	33	[3]57
Abies balsamea (Balsam fir)	4	16	60	[1]44	20	56	[1]181
Pinus sylvestris (Scotch pine)							
French auvergne	4	34	79	[1]45	45	83	[1]84
Austrian hills	4	86	89	3	63	106	[1]68
Scotch highland	4	66	85	[1]19	59	105	[1]78
Turkish[2]	4	16	14	-2	47	60	[1]28

[1]Increases significant at 95% level.

[2]Stock damaged before planting.

[3]Controls not replicated.

temporary sink. To further evaluate N assimilation by herbaceous vegetation, small plots irrigated with wastewater have been either mowed to control height growth of grass and weeds, sprayed with herbicide to eliminate grass and weeds, or left undisturbed. Figure 2 shows the $NO_3 + NO_2$-N concentration in percolate collected during the first growing season using suction lysimeters below the rooting zone. The high concentrations associated with spraying reflect mineralization and leaching of dead vegetation as well as the leaching of N applied with the wastewater.

FOLIAR BORON

At middleville, irrigation has significantly ($P \leq 0.10$) increased boron concentration in leaf tissue of 'Raverdeaux' poplar, *Populus canescens* x *P. grandidentata*, green ash, tulip poplar, white cedar, and red pine (Figure 3). Averages for Japanese larch and red oak were higher on irrigated plots than on controls, but differences were not sig-

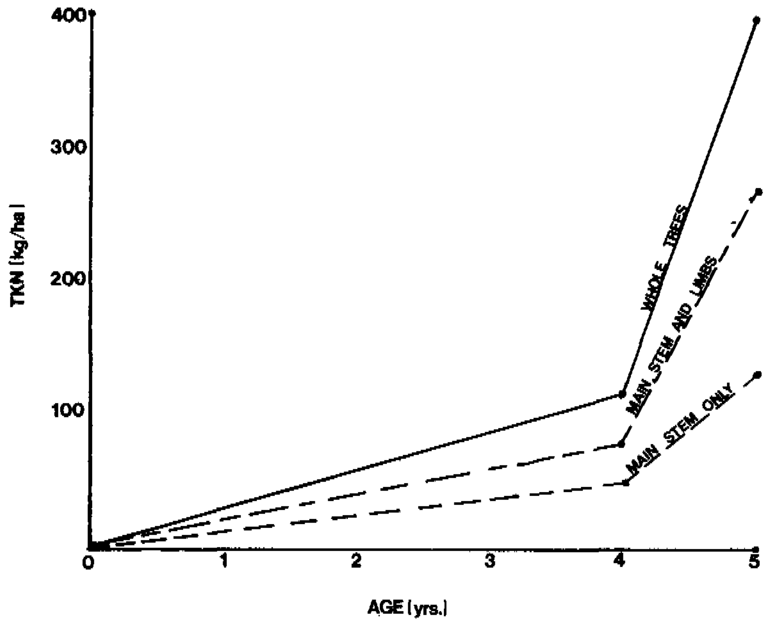

Figure 1. TKN assimilated in a plantation of 'Raverdeaux' poplar spaced 1.25 m x 1.25 m with 90 percent survival.

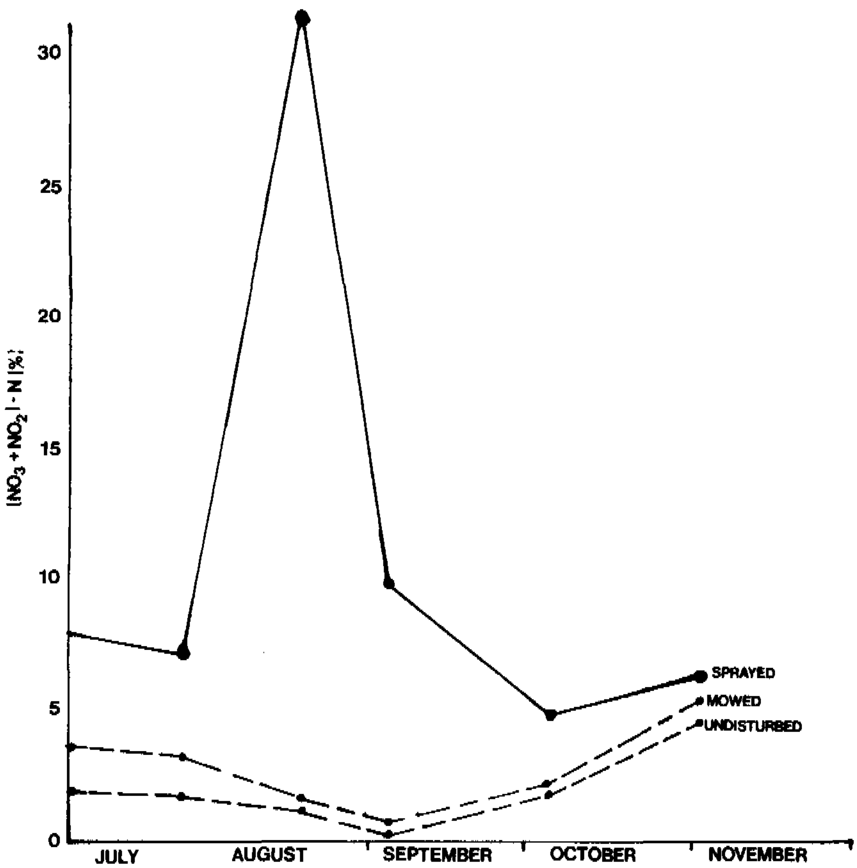

Figure 2. Nitrate in ground water under irrigated plots sprayed to eliminate grass and weeds, mowed periodically or left undisturbed.

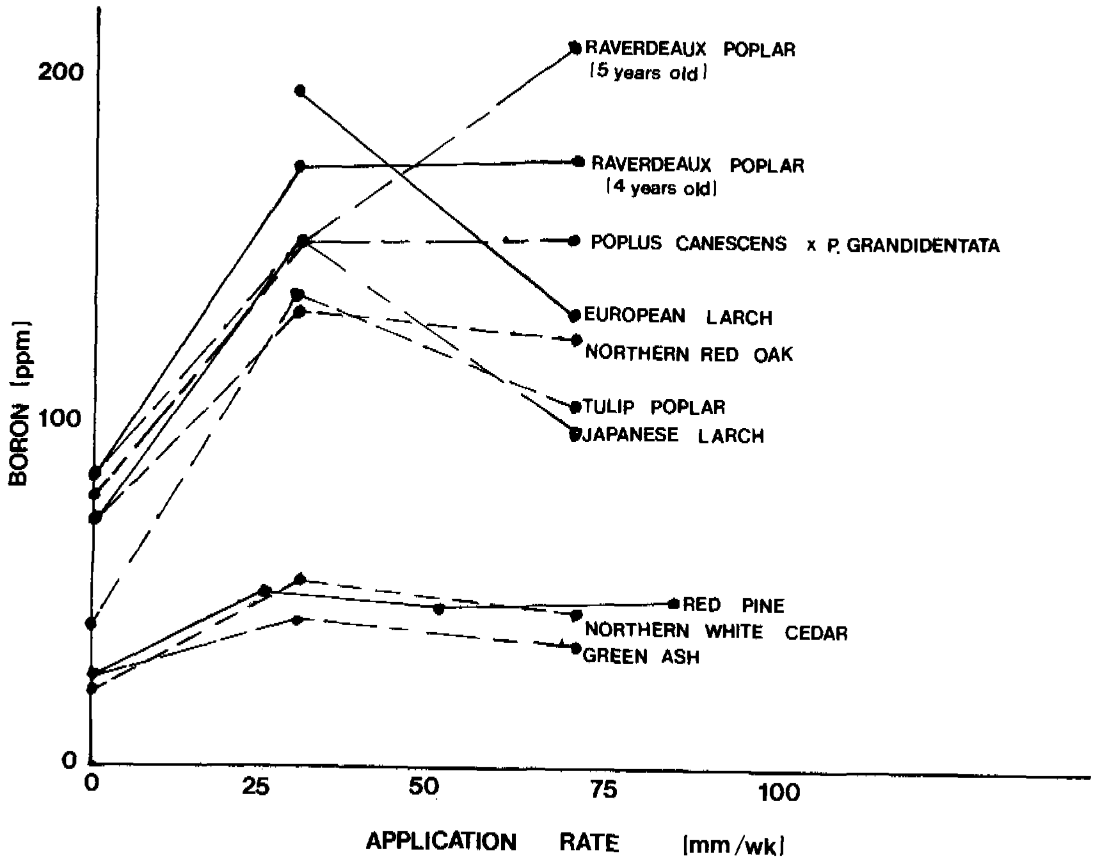

Figure 3. Foliar boron concentration after 5 years of irrigation with wastewater at Middleville, Michigan.

nificant and there were no differences with irrigation rate except in the 1972 planting of 'Raverdeaux' poplar. Visible symptoms of boron toxicity have been observed only in red pine where first-year needles of some trees have yellow tips and second-year needles have necrotic tips up to 1 cm long (Neary *et al.* 1975). The significance of these symptoms is unknown. Stem analyses have shown no decrease in growth of affected trees.

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HEALTH CONSIDERATIONS

LAND APPLICATION OF SEWAGE WASTES: POTENTIAL FOR CONTAMINATION OF FOODSTUFFS AND AGRICULTURAL SOILS BY VIRUSES, BACTERIAL PATHOGENS AND PARASITES

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Land disposal of sludge and effluents from sewage treatment plants is being utilized on an ever-expanding basis in the United States. Removal of microbial pathogens by sewage treatment plants has not been shown to be completely effective, however. Few of the sludges and effluents produced by treatment plants are monitored for the presence of pathogens, and methods are not available to detect viral hepatitis and gastrointestinal viruses. In addition, mechanical, biological, and physical factors periodically interfere with normal sewage treatment, resulting in the bypass of raw sewage for indeterminate time periods.

Laboratory and field studies have shown that viruses and bacterial pathogens may persist for weeks and even months on vegetable crops and in soils that have been irrigated and/or fertilized with sewage wastes. Some parasitic eggs/cysts have been shown to persist in soils for periods exceeding 3 years. Microbial die-off in soil and on vegetable surfaces is a rate phenomenon directly related to the original contamination levels and environmental conditions. Factors such as solar radition, temperature, humidity, and rainfall directly affect the persistence of microorganisms. If viruses, bacterial pathogens, or parasites are present on food crops contaminated with sewage wastes, these organisms may persist for a longer time than that required for growth, harvesting, and distribution of the food to the consumer.

Because most sewage treatment plant processes cannot produce sludges and effluents known to be devoid of pathogens, the Food and Drug Administration recommends that food crops that are to be consumed raw by humans should not be irrigated and/or fertilized with wastes from sewage treatment plants. To avoid possible cross-contamination, foods that enter the home or food establishment in the raw state and are later cooked should not be irrigated or fertilized with sewage wastes unless the latter are shown to be free of pathogens. In addition, because of the known persistence of parasites, sewage treated land should not be used for growing food crops that are eaten raw until at least 3 years after the last sewage application.

INTRODUCTION

The increasing visible and invisible pollution of waters in the United States and throughout the world has resulted in legislation restricting sewage discharge into streams, standing waters, and oceans. The limitations of known treatment processes in removing contaminants and pollutants have tended to shift disposal of sewage and effluents from the water to the land in an attempt to utilize the chemical, physical, and biological properties of the soil. The living filter that the land provides appears to present a solution that not only produces an acceptable water, but also provides a method of using sewage to grow crops and return income to the community.

Land disposal of sewage sludge and effluent is being used by more than 1,000 communities (1,2,3) in the United States and is being contemplated by hundreds of other municipalities as a method of secondary or tertiary sewage treatment. The fertilizer content and water volume are of agricultural value, especially in arid areas, and many waste treatment systems incorporate the use of sludge on crops in an attempt to control groundwater pollution and soil saturation by chemicals.

Although land disposal appears to be a means of solving some of our water pollution problems, the potential exists for contamination of our soils, crops, and groundwaters by pathogenic bacteria, viruses, and parasitic animals (4-14). Our ever-increasing requirement for water indicates that reuse of wastes will soon be essential for maintenance of our community and industrial needs. Every effort should therefore be made to eliminate the sources of biological and chemical pollution so that wastes can be effectively and economically used without fear of contamination (15,16,17).

FACTORS AFFECTING THE MICROBIOLOGICAL QUALITY OF SEWAGE EFFLUENTS

Efficacy of Sewage Treatment Plants

A potential public health problem exists upon sewage discharge to the environment because of the reported inadequacy of presently used effluent and sludge disinfection and treatment procedures. Removal of pathogens by sewage treatment plants has not been shown to be completely effective. If such wastes are to be used on food crops, it is essential that the sewage facility be able to remove all microbial pathogens (18).

The efficiency of virus removal from effluents of sewage treatment plants is estimated to range from 0% to 99.9%. A properly operated activated sludge plant, combined with adequate chlorine treatment, could remove 1×10^6 virus units per 100 ml from sewage. However, there is some question as to the continuous efficiency of conventional sewage treatment operations. Even under the controlled conditions of pilot plant activated sludge processes, unknown factors periodically reduce virus removal to less than 60%. The virus removal efficiency of field installations would be expected to fluctuate even more (19-24). Mammalian viruses do not propagate after entering a sewage system, but the small size and persistence of viruses in sewage waters result in passage of the organisms through many of the clarification and holding operations of the treatment plant.

Many sewage treatment plants bypass raw sewage because rainfall volume exceeds plant capacity, and also an unknown percentage are operating beyond their treatment capacity because of excessive population growth in their communities. Mechanical breakdowns often occur in sewage treatment plants, and discharge of incompletely treated sewage occurs for indeterminate periods of time. An example of such an operational problem was reported by Taylor (25), who cited a study of sewage treatment plants in California during 1964 as reported by Ongert, et al. This survey indicated that 56% of the plants that were visited had mechanical problems during the year, and 33% of the treatment plants reported bypassing raw sewage for periods of 6 hours to 300 days.

Influent Quality

The microbiological quality of sludges and effluents from treatment plants depends not only on the efficiency of the treatment plant, but on the contributors to the influent as well. An increase in the incidence of disease or carrier state in the human and animal population results in an increase in the number of pathogenic microorganisms entering a sewage treatment plant. Contamination of the food supply could result in the seeding of infection into the population, with a subsequent increase in contact infection and a resulting higher input of organisms to the sewage system. If the cycle persists, an endemic, and possibly epidemic, disease environment could develop.

During the last 100 years, the incidence of communicable disease in the United States has been reduced through the use of preventive medicine and sanitation. Certain diseases have been eliminated even though they exist in other parts of the world, sometimes in epidemic proportions. In some countries, viral, bacterial, and parasitic diseases are endemic in the human population. Such populations produce highly infectious sewage, which, unless properly treated, may contribute to the maintenance of high levels of disease in the population. In some instances, raw sewage is used for the production of human and animal foods.

Elimination of the Dilution Factor

Data produced by several investigators show that when land disposal methods are used, the majority of microorganisms in the wastes will be removed from sludges and effluents and remain in the upper portion of the soil (26-28). Instead of a dilution, as would be expected when sewage wastes are discharged into flowing bodies of water, the reverse situation occurs during land disposal, and concentration of pathogenic organisms takes place in the soil. If such land is used for the growth of food crops or other types of vegetation, the potential for contamination increases with each inefficiency in the sewage treatment system. A number of studies have shown that viruses and other organisms are attached to or associated with particulate material in sewage wastes (29-35). These particulates would tend to concentrate on the surface layers of the soil. If the treatment processes are ineffective over a period of time, sufficient concentration of microorganisms may take place in the upper levels of the soil to result in a possible health hazard. If environmental conditions result in limited sunlight, increased humidity, and decreased temperature, survival of microorganisms in such soils could be prolonged significantly, and the potential for adsorption of viral, parasitic, and bacterial pathogens onto the surface of plants would be increased. Food crops such as lettuce, cabbage, and other broad leafed vegetables would expose extensive contact areas to the soil surface, and root crops would be continuously exposed to the contaminants.

CORRELATION OF DISEASE WITH SEWAGE

Epidemiological data on the association of viral disease with sewage are limited, except in the case of infectious hepatitis (36). However, parasitic disease data in one study showed that sewage plant workers had a higher incidence of infection than controls and that the variety of parasites causing infection was higher in the sewage plant group than in the control groups (1). This study also demonstrated that the incidence of parasitic disease in consumers of sewage-irrigated crops was higher than that of the controls. Viral and bacterial diseases are probably transmitted at a similar level.

OCCURRENCE AND PERSISTENCE OF PATHOGENS IN SEWAGE

Unless sterilized all treated wastewaters retain some of the original organisms, and the accumulated solids may show a higher concentration of organisms than the liquid wastes (6).

Enteric Bacteria

Wastewaters and sludges frequently contain Salmonella that may survive for long periods of time (37-42). Although some studies have shown no correlation between the number of pathogens and other enteric organisms in wastewaters, others have reported a positive relationship between fecal coliforms and Salmonella (24,43). When fecal coliforms exceeded 1,000/100 ml of water, there was a 96% chance for the occurrence of Salmonella (44). In another study, Salmonella was demonstrated in greater than 80% of irrigation waters with an average coliform density of 250,000/100 ml (45). The waters that were negative for Salmonella had an average coliform density of 34,000/100 ml. Ratios of 255,000 fecal coliforms to 4,800 enterococci to 1 Salmonella have been reported (38). Soil and water contaminated with fecal material have been reported to be health hazards when used in the production of edible crops, and fecal organisms have been detected on field-grown vegetables (38-42,43,46-51).

Several environmental factors affect survival of enteric bacteria in/on contaminated soils, fruits, and vegetables (39,45,50,52-54). Moisture seems to be the most important microbial growth regulating factor in all soils. Growth-promoting temperatures and high soil moisture content may result in growth of some bacteria (53,54). High relative humidities (80% to 95%) in combination with high temperature (27°C) resulted in the growth of Pseudomonas aeruginosa on lettuce and beans (47). Whereas high temperatures may promote growth, refrigeration temperatures (< 4°C) enhance bacterial survival. Typhoid organisms on vegetables have survived refrigeration for 10 weeks (51). The addition of fecal material such as sewage or sludge apparently has no effect on survival or reduction of enteric organisms (51,52). A soil pH of 6 to 8 favors enteric bacteria survival, and little effect has been reported at pH values ranging from 5 to 10 (51). Only very acid peat soils appear to adversely affect bacterial survival (52).

Bacterial die-off in soil and on vegetable surfaces is a rate phenomenon directly related to original contamination levels and environmental conditions. All data reviewed showed that survival of enteric bacteria in soil was usually measured in months, and it was apparent that enteric bacteria can survive longer than the growing season for vegetables (39,42,50-57). Fruits and vegetables grown in contaminated soils could thus present a health hazard if eaten raw, even after a germicidal wash (1,40,44, 56).

Tubercle Bacilli

The numbers of tubercle bacilli that may occur in sewage are not easily estimated because of the difficulty in isolating the organisms from material that contains extremely high levels of other organisms. The number of viable organisms depends on the incidence of tuberculosis (TB) in a community (58). A study of one community where the incidence of tuberculosis was known reported that all sewage wastes contained detectable TB organisms when the ratio of infected to noninfected individuals exceeded 1:600 (59). The tubercle bacillus was also found to persist in sludge drying beds for up to 15 months. Another study found that TB organisms inoculated into sludge were reduced only 85% after digestion for 35 days, and that there was little if any decline in numbers after 25 days in sludge drying beds (60). Weiser, et al. (61) were able to isolate Mycobacterium from sludge beds used in treatment of wastes from a tuberculosis hospital, and Tison, et al. (62) isolated M. tuberculosis from a sanatorium effluent that had received full treatment, including chlorination.

We were able to recover Mycobacterium bovis (BCG) from soil for 29 days following irrigation with inoculated sludge and effluent (63). At this point, interference from soil organisms masked any further presence of the BCG indicator. In addition, mycobacteria were isolated from lettuce 35 days after irrigation and from radishes after 13 days. The crops were essentially lost after these time periods because of depletion from sampling, insect damage, and disease. A rapid initial drop in numbers of mycobacteria on the crop was probably due to the washing effect of rainfall, but the persistent low levels isolated for up to 35 days indicated firm attachment by some organisms or recontamination from soil.

Viruses

Information pertaining to virus contamination and persistence in and on vegetable crops and soils is limited. In one study of crops irrigated with sewage effluent, two of 60 field samples of vegetables were found to be positive for enteroviruses, and the viruses persisted on the crops for up to 2 months (26,64). In another study, three of 146 vegetable samples were found to be positive for coxsackievirus B; five of 170 soil samples and three of 60 wastewater samples were positive for the same virus (65). Additional studies with enteroviruses demonstrated persistence on contaminated cabbage, pepper and tomato plants for 4, 12 and 18 days, respectively (27). An extensive study was performed by Bondarenko and Grigori on market fruits and vegetables that had been harvested from sewage-irrigated fields (66). The high recovery rates from carrots, beets, and potatoes indicated extensive soil contamination with coxsackie and echoviruses. Virus contamination was also noted on sorrel, cucumbers, tomatoes, apples, cherries, rhubarb, and lettuce. A total of 457 different types of fruits and vegetables were examined, and 9% were positive for enterovirus. The crop irrigation water was designated only as drainage water. In a drip irrigation study, attenuated polioviruses were reported to persist on cucumbers for 8 days, the duration of the experiment(67).

We reported in earlier publications that poliovirus 1 persisted for 36 days on lettuce and radish crops that were spray-irrigated with inoculated sewage sludge and effluents (28,68,69). Crops grown in soil previously irrigated with effluent inoculated with poliovirus were shown to be contaminated for at least 13 days. During two separate spring seasons, poliovirus could be recovered from soils for 7 to 11 days after irrigation with sludge that had been inoculated with virus. During cooler periods of the year, poliovirus was shown to survive in soil for at least 60 days after final sludge irrigation. When similar soil was flooded with effluent, viruses persisted at least 89 days. These data indicate that when sludges or effluents are disposed on the land during cooler periods of the year, viruses may persist for months. In addition, two research groups have shown that virus uptake through the root system of plants is possible, indicating that both internal and external portions of the plant may contain virus contaminants (70-73).

A number of factors such as solar radiation, temperature, humidity and rainfall affect virus persistence on vegetation and in soil. Studies by other investigators demonstrated that viruses may pass through the surface layers of some sandy soils and be protected from solar radiation and high surface temperatures (74).

At present, no simple laboratory methods are available for detecting hepatitis A or the gastrointestinal viruses. These organisms are produced in high numbers in fecal discharges of individuals with clinical and subclinical disease. It is possible to monitor for other viruses and bacterial pathogens in sewage sludges and effluents, but such studies are expensive and not commonly used.

The enterovirus input into a sewage treatment plant has been estimated to range from 5 to 21,000 units per 100 ml (75), with an average of 200 to 7,000 units per 100 ml in the United States. Sewage from residential areas usually has a higher viral concentration than sewage from industrial plants or combined sanitary-storm sewer installations. In addition to enteroviruses, 26% to 57% of the sewage samples examined contained reoviruses, and 25% contained adenoviruses (76).

Parasites

Little attention has been given to the presence of parasites in sewage and their potential for contaminating food crops. This deficiency is probably related to the belief that only a low incidence of parasitic infection occurs in the United States. Such may not be the case, however, because of positive infections of large numbers of Vietnam War veterans and immigrants, food imports from countries with a high parasitic incidence, and extensive travel abroad of U. S. citizens.

Data from a limited number of studies (Table 1) indicate that parasites are probably present in all sewage wastes. The number and type of parasites in the sewage depend on the human and animal contributors to the system. It has been estimated that an individual infected with _Entamoeba histolytica_ may produce up to 14,000,000 cysts per day in his feces (77). A good example of one parasite survival form that is being consistently found in sewage is the _Ascaris_ egg. One female _Ascaris_ may pass up to 250,000 eggs daily in the feces of an infected human (78). These eggs have been detected in both sludges and effluents of municipal sewage treatment plants (79). The persistence of such eggs is shown in Table 2. Under optimum conditions, _Ascaris_ eggs have been shown to survive in soil for more than 6 years.

Many virus and bacterial infections resulting from improper disposal of sewage wastes can be controlled by immunization and chemotherapy, but parasitic diseases are difficult to prevent and cure. Many such infections result in long-term, debilitating disease. In Colombia, South America, 20% to 60% of the population is continuously infected with _E. histolytica_ (80). In many parts of the world, especially emerging third world countries, parasitic diseases are endemic and result in repeated reinfection, adverse public health problems, and economic depression.

CURRENT REGULATION OF SEWAGE DISPOSAL ON LAND

Information compiled by the National Commission on Water Quality indicates that 22 of the 54 States and Territories have formal regulations pertaining to land application of wastewaters (91). Thirty-eight states require a minimum of secondary sewage treatment. Land application of effluents is prohibited in the District of Columbia, discouraged in Rhode Island, and not practiced in Iowa and Nebraska. Eleven states appear to have no restrictions on land application of wastewater. Five states have regulations pertaining to the use of sewage wastes on food crops. Ohio excludes waste utilization on root crops and leafy vegetables. Oregon and South Carolina prohibit use on crops consumed by man with additional restrictions on grazing lands and feed for dairy cattle. Texas and Virginia exclude use on food crops that are consumed raw (92).

Land disposal of sewage wastes would be of no value to many communities because of lack of suitable land within an economically feasible distance from their treatment facilities. In addition, the population density of large metropolitan areas makes land disposal of effluents unrealistic, since they have volumes ap-

Table 1. Parasite Recovery from Sewage Wastes

Source	Type of Parasite	Number	Reference
Municipal anaerobically digested sludge	*Ascaris sp. *Toxocara sp. *Toxascaris leonina *Trichuris sp. ΘTaenia and Hymenolepis sp. ΨEimeria and Isospora sp.	11/120/100 g 101/340/100 g 0-16/100 g 0-23/100 g 0-33/100 g 0-372/100 g	(81)
Army Camp sludges (17)	ΘHymenolepis, *Ascaris, *Trichuris, ΨEntamoeba histolytica, *hookworm	36% positive	(82)
Municipal, primary treatment	*Ascaris and ΨEntamoeba coli	15-27/1 (in settled sewage) 6-24/1 (in effluent)	(83)
Municipal effluent	ΨEntamoeba histolytica	2.2/1	(84)
Municipal wastes	helminths	6.2/1 (primary effluent), 1.25/1 (secondary effluent)	(4,85)
Municipal wastes	helminths	700/1 (raw sludge) 200/1 (activated sludge)	(86)

Helminths
*Nematode (egg) or (larva)
ΘCestode (egg)
Ψ Protozoa (cyst)

Table 2. Reported Parasite Persistence

Source	Parasite	Time of Persistence	Reference
Lagooned sludge	nematode eggs	At least 5 years	(87)
Semi-anaerobic lagooned silt	Ascaris eggs	≥ 6 weeks	(88)
Sludge	Ascaris eggs hookworm larvae	> 81 days > 62 days	(89)
Soil	Ascaris eggs	> 6 years	(90)
Soil	Ascaris eggs	Several years	(91)

proaching 1 billion gal/day. In areas where land disposal is suitable, every attempt should be made to grow crops other than those in the food chain to minimize the potential public health problem of disease transmission through the food supply (93-104).

CONCLUSIONS AND RECOMMENDATIONS

Viral, bacterial, and parasitic pathogens found in sewage will survive on crops and in soils for periods ranging from several days to months or even years Because some of these organisms can persist for periods longer than a single growing season and longer than that required for distribution to the consumer (105,106), land application of sewage wastes may pose a potential health hazard. This hazard is compounded by inadequate sewage treatment, lack of microbiological monitoring of effluent and sludge by sewage treatment plant personnel, and the limited regulations or policies governing application of sewage wastes to land.

Because most sewage treatment plant processes do not produce sludges and effluents known to be devoid of pathogens and because of the limitations in monitoring systems, the Food and Drug Administration recommends that food crops that are to be consumed raw by humans should not be irrigated and/or fertilized with wastes from sewage treatment plants. Because of the known persistence of parasites, sludge treated land should not be used for growing food crops to be eaten raw by humans until at least three years after sludge application has ceased. In addition, sewage wastes, unless shown to be free of pathogens, should not be used to irrigate and/or fertilize food crops that enter the home or food establishment in the raw state, even when such foods are to be cooked, because of the potential for cross contamination. Proposed Environmental Protection Agency rules recommend that wastes applied directly to the land be stabilized. In addition, crops normally eaten raw may not be grown for at least one year following the application of such wastes. A longer delay may be necessary if there are positive indications of viable *Ascaris* ova or other pathogens (107).

Municipalities contemplating sewage disposal on land should investigate the use of crops other than those in the food chain until treatment and monitoring methods are in use that would preclude the seeding of crops with pathogens.

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HEALTH CONSIDERATIONS

ADSORPTION OF SELECTED ENTEROVIRUSES TO SOILS

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Removal of viruses from effluents applied to land is believed to be facilitated by adsorption of virus particles to soil. Thus, factors influencing virus adsorption will determine the efficiency of virus removal as well as the long-term behavior of viruses in soil. Of the 27 different enterovirus types studied, only echovirus types 1, 12, and 29 adsorbed significantly less to a sandy loam soil than did other reference viruses. No difference in adsorption was observed between the reference strains and recent isolates of poliovirus types 1, 2 and 3, but a great deal of intratypic difference in adsorption was found among echovirus 1 and coxsackievirus B4 isolates. Thus, adsorption of 7 different strains of echovirus type 1 to a loamy sand soil ranged between 0 and 99%. These results indicate that adsorption of virus to a given soil is highly strain-dependent. A selected number of viruses were studied for their adsorptive behavior with nine different soil types. All viruses studied adsorbed well to soils having a saturated pH below 5.0.

INTRODUCTION

Since Public Law 92-500 makes it mandatory to stop the discharge of treated or untreated sewage into large bodies of water by 1985, land application of wastewater has emerged as a viable alternative for renovation of sewage water. However, since waste treatment systems are incapable of removing all the pollutants present in the sewage, it is necessary to study the fate of such pollutants after application on land.

Little is known about the fate of human viruses applied to soil. It is obvious that if viruses are not retained by soil, they may migrate vertically, resulting in groundwater contamination. Most virus removal by soil is believed to be due to the process of virus adsorption to soil, in contrast to bacteria which are removed in the soil surface by filtration, sedimentation, and adsorption (Gerba et al., 1975).

The few studies that have been done on viral adsorption to soil have used poliovirus and certain bacteriophages as the model viruses for predicting the behavior of all enteric viruses (Burge and Enkiri, 1978; Drewry and Eliassen, 1968; Gerba et al., 1975; Lefler and Kott, 1974; Young and Burbank, 1973). Use of coliphages as an indicator for the presence of human enteroviruses in water has been advocated because of the ease of handling phages in comparison to working with animal viruses (Gerba et al., 1975; Lefler and Kott, 1974), but it has been found that coliphages do not behave in the same manner as poliovirus in their adsorption to soil (Young and Burbank, 1973). Studies concerning enteric virus adsorption to soil have only been conducted with poliovirus (Duboise et al., 1976; Gerba et al., 1975; Lance et al., 1976), whose behavior may not reflect that of other enteroviruses. The present study was undertaken to compare the adsorptive behavior of different types and strains of enteroviruses to different types of soils.

Table 1. Major Characteristics of Soils*

Sample code	Soil series	Family	Surface area (m^2/g)	Per-cent clay	Per-cent silt	Per-cent sand	Cation exchange capacity (meq/l)	Percent organic matter	Saturated pH
B	Vernon	Fine, mixed, thermic	84	39	13	48	32	0.3	4.5
C	Clarita	Fine, montmorillonitic, thermic	203	54	20	26	71	4.2	7.1
D	Windthorst	Fine, mixed, thermic	155	53	16	13	53	1.4	4.9
F	Chigley	Fine, mixed, thermic	52	28	13	59	23	1.4	8.0
H	--	Not classified	105	36	24	40	30	0.78	8.0
K	--	Sandy, siliceous, hyperthermic	--	--	--	--	6.5	3.64	7.1
T	Anthony	Coarse-loamy, mixed (calcareous), thermic	38	13	10	77	4.2	0.27	8.2
X	Rubicon	Sandy, mixed, frigid	18.4	4	4	92	5.6	0.4	5.5
FM	--	Loamy sand	--	3	8	89	--	--	7.8

* For additional characteristics, see Enfield et al. (1976).

MATERIALS AND METHODS

Soils

Nine soils from different sources were used in this study. The major physicochemical properties and source of each of these soils are shown in Table 1. Additional information on the properties of these soils can be found in an article by Enfield et al. (1976).

Viruses

The following enteroviruses were obtained initially as reference reagents from the Research Resources Branch, National Institute of Allergy and Infectious Diseases, National Institutes of Health, Bethesda, Md.: echovirus types 1-8, 11-13, 22, 24-29, and 31; poliovirus types 1, 2, and 3; and coxsackievirus types B1-B6. Several enteroviruses isolated from groundwater beneath a wastewater land disposal site were also studied. These included: 5 strains of echovirus type 1; 2 strains of coxsackievirus type B4; 3 strains of poliovirus type 2; and 4 strains of poliovirus type 3. In addition, one strain each of echovirus type 1 and poliovirus type 2 which had been isolated from estuarine water were used in adsorption studies. These viruses were identified by use of combination antiserum pools as described by Melnick et al. (1973). The viruses were initially concentrated from groundwater and seawater by membrane adsorption-elution methodology (Farrah et al., 1976; Gerba et al., 1978).

Virus Assay

Viruses were grown and assayed in the BGM cell line using the plaque assay technique and dilution procedures as described elsewhere (Gerba et al., 1978) and reported in PFU (plaque-forming units).

Adsorption Studies

The virus being studied was suspended in either deionized reverse-osmosis-treated tapwater (RO water), effluent from an activated sludge sewage treatment plant, solutions of $CaCl_2$ in RO water, or soil extract in RO water. The soil extract was prepared by addition of 50 ml RO water to a 50-ml volume of dry soil. The mixture was shaken for 30 min and the soil removed by centrifugation. The supernatant was then filtered through a Millipore HA membrane filter of 0.45-μm pore size. Adsorption of virus to soil was determined by addition of a 2-ml suspension of virus containing 10^6-10^7 PFU, in one of the solutions indicated, to 2 g of test soil. The mixture was then shaken on a shaker table at 200 rpm for 30 min. The soil was then removed from suspension by centrifugation for 5 min at 3000 rpm and the supernatant was assayed for virus. A control suspension of virus in the solution being tested, but without soil, was also treated in the same way. The difference in titer between the control and the sample containing soil was used to determine the quantity of virus adsorbed to the soil. Unless otherwise indicated, all experiments were done using RO water. Data on virus adsorption represent the average results of two to five experiments.

In studying the effect of soil concentration on virus adsorption, the ratio of soil to water was always kept constant, i.e., 1 g of soil to each 1 ml of RO water.

RESULTS

The effects of virus concentration, soil concentration, and time on virus adsorption were initially evaluated using FM (Flushing Meadows) soil and poliovirus type 1. The results, shown in Table 2, indicated that only 1 gram of FM soil was necessary for adsorption of more than 99% of 10^6 PFU of this virus. From these results, it was decided to use 2-gram amounts of soil in the remaining experiments.

Table 2. Effect of Soil Quantities on Adsorption of Poliovirus Type 1 to FM Soil

Amount of soil (g)*	Percent of virus adsorbed
0.5	74.0
1.0	99.3
2.0	99.9
4.0	99.2
8.0	99.7
10.0	99.9
15.0	99.9
20.0	99.9

* Equal volumes of soil and RO water were used.

Using FM soil and RO water, the rates of adsorption of echovirus 1 and poliovirus 1 were determined; the results are shown in Table 3. Most of the poliovirus 1 was found to adsorb within 1 min after contact with the soil, while adsorption of the echovirus 1 leveled off after a 4-min contact period. While more than 99% of the polio 1 adsorbed to the soil, only about 50% of the echo 1 adsorbed to the FM soil during the 1-hour contact period. No significant inactivation of the viruses in control samples without soil occurred during this period of time. From these results, it was decided to use a 30-min contact period for adsorption studies, to ensure that the greatest amount of viral adsorption had occurred.

Table 3. Effect of Time on Virus Adsorption to FM Soil

Time (min)	Percent adsorption of:	
	Polio 1 (LSc)	Echo 1
0	94	14
1	99	13
2	93	34
3	99	23
4	98	53
5	98	46
10	98	50
20	98	42
30	99	57
45	99	59
60	99	50

Table 4. Effect of Virus Concentration on Virus Adsorption to FM Soil*

PFU added	PFU recovered	% adsorption
4.5×10^5	<10	>99.99
2.6×10^6	3.8×10^3	99.90
2.8×10^6	7.5×10^3	99.70
1.3×10^9	5.6×10^5	99.99
1.5×10^{10}	4.0×10^6	99.97
1.6×10^{10}	1.5×10^7	99.91
4.5×10^{10}	6.0×10^7	99.87

* Increasing amounts of polio 1 (LSc) in 2 ml of suspension were added to 2-gram amounts of a loamy sand soil (FM).

The adsorption of varying quantities of polio 1 to FM soil was compared and the results are shown in Table 4. The results indicated that the soil was not saturated with virus even after addition of 4.5×10^{10} PFU of virus, and that the percent adsorption was similar for all the concentrations of virus studied.

The adsorption of 27 different reference enteroviruses to FM soil is shown in Table 5. The viruses used were supplied by the National Institutes of Health as standard reference material for each viral serotype. Most of the viruses adsorbed very well to the FM soil, with more than 90% of the added virus adsorbing to the soil. Exceptions were echo 1, 12, and 29, of which only 55.0, 78.0, and 14.0%, respectively, adsorbed to the soil.

Table 5. Adsorption of Different Enterovirus Types to FM Soil

Virus type	% adsorption	Virus type	% adsorption
Echovirus		Poliovirus	
1	55.0	1	99.9
2	99.4	2	98.0
3	98.8	3	99.6
4	96.0		
5	99.8	Coxsackievirus	
6	99.99	B1	99.99
7	99.9	B2	99.2
8	96.0	B3	96.0
11	99.9	B4	99.99
12	78.0	B5	99.9
13	91.0	B6	98.0
22	99.99		
24	94.0		
25	95.0		
26	99.99		
27	99.99		
29	14.0		
31	91.0		

Various strains of naturally occurring enteroviruses isolated from groundwater and polluted seawater were also compared for their ability to adsorb to FM soil; the results are shown in Table 6. No significant difference was observed between poliovirus isolates in their adsorption to FM soil, but a great deal of variability was observed among the echo 1 and coxsackie B4 isolates. The adsorption of echo 1 isolates varied from 99 to 0%. These results indicated that adsorption of virus to a given soil is very strain-dependent.

Table 6. Adsorption of Different Enterovirus Strains to FM Soil

Virus type	Strain number*	% adsorption
Polio type 2	R109	99.8
	R111	99.5
	R113	98.0
	9CH-1**	99.6
Polio type 3	R201	99.9
	R202	99.9
	R203	99.9
	R204	99.9
Echo type 1	4CH-1**	96.6
	R115	99.7
	V212	46.0
	V239	0
	V248	30.0
	V249	35.0
Coxsackie B4	V216	30.0
	V240	0

* Except as otherwise indicated, the viruses were isolated from groundwater beneath a wastewater land treatment site.

** *Viruses isolated from polluted estuarine water.*

In Table 7 is shown the adsorption of different enterovirus types, and of strains of echo 1 and coxsackie B4, to FM soil in RO water, treated sewage, soil extract, and solutions of $CaCl_2$. In general, strains of echo 1 adsorbed less in the presence of sewage and soil extract than in RO water. Little difference in adsorption was seen among the other viral groups in these solutions. No significant enhancement of virus adsorption occurred in the presence of 0.001 M $CaCl_2$. Adsorption of all of the viruses, however, appeared to be enhanced in the presence of 0.01 M $CaCl_2$. These data are consistent with previously published data on virus adsorption and ionic strength (Gerba et al., 1975; Lance et al., 1976).

The adsorption of these viruses to nine different soils was compared (Table 8). Again, a great deal of variability was found among the types and strains of virus in their adsorption to different soils. The exception was soil B, to which all of the viruses adsorbed very well. In general, all viruses adsorbed poorly to soil K. Poliovirus generally adsorbed better than most of the other viruses to all of the soils studied. Low soil pH appeared to be the only obvious factor influencing virus adsorption to the soils studied. All of the viruses exhibited the greatest amount of adsorption to soils B and D, both of which had a saturated pH below 5.0.

Table 7. Effect of Suspending Medium on Virus Adsorption to FM Soil

Virus type	Percent virus adsorption with: RO water	Treated sewage	Soil extract	0.001 M $CaCl_2$	0.01 M $CaCl_2$
Reference Strains:					
Polio 1 (LSc)	99.9	99.3	96	96	99.0
Echo 1	55	0	38	4	82
Echo 7	99.9	94	97	96	97
Echo 12	78	84	97	14	99.0
Echo 29	14	12	54	23	99.2
Coxsackie B3	99.4	93	87	72	99.0
Recent Isolates:					
Echo 1 (V212)	46	0	0	0	78
Echo 1 (V239)	0	14	0	7	80
Echo 1 (V248)	30	12	0	7	75
Echo 1 (V249)	35	6	0	34	70
Coxsackie B4 (V216)	30	27	6	15	85
Coxsackie B4 (V240)	0	17	37	0	80

Table 8. Adsorption of Enteroviruses to Different Soil Types

Soil type	Percent virus adsorption								
	Reference strains				Recent isolates				
	Polio 1 (LSc)	Echo 1	Echo 7	Coxsackie B3	Echo 1			Coxsackie B4	
					V212	V239	V248	V216	V240
B	99.9	99.7	99.0	99.6	99.7	98	99.8	99.3	98.7
C	95	21	98	92	23	10	25	2	0
D	94	90	68	83	70	65	99.9	67	92
F	99.0	11	99.0	97	19	53	33	32	26
H	99.6	12	99.5	97	38	49	25	34	24
K	42	28	20	56	44	5	0	12	56
T	82	13	22	35	1	52	33	34	35
X	56	78	8	73	79	68	82	44	73
FM	99.9	60	99.9	97	32	22	16	19	97

DISCUSSION

The limited amount of information available on virus adsorption to soils is almost entirely based on studies using poliovirus type 1 and bacteriophages (Gerba et al., 1975). Results with these viruses generally indicate excellent removals of virus in soil columns and adsorption in batch studies. Only the nature of the soil was recognized as a major factor influencing the adsorption of the virus, and it was concluded that viruses were largely removed within a few centimeters of the soil surface during infiltration of virus-laden wastewater (Drewry and Eliassen, 1968; Gerba et al., 1975; Lefler and Kott, 1974). With the development of methods for the concentration of enteroviruses from large volumes of water, it became possible to look for naturally occurring viruses in groundwater beneath sites used for the land treatment of sewage (Gilbert et al., 1976). The results of several field studies indicated that naturally occurring viruses can penetrate long distances into the groundwater near some wastewater land treatment sites (Schaub and Sorber, 1977; Wellings et al., 1975). Both field and laboratory studies have indicated that environmental factors such as rainfall could result in the elution of viruses from soil and further subsurface migration (Wellings et al., 1975; Duboise et al., 1976; Lance et al., 1976). These results illustrated the need for additional work on understanding factors influencing the subsurface travel of enteric viruses in groundwater.

The purpose of this study was to determine if poliovirus adsorption to soil truly reflected the behavior of other members of the enterovirus group, including recently isolated strains. It quickly became evident that, while poliovirus to a large extent reflects the behavior of most reference laboratory strains of enteroviruses in adsorption to soil (Table 5), it was not reflective of many strains recently isolated from sewage-polluted waters. In the initial screening of enteroviruses, the adsorption of laboratory strains to FM soil was evaluated (Table 5). Of the 27 different enterovirus types, only echo 1, echo 12 and echo 29 adsorbed significantly less than the other reference enteroviruses. In addition, the rate of echo 1 adsorption was found to be less than that of polio 1 (Table 3). No difference in adsorption was observed between the laboratory and natural isolates of poliovirus (Table 6), but a great deal of variability was observed between the natural isolates of echo 1 and coxsackie B4. Adsorption of echo 1 strains to FM soil ranged from 99 to 0%. Polioviruses are the enteroviruses most commonly isolated from sewage because of the widespread use of oral poliovirus vaccine. Thus, all of the poliovirus isolates are probably vaccine progeny strains and not true wild-virus variants. From this work it is apparent that virus adsorption to soil is highly dependent on the strain of virus. Differences in adsorption between different strains of the same virus type might result from variability in the configuration of proteins in the outer capsid of the virus, since this will influence the net charge on the virus (Gerba et al., 1975). The net charge on the virus would affect the electrostatic potential between virus and soil, and thereby could influence the degree of interaction between the two particles.

Soluble organics have been shown to compete with viruses for adsorption sites on mineral surfaces (Gerba et al., 1975), and it is not surprising that the viruses studied, in general, adsorbed less to FM soil in the presence of secondarily treated sewage (Table 7). Adsorption of all of the viruses was enhanced in the presence of 0.01 M $CaCl_2$, again indicating that the surface charge of the virus is involved in the adsorption of viruses to soil (Gerba et al., 1975).

Overall, polio 1 adsorbed best of all the viruses to the nine different soils studied. A great deal of variability was seen among the different viruses in their adsorption to soils (Table 8). Soil B was the only soil to which 98% or more of each virus studied adsorbed. This soil and soil D, which was also a good virus adsorber, were characterized by having a saturated pH below 5.0. Soil X, which had a saturated pH of 5.5, was not an overall good adsorber of virus. Thus, a low soil pH appeared to favor virus adsorption of all of the viruses studied. It has previously been reported that bacteriophage adsorption to soils correlates with cation exchange capacity and soil surface area (Burge and Enkiri, 1978), but no clear overall correlation was evident between these factors and virus adsorption. However, the differences in pH make it difficult to study other factors. The data presented in Tables 1 and 8 are currently being analyzed by statistical methods to provide additional information on the relative importance of various soil characteristics in virus adsorption.

It is now clear that not all viruses, even strains of the same type, will behave the same under identical conditions in soil systems. Preliminary studies on the removal of different enteroviruses during migration through soil columns indicate that viruses which adsorb poorly in batch studies also have a lower removal rate during soil filtration. Thus, batch studies may reflect to some extent the degree of virus migration into the subsurface environment during the land treatment of wastewater. More work will be needed to determine if batch studies can be definitely correlated with virus movement in soils.

It may not be surprising that all of the naturally isolated viruses adsorbed well to soils of low pH, since the technique used to concentrate them from field samples involved adsorption to membrane filters at low pH (Farrah et al., 1976; Gerba et al., 1978). Thus, viruses that did not have this characteristic would not have been isolated.

We are at present evaluating the adsorption of adenoviruses, rotaviruses, and coliphages to soils to determine if the other major groups of viruses found in sewage behave differently than do the enteroviruses.

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THE OCCURRENCE OF HUMAN ENTEROVIRUSES IN A LONG ISLAND GROUNDWATER AQUIFER RECHARGED WITH TERTIARY WASTEWATER EFFLUENTS

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The occurrence of human enteroviruses in a Long Island groundwater aquifer recharged with tertiary wastewater effluents.

A two-year study of the impact of human viruses on a tertiary-groundwater recharge system located on Long Island is currently nearing completion. Raw influents, chlorinated tertiary effluents and groundwater from beneath a uniquely designed recharge basin were assayed on a weekly basis for the presence of indigenous human enteroviruses and coliform bacteria. While high concentrations of viruses were routinely isolated from sewage influents, the chlorinated tertiary effluents were positive for virus in only 3 of 20 samples. In spite of the high quality effluent being recharged, viruses were detected in the groundwater aquifer on several occasions indicating their ability to percolate through the unsaturated zone. This finding was confirmed by the results of two poliovirus seeding experiments carried out at the field installation. At both high (75-100 cm/hr) and low (6 cm/hr) infiltration rates seeded polioviruses were detected at all sampling levels as well as in the groundwater aquifer, 7.62 m below the recharge basin. It would appear that low infiltration rates promote better virus removal in the type of soil naturally occurring in this region of Long Island.

1. INTRODUCTION

Increased demands for additional sources of potable water have resulted in the need to supplement groundwater reserves. Among methods proposed to augment groundwater supplies are those involving recharge with renovated domestic wastewater including: spray irrigation; overland flow; deep well injection; and basin recharge. The use of wastewater in any of the above schemes of aquifer supplementation has often met with opposition because of the potential hazard posed by the presence of the human viruses which commonly occur in sewage (Bernard, 1973; Gerba et al., 1975).

Laboratory studies have identified several factors which affect virus removal during the passage of wastewater through soil. Drewry and Eliassen (1968) indicated that adsorption rather than filtration was the probable mechanism of virus removal during sand or soil percolation. Gerba et al., (1975) reported that the adsorption process was strongly influenced by a number of factors including the pH of recharged water, the chemical composition and moisture content of the soil, and infiltration rate through the soil. The ionic strength of the adsorbing environment has also been shown to be important in the attachment of virus to soil particle (Duboise et al., 1976; Lance et al., 1976; Wellings et al., 1975). Clean, dry sand was shown to have little removal capabilities when saturated with a virus-containing solution (Berg, 1973), while previously moistened sand demonstrated an improved removal efficiency (Nestor and Costin, 1971). Drewry and Eliassen (1968) reported that soils containing high concentrations of clay and silt were extremely effective virus adsorbants. Bitton (1975) indicated that this efficiency was due primarily to their large surface area.

While adsorption mechanisms have been the subject of a number of laboratory studies, comparatively few have addressed the presence of naturally occurring viruses at operational recharge installations. Wellings et al., (1975) reported the isolation of poliovirus and coxsackievirus from groundwater beneath a cypress dome used for the recharge of secondarily-treated effluent. Schaub and Sorber (1977) also demonstrated the sporadic occurrence of enteroviruses in recharged groundwater. Gilbert et al., (1976) were unable to detect viruses in groundwater samples taken at the Flushing Meadows recharge project. Recently, Vaughn et al., (1978) demonstrated the presence of a variety of enteroviruses in groundwater aquifers adjacent to wastewater recharge basins at three separate recharge sites located on Long Island.

A major portion of the above studies were conducted with sewage effluents which received no more than secondary treatment. The present report describes the results of routine viral monitoring and field experimentation at a uniquely designed recharge installation which uses tertiary-treated effluents.

2. METHODS AND MATERIALS

A. Test Site

The site selected was located at the 12 Pines treatment facility in Medford, New York. The plant (1.2 MGD capacity) combines conventional primary and secondary treatment processes with tertiary treatment (denitrification-filtration) and chlorination. Treated

TABLE 1

Chemical and Physical Characteristics of 12 Pines Tertiary-Treated Sewage Effluent and Renovated Well Water

	Mean Values			
	Effluent		Well Water	
Turbidity (NTU)	4.58	(1.2-9.0)[a]	7.2	(2.5-15)
Conductivity (µmho)	439.50	(393-500)	459	(389-500)
pH	6.61	(6.10-7.20)	6.11	(5.2-7.1)
Total alkalinity (mg/l)	95.38	(48-115)	67.12	(31-98)
Chloride (mg/l)	54.13	(47-64)	53.5	(50-56)
Sulfate (mg/l)	35.63	(28-54)	36	(29-56)
Nitrate-nitrogen (mg/l)	3.38	(0.07-16.0)	7.54	(.09-17)
Nitrite-nitrogen (mg/l)	0.11	(0.006-0.49)	.026	(.003-.083)
Ammonia-nitrogen (mg/l)	5.34	(0.06-15)	.38	(.16-1.1)
Total Kjeldahl nitrogen (mg/l)	6.63	(0.9-17)	2.27	(.6-10)
Ortho phosphate (mg/l)	6.13	(4.8-6.8)	6.07	(4.8-7.0)
Fe^{+++} (mg/l)	<0.05	(<0.05)	0.05	(<0.05-0.1)
Mn^{++} (mg/l)	0.02	(0.01-0.03)	.078	(.02-.15)
Mg^{++} (mg/l)	4.53	(2.7-7.8)	4.26	(2.6-7.0)
Ca^{++} (mg/l)	19.0	(15-24)	21.37	(17-33)
Na^{+} (mg/l)	58.25	(54-68)	59.12	(54-68)
K^{+} (mg/l)	12.25	(11-13)	12.87	(12-14)
Total suspended solids (mg/l)	5.00	(1-14)	nt[b]	
Total organic carbon (mg/l)	15.13	(10-29)	nt	

[a] Numbers in parentheses represent the range of values.

[b] Samples not tested.

sewage is then discharged into nearby recharge basins, or a portion diverted for flooding on the test recharge basin. The physical and chemical characteristics of the tertiary effluent and the renovated wastewater may be seen in Table 1. Thirty yards northeast of the recharge basins is the study facility built and operated by the U.S. Geological Survey (Figure 1). The structure is a miniaturized version of the adjacent recharge basins, consisting of a circular test basin 6.09 m in diameter (25 m^2) whose surface is approximately 7.62 m above the static water level. A manhole has been carefully constructed through the center of the basin to a depth of 6.4 m so as not to disturb the natural soil. Within the manhole, gravity samplers (38.7 cm^2 capture area) have been constructed at depths of 0.75, 2.25 and 5.34 m which extend 0.9 m into the surrounding recharge basin. At the bottom of the manhole, a well has been sunk into the water table to allow testing of waters which have percolated through the system. The test basin has been equipped with instrumentation for measuring water level, infiltration rate, temperature and conductivity. The soil in the basin consists primarily of coarse sand and fine gravel and contains an average of 0.75% silt and clay (Table 2). The study facility operates at a normal loading rate of 180,000 l per day and combines the advantage of a sufficiently large operating surface with the ability to control variations which might be experimentally applied to the system. Sewage was continuously applied to the basin during the indicated study period.

B. Sample Collection

Samples of raw sewage (influent), tertiary effluent, and groundwater from beneath the test basin were collected at weekly intervals. Raw sewage grab samples were collected in sterile 4 l containers. Tertiary effluent (100 l) and groundwater (400 l) samples were collected in sterile 220 l tanks (Plast-i-cube, Greif Brothers Corp.). Between collections, tanks were thoroughly rinsed with tap water, sanitized with 0.12 N hydrochloric acid (thirty minutes)

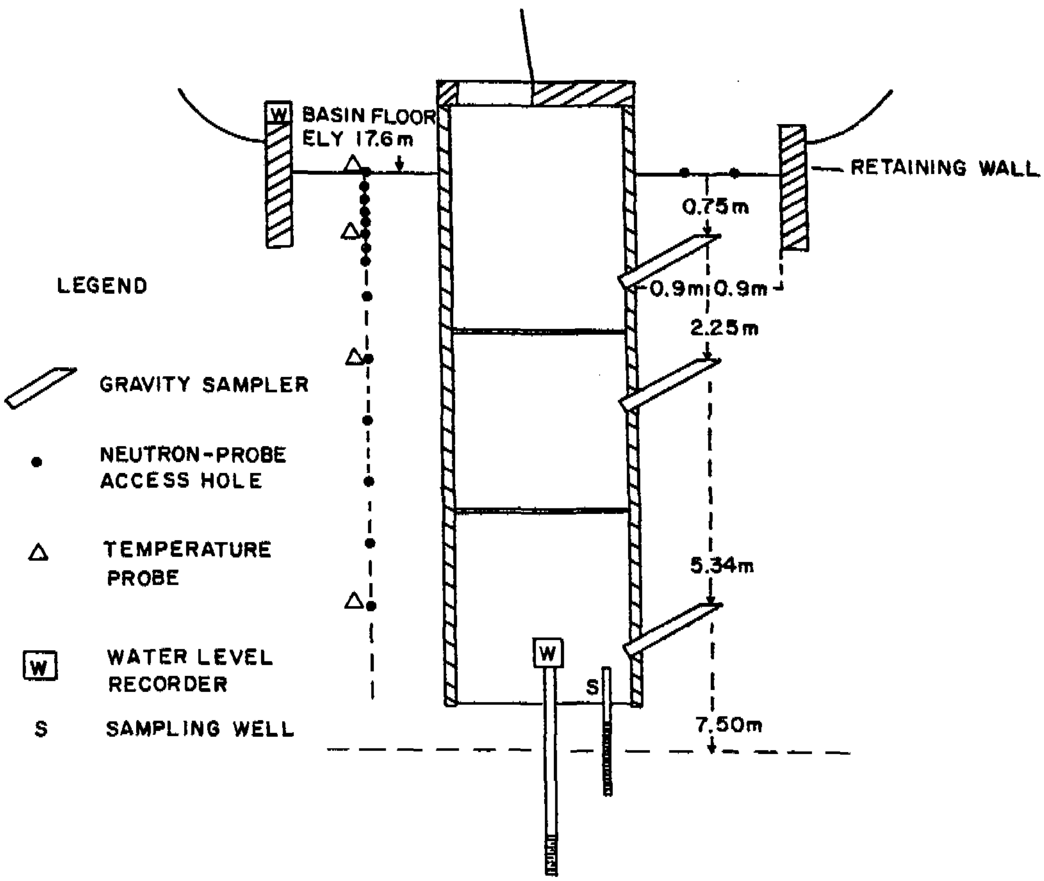

Figure 1. Schematic of the test basin facility.

TABLE 2

Percentage of Particles in Each Size Range in Soil Samples from Unsaturated Zone, Below Test Basin

(all values in percent)

Particle diameter (mm)	Depth below basin surface (m) 0.75	2.25	5.34
Silt and clay			
<0.0625	0.30	0.75	1.2
Sand			
0.0625-0.125	0.4	1.8	1.6
0.125-0.250	3.3	23.5	10.9
0.250-0.5	14.1	27.5	25.2
0.5-1	36.4	25.8	36.3
1-2	13.4	4.1	6.2
Gravel			
2-4	6.9	4.2	4.3
4-8	6.4	4.1	4.3
8-16	9.6	8.0	6.2
16-32	9.3	0.0	3.5
32-64	0.0	0.0	0.0

and rinsed once again with tap water. Immediately before collection at each site, tanks and pumping equipment were rinsed with 40 to 80 l of water to be sampled. These precautions were taken to obviate the chance of cross-contamination between samples. During the collection of tertiary effluents, residual chlorine was neutralized with sodium thiosulfate (Standard Methods, 1976).

C. Virus Concentration Procedures

Viruses in raw sewage samples were concentrated by an inorganic flocculation procedure (Farrah *et al.*, 1976). The resulting concentrates (20-50 ml) were supplemented with 10% fetal calf serum and stored at -72 C.

Viruses in large volume water samples were initially concentrated by means of an Aquella Virus Concentrator (Carborundum Corporation) using a series of prefilters to remove debris. Viruses were adsorbed to fiberglass depth cartridge filters (K27), and epoxy-fiber glass-asbestos filters (1.0 and 0.45 um Cox) at a pH of 3.5 and 0.5 mM $AlCl_3$ (Farrah *et al.*, 1976). Elution of adsorbed virus was carried out with 2 l volumes of 0.1 *M* glycine at pH 11.5. Eluates were then neutralized to pH 7.5 in an equal volume of pH 2.0 glycine. The concentration procedures routinely yielded a 4 l volume which was reconcentrated in the laboratory by means of an inorganic flocculation procedure (Farrah *et al.*, 1976) to a final volume of 50-100 ml. After the addition of 10% fetal calf serum, samples were stored at -72 C to await assay.

D. Isolation and Identification

Viral enumerations from field samples were carried out on monolayers of low-passaged Buffalo Green Monkey Kidney Cells (BGM - Microbiological Associates), which were grown on minimum essential medium with Hanks balanced salt solution supplemented with 10% fetal calf serum. Five-tenths ml sample volumes were placed on cell monolayers in 25 cm^2 flasks, and incubated for 1 hr to facilitate virus attachment. After decanting excess sample material, cells were overlain with 4 ml neutral red agar medium (Melnick and Wenner, 1964), and incubated at 36 C under 5% CO_2 for a period of 10 days. Daily readings were taken to determine the presence of viruses which appeared as plaques.

Following the incubation period, plaques were picked and enriched on monolayers of BGM cells propagated in twenty-four well cluster dishes (Costar). Isolates were identified in microtiter plates by serum neutralization techniques (Melnick et al., 1973) using enterovirus typing pools.

E. Coliform Studies

In order to correlate virus data with a recognizable biological pollution indicator, total and fecal coliform numbers were determined for all samples collected. Coliform enumerations were carried out using standard "most probable number" methods (Standard Methods, 1976).

F. Basin Seeding Experiments

On two occasions, effluents entering the test basin were seeded with poliovirus type 1 (LSc) in order to assess the virus-removing capacity of the basin under the stress of high virus concentrations. The procedures for each experiment differed and will be described separately.

a) Experiment #1

Monodispersed poliovirus (LSc) stocks were propagated on monolayers of Buffalo Green Monkey Kidney cells (BGM) in Blake bottles according to the procedures described by Jakubowski et al., (1975). Poliovirus was added to 8500 l of unchlorinated tertiary-treated wastewater entering the previously drained basin. Viruses were inoculated through an injection pipe which assured uniform distribution of virus particles. The final concentration of poliovirus in the seeded effluent was 7.0×10^4 PFU/l.

The experiment was initiated after maintaining a basin infiltration rate of 6 cm/hr for a period of 2 weeks. The seeded effluent was allowed to drain to a depth of 4-6 cm above the basin floor before resumption of normal recharge procedures with unseeded tertiary effluent. The intention of this manipulation was the production of a "band" of water containing high numbers of virus whose progress through the basin could be traced.

Samples for virus assay taken during the experiment included: tertiary sewage (40 l) and observation well water (400 l) samples taken prior to virus seeding in order to supply background information; 2 l composite samples collected at intervals from the gravity samplers located at 0.75, 2.25 and 5.34 m depths in the basin; observation well samples (400 l each) taken at intervals for a period of several days. Large volume samples (40-400 l) were processed using previously described virus concentrator methods. Virus particles in 2 l gravity samples were concentrated via an inorganic flocculation method (Farrah et al., 1976).

All samples were assayed on monolayers of BGM cells using a plaque overlay technique. Because of the low numbers expected, 9-10 ml volumes of each sample reconcentrate were analyzed for virus content.

b) Experiment #2

The initial seeding experiment was carried out at an infiltration rate of 6 cm/hr. The second experiment was designed to assess the effects an increased infiltration rate would have on virus removal in the basin. Such increased infiltration rates are commonly experienced on Long Island following basin renovation by scarification. For several weeks preceding the experiment, the basin received no effluent which promoted moderate drying. One week before the experiment, the previously clogged top 2.5 cm of basin bottom were removed and replaced with clean sand, which had been removed during initial construction of the basin. As a result of these measures, an infiltration rate of 75-100 cm/hr was realized. A 4000 l unchlorinated effluent sample was seeded with poliovirus (LSc) and allowed to drain through the basin as before. Normal recharge operations were then resumed. Because of the rapid infiltration rate, 1 l gravity samples were taken at intervals which were determined according to the movement of the seeded "front" through the basin. Within an hour of the passage of the "front" through each of the sampling levels, interval sampling was curtailed in lieu of large volume composite samples, which were collected at each level over a period of 2-3 hours. Viruses contained within these samples were concentrated by an organic flocculation method (Katzenelson et al., 1976). Groundwater samples (380 l) were also collected during the experiment, and processed through an Aquella Virus Concentrator using a modified 3% beef extract elution method (Landry et al., in press).

Tertiary effluent was applied to the basin for a period of 5 hr. All

TABLE 3

Enterovirus and Coliform Isolations
from 12 Pines Raw Sewage Influent

Sample No.	Date Collected	Virus PFU[a]/1	Coliforms/100 ml	
			Total	Fecal
A1	11/10/76	473.7	$4.6x10^{6}$	---
A8	12/2/76	452.6	$1.1x10^{7}$	$4.6x10^{6}$
A11	12/6/76	2368.4	$1.1x10^{7}$	$4.6x10^{6}$
A14	1/21/77	5368.4	$1.2x10^{7}$	$5.0x10^{6}$
A16	5/6/77	47.4	---	---
A20	5/11/77	505.3	$4.3x10^{7}$	$1.5x10^{7}$
A23	5/18/77	155.3	---	---
A24	5/24/77	113.7	$9.3x10^{7}$	$9.3x10^{7}$
A29	5/26/77	118.4	$4.3x10^{8}$	$4.3x10^{8}$
A32	6/2/77	231.6	---	---
A35	6/7/66	707.4	---	---
A36	6/20/77	20	$1.1x10^{8}$	$4.3x10^{5}$
A41	6/27/77	2682.1	$2.4x10^{8}$	$2.3x10^{7}$
A42	7/13/77	137.4	$2.3x10^{7}$	$2.3x10^{7}$
A45	7/18/77	394.7	$4.3x10^{7}$	$2.3x10^{7}$
A48	7/27/77	490.3	$7.5x10^{7}$	$4.0x10^{5}$
A53	8/2/77	454.7	$1.1x10^{9}$	$1.5x10^{7}$
A56	8/8/77	10.6	$1.5x10^{8}$	$9.3x10^{7}$
A59	10/11/77	179.5	$1.1x10^{9}$	$1.1x10^{9}$

[a]PFU-plaque forming units

TABLE 4

Enterovirus and Coliform Isolations
from 12 Pines Tertiary Sewage Effluent

Sample No.	Date Collected	Virus PFU[a]/1	Coliforms/100 ml	
			Total	Fecal
A2	11/10/76	81.0	$3.8x10^{6}$	---
A3	11/10/76	0	$2.3x10^{5}$	---
A7	11/29/76	0	<3	<3
A9	12/2/76	15.5	$2.3x10^{1}$	<3
A12	12/6/76	0	$1.5x10^{1}$	$1.5x10^{1}$
A17	5/6/77	1.1	---	---
A19	5/11/77	0	$1.7x10^{1}$	8
A22	5/18/77	0	---	---
A25	5/24/77	0	$4.6x10^{2}$	$2.3x10^{1}$
A28	5/26/77	0	$2.4x10^{3}$	<3
A31	6/2/77	0	---	---
A34	6/7/77	0	---	---
A38	6/20/77	0	$4.6x10^{4}$	$4.6x10^{4}$
A40	6/27/77	0	$2.4x10^{4}$	$4.3x10^{3}$
A44	7/13/77	0	<3	<3
A46	7/18/77	0	<3	<3
A49	7/27/77	0	4	<3
A52	8/2/77	0	<3	<3
A55	8/8/77	0	4	<3
A58	10/11/77	0	<3	<3

[a]PFU-plaque forming units

samples were analyzed for virus content as previously described.

3. RESULTS

A. Enumerations From Field Samples

The treatment system under study was operating at one-quarter to one-half capacity during the sampling period described below. To date, approximately 2727 m of water have been applied to the test basin.

As would be expected, raw influent samples routinely yielded large numbers of viruses and coliform bacteria (Table 3). A significant reduction in isolation frequency was noted in the treated effluent (Table 4), where viruses were detected on only three occasions. In spite of the infrequency of isolations from effluents, viruses were detected in the groundwater beneath the recharge basin on six occasions (Table 5) indicating the ability of viruses to penetrate the unsaturated zone above the water table. Little correlation was observed between virus and coliform occurrences in most of the samples analyzed. Variations which occurred in coliform levels throughout the sampling period cannot be explained at this time.

Viruses identified from raw influent and tertiary effluent included: Poliovirus types 1 and 2; Coxsackievirus types A16, B2, B5, and B6; and ECHO virus types 18, 21, 25, and 27. To date, only one of the groundwater isolates was positively identified as being Poliovirus type 2.

B. Basin Seeding Experiments

Experiment #1

Table 6 summarizes the data resulting from the first seeding experiment which was carried out at a low infiltration rate (6 cm/hr). Significant reduction in viral numbers resulted from

TABLE 5

Enterovirus and Coliform Isolations for 12 Pines Renovated Wastewater (Groundwater Observation Well)

Sample No.	Date Collected	Virus PFU[a]/l	Coliforms/100 ml Total	Fecal
A4	11/10/76	0.66	$9.3x10^2$	---
A5	11/18/76	0	$4.3x10^2$	$9.6x10^1$
A6	11/29/76	0	$9.3x10^2$	$4.3x10^3$
A10	12/2/76	0	$4.3x10^3$	$4.3x10^2$
A13	12/6/76	0	$1.2x10^2$	4
A15	5/6/77	0.53	---	---
A18	5/11/77	0	$4.9x10^1$	2
A21	5/18/77	0	---	---
A26	5/24/77	0	$7.5x10^1$	9
A27	5/26/77	0	$4.6x10^2$	$2.3x10^1$
A30	6/2/77	0.21	---	---
A33	6/7/77	0	---	---
A37	6/20/77	0	$7.5x10^2$	$3.9x10^2$
A39	6/27/77	0	$2.4x10^3$	$4.3x10^2$
A43	7/13/77	0	$4.3x10^1$	4
A47	7/18/77	0	$1.1x10^4$	$4.3x10^2$
A50	7/27/77	0	$2.4x10^3$	$2.3x10^1$
A51	8/2/77	0.60	$2.5x10^1$	<3
A54	8/8/77	0.66	$2.1x10^3$	9
A57	10/11/77	0	$2.8x10^3$	$3.0x10^2$
A62	10/18/77	0.14	$4.3x10^2$	$7.0x10^1$
A64	10/26/77	0	<3	<3
A67	10/31/77	0	<3	<3

[a]PFU-plaque forming units.

TABLE 6

The Recovery of Poliovirus at Various Depths During Sewage Recharge at a Low Infiltration Rate (6 cm/hr). Initial Seeded Virus Concentration = 7×10^4.

Depth Below Surface of Basin Floor (m)	Sample #	Time of Collection after Seeding (hr)	Virus PFU/l
0.75 (Level 1)	1	1.45-2.25	0
"	2	2.25-3.00	0
"	3	3.00-4.63	7.90
"	4	4.63-6.66	16.90
"	5	6.66-8.53	20.00
"	6	8.53-9.25	17.70
"	7	9.25-10.08	25.25
"	8	23.63-24.25	1.90
"	9	28.81-29.33	3.70
2.25 (Level 2)	1	2.26-2.91	5.45
"	2	2.91-3.61	0
"	3	3.61-5.26	3.18
"	4	5.26-7.66	3.60
"	5	7.66-10.08	11.60
"	6	10.08-11.16	2.20
"	7	11.16-11.91	23.30
"	8	23.68-24.33	1.59
"	9	28.78-29.41	19.50
5.34 (Level 3)	1	6.83-9.41	0
"	2	9.41-11.91	0
"	3	23.83-25.08	38.20
"	4	28.75-29.83	0
7.62 (Level 4)	1	0	0
"	2	12	0
"	3	24	0.35
"	4	30	0
"	5	48	0
"	6	54	0.08
"	7	72	0.07

passage through the unsaturated zone, the greatest removals occurring between the 2.25 and 5.34 m levels. In spite of these encouraging removal rates, some virus particles were apparently able to penetrate the entire depth of the unsaturated zone. Peaks in viral numbers were noted between 6 and 10 hours after seeding in the 0.75 m level (level 1); at 11 and 28 hours in the 2.25 m level (level 2); between 23 and 25 hours in the 5.34 m level (level 3); and at 24 hours in the observation well (level 4). The almost simultaneous appearance of virus in the first two levels within three hours of seeding suggested that the virus-laden band of wastewater did not move uniformly between these levels. There was evidence of consistent virus movement from level 1 to level 2, but only a brief period of movement from level 2 to level 3 (occurring between hour 23 and 25). Few viruses were shown to have successfully moved from level 3 to the groundwater aquifer (level 4).

Experiment #2

The second experiment indicated that decreased viral retention occurred as a result of high infiltration rate (Table 7). Large numbers of virus were detected at all sampling levels. The seeded effluent apparently moved through the first sampling level in a sharply defined band. By the time the band reached levels 2 and 3, it had been rendered less compact by diffusion within the soil column. The high virus titer encountered in the first sample from level 3 may have resulted from channeling through portions of the basin. Reduced numbers of virus were recovered from the aquifer (level 4). The reduction was likely caused by virus

TABLE 7

The Recovery of Poliovirus at Various Depths During
Sewage Recharge at a High Infiltration Rate (75-100 cm/hr).
Initial Seeded Virus Concentrations = 1.84×10^5/1.

Depth Below Surface of Basin Floor (m)	Sample #	Time of Collection after Seeding (hr)	Virus PFU/1 ($\times 10^4$)
0.75 (Level 1)	1	0.60-0.81	78.00
"	2	0.86-0.91	97.50
"	3	1.03-1.06	26.50
"	4	1.20-1.23	1.46
"	5	1.41-1.45	1.94
"	6	1.58-1.61	1.22
"	7	1.75-1.78	0.79
"	8	2.00-2.03	0.90
"	9	1.41-2.45	0.03
2.25 (Level 2)	1	1.20-1.30	2.44
"	2	1.41-1.50	8.58
"	3	1.58-1.63	6.70
"	4	1.75-1.81	5.40
"	5	2.00-2.08	1.81
"	6	2.25-2.31	1.38
"	7	2.50-2.56	0.31
"	8	2.58-3.86	0.05
5.34 (Level 3)	1	1.78-2.15	8.82
"	2	2.16-2.28	0.54
"	3	2.33-2.41	0.27
"	4	2.50-2.55	1.81
"	5	2.81-2.85	10.10
"	6	3.00-3.03	9.80
"	7	3.28-3.31	3.32
"	8	3.50-3.53	1.96
"	9	3.53-4.03	0.12
7.62 (Level 4)	1	2.50-2.66	0.11
"	2	4.66-4.83	0.001

dilution following entrance into the groundwater table. It is not known whether the replacement of a portion of the clogged basin surface had any significant effect on overall virus passage through the basin.

4. DISCUSSION

The recent development of improved virus-concentrating methods has greatly facilitated the routine isolation of human viruses from large volumes of water. Such methods, however, cannot guarantee a 100% efficiency of virus concentration. The field data presented in the preceding section must therefore be considered to be representative of the minimum numbers of virus in each sample. The inability to detect viruses within the constraints of our testing system cannot preclude the possibility of virus occurrence in very low concentrations.

Currently practiced sewage treatment methods cannot insure the removal of all human viruses, and their isolation from treated wastewater effluents has been the subject of numerous reports (Buras, 1976; Clarke et al., 1951; Metcalf et al., 1972). The presence of these organisms has been viewed as a potential health hazard to wastewater reuse operations, especially those involving groundwater recharge (Bernard, 1973; Berg, 1973; Hori et al., 1970). To date, relatively few field studies have been carried out which addressed the question of naturally occurring viruses in wastewater recharge systems (Gilbert et al., 1976; Schaub and Sorber, 1977; Wellings et al., 1975;

Vaughn et al., 1978).

The present study provides information concerning virus removal in a basin recharged with tertiary effluent. Viruses were detected in groundwater on several occasions during field sampling in spite of the fact that tertiary effluents were shown to contain few virus particles. The progress of aquifer-entrained viruses in the study area cannot be commented upon, but three previous reports have indicated the possible horizontal movement of viruses through groundwater aquifers (Schaub and Sorber, 1977; Wellings et al., 1975; Vaughn et al., 1978).

Data from field studies were verified by the results of the basin seeding experiments. In both instances, poliovirus type 1 was shown to be capable of penetrating to the aquifer (Note: Preliminary field data from this laboratory indicates that poliovirus type 1 may adsorb more readily to soil surfaces than other members of the enterovirus group [Vaughn et al., 1978] . Basin seeding data may therefore represent a conservative model). Infiltration rates used during the seeding experiments were consistent with the average "low" (i.e. resulting from partially clogged basin) and "high" (i.e. following basin scarification) rates which normally occur at Long Island recharge installations. A comparison of the two experiments indicated that lower infiltration rates resulted in a greater efficiency of virus removal during percolation through the unsaturated zone. In a similar field experiment, Shaub and Sorber (1977) isolated indigenous enteroviruses in groundwater following rapid effluent infiltration through sand basins. The basins contained similar clay and silt concentrations to those of the present study.

The results of the present study indicate that infiltration rate should be an important consideration for basin recharge operation. The findings of this study pertain to the type of soils described above, and may not be directly applicable to regions containing significantly different soil compositions.

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HEALTH CONSIDERATIONS

EVALUATION OF THE OVERLAND RUNOFF MODE OF LAND WASTEWATER APPLICATION FOR VIRUS REMOVAL

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ABSTRACT

Enteric and tracer virus removal capabilities during land application by the overland runoff mode of domestic wastewater treatment were evaluated. Raw, primarily and secondarily treated wastewaters were sprayed onto grass-covered 36-meter long soil plots. The plots contained a fine sandy loam topsoil overlying an impermeable clay subsoil with a general slope of 3 percent. Tracer bacteriophage f2 seeded into the applied wastewaters, was measured at various points along the length of the plots. Modest tracer virus removals of between 30-60 percent were observed when the soil plot effluents were sampled. In timed studies it was determined that the tracer virus advanced to the bottom of the slopes at the same rate as wastewater. Tracer virus was detected at plot effluents within 50-90 minutes after application depending somewhat on the hydraulic loading rate. Soil sampling revealed that some f2 virus was associated with the wastewater saturated topsoil but little penetration of virus into the soil profile was observed. Indigeneous enteric viruses were also sought in the overland runoff system utilizing the bentonite virus adsorption technique. The enteric virus levels were reduced by approximately 85 percent during their migration down the treatment slopes. Comparison of wastewater chemical properties and virus removal characteristics during overland runoff did not reveal any obvious correlations which could be used to predict virus treatability. Laboratory soil adsorption studies revealed that poliovirus adsorbed to soil much more readily than f2 coliphage. Poliovirus adsorption improved on test soils with increasing depth of soil sample.

INTRODUCTION

Land treatment of wastewater is considered a preferred alternative to tertiary treatment of wastewater because it recycles nutrients and water back to the land. The overland runoff mode of land treatment is being utilized or planned at a number of locations throughout the United States. Proper use of the overland runoff mode may allow significant reductions in a number of chemical and biological components of wastewater.

Treatment of the wastewater constituents is principally through adsorption, filtration, biological metabolism/conversion, and other complex interactions as the wastewater flows down the soil slope. Because the applied wastewater cannot percolate into the impermeable subsoil, most of the liquid reaches the bottom of the treatment slope where it can be discharged to a receiving stream or receive subsequent treatment.

The general consideration of factors contributing to virus removal during land wastewater treatment have been excellently reviewed and summarized by Gerba _et al_., (1) and Bitton (2).

The treatability of viruses by overland runoff of wastewater has not been examined in detail. Therefore, the intent of this study was to provide basic information on the interactions of viruses with the components of overland runoff systems. Of greatest importance was determination of the locations where virus was removed within the system, determination of the effects of wastewater pretreatment and maturity of receiving soils upon the total virus removal efficiency, and comparison of virus removal with removal of other wastewater parameters to identify any similarities which could be useful in monitoring virus removal.

The study location was most desirable because the site and treatability characteristics have been well characterized and because raw, primary and secondary treated wastewater received on the site could be examined in identical soils, application regimes, and climatological conditions.

MATERIALS AND METHODS

Study Site

During October 1976 studies were conducted on 7 overland-flow soil plots, each having dimensions of 11 m by 36 m with a ground slope of approximately 3%. The soil plots consist of a surface layer of 4-8 in. of Windthorst, a fine sandy loam underlain by impermeable, hard-packed, sandy clay. Plots 1, 2, and 3 are the original test plots and they have been in operation for 5 years. Plots 4 through 7 began operation in Jan. 1976. The older plots have a ground cover of native bermuda grass; the newer plots are covered with fescue and rye grasses.

Secondary-treated wastewater from a trickling filter process is applied to plots 1, 6, and 7 at 40, 20, and 40 cm/week, respectively. Plots 2, 4, and 5 receive primary-treated sewage at respective rates of 20, 15, and 20 cm/week. The wastewater application period for both primary and secondary plots is 6 days/week from 7 am to 5 pm. Raw settled sewage, skimmed to remove flotables, is applied to plot 3 at a rate of 20 cm/week. The application cycle for plot 3 is 45 hr/week in 5 hr periods. The wastewaters are applied to the overland runoff plots by spray with a low pressure fan-type sprinkler mounted on a boom rotating 360°.

Sprayed wastewater and plot effluents were sampled each week and analyzed for routine chemical wastewater parameters, utilizing standard USEPA methods, by R.S. Kerr Laboratory personnel (3). Total and fecal coliform counts were also performed periodically, using their standard membrane filter techniques.

Tracer Preparation and Wastewater Seeding

Large quantities of f2 bacteriophage required for the tracer studies were produced as described by Loeb and Zinder (4), and as modified by Krusé, <u>et al</u>. (5) on <u>Escherichia</u> <u>coli</u> strain K13 in a New Brunswick fermentor where a concentration of 10^{12} plaque forming units (pfu) f2/ml was attained.

Using a metering pump, a constant dosage of 10^5 pfu/ml f2 virus tracer was applied to the soil plots through the wastewater distribution systems over two separate periods to determine temporal relationships for virus migration and clearance on the plots.

Tracer virus additions were initiated on the second day of the normal weekly application cycle. Tracer f2 dosing was terminated midway through the second day of seeding, thus providing approximately 1 1/2 days of continuous virus dosing in the applied wastewaters. The above procedure was repeated on the 24th and 25th day of the study.

Wastewater Sampling

Plot influent. One liter composite samples were taken daily from the raw, primary, and secondary wastewater influent sprayed onto the cells to determine input levels of tracer f2, and indigeneous enteric viruses (which were concentrated from the sample). Light opaque collection vessels with a funnel opening were placed at ground level under the spray and remained there for 8 hr.

Plot effluent. Grab samples of effluent at the bottom of the overland flow treatment slopes were taken for evaluation of the capacity of the overland-flow system to remove tracer and indigeneous virus on each plot at designated times during the application cycle. At hour 8 of each sampling day 4 liters of each plot's effluent were also collected for enteric virus concentration and analysis.

Midpoint samples. Four ml samples were taken from a small collection weir at 18 m midpoints on each plot to provide information on the rate of migration for tracer virus removal within the system.

Soil Sampling

Shallow soil corings were taken with a stainless steel tubular coring device to provide data on the removal of tracer f2 by soil and to detect any accumulation of virus at specific soil depths. The corings of the permeable top soils down to the clay subsoil were taken at the midpoint of each plot prior to wastewater application on designated days. Corings were carefully separated to yield surface, 2 cm and 8 cm samples. Each sample was vigorously mixed 1:5 w/v with tryptose phosphate broth to elute tracer virus. The solids were allowed to settle out and the supernatant fluids were collected and frozen until assay.

Virological Analyses

Storage and shipment. Test samples of tracer f2 in 5 ml polypropylene snap lock tubes were handled as described by Schaub and Sorber (6).

Enteric virus concentrates, soil eluates, and seeded wastewater samples, all frozen, were packed in dry ice and shipped by air to USAMBRDL for assay.

F2 coliphage assay. The tracer virus stock was assayed as formerly described by Schaub and Sorber (6). Counts were recorded as pfu/ml.

Enteric virus concentration and assay. The movement and removal of indigeneous enteric viruses in the plots was evaluated by assay of concentrated wastewater samples. The 1 liter composite samples of sprayed raw, primary, and secondary wastewater were collected, as well as 4 liter plot effluent samples and were returned to the lab and concentrated for enteric viruses by the bentonite adsorption procedure. Seventy mg/l bentonite clay suspension was added, followed by $CaCl_2$ to provide a concentration of 10^{-2} M. The samples were agitated periodically for 20 min to allow virus to adsorb to the bentonite. Each sample was then filtered by negative pressure through a 142 mm diameter AP25 fiberglass prefilter in a pressure filter (Millipore Corp., Bedford, MA) which retained the bentonite and adsorbed virus. The virions were eluted from the bentonite by applying 18 ml tryptose phosphate broth (pH 7.2) to the filter pad. The eluates with the concentrated virus were then drawn into a 125 ml vacuum flask and were placed in 20 ml polypropylene snap lock tubes and frozen until assay at USAMBRDL. Sample analysis was by virus plaque assay as formerly described by Schaub and Sorber (6) on BGM cell monolayers.

Laboratory Virus Adsorption

Soil cores taken from the overland runoff site were tested for their capability to adsorb poliovirus I and f2 tracer viruses under lab conditions. Soil samples from the surface, 2 cm and 8 cm depths of overland flow plots receiving primarily and secondarily treated effluent and raw wastewater were examined. Prior to testing, each soil sample was steam sterilized. Then 2 gm of soil was mixed with 10 ml of either deionized distilled water containing 10^{-3} M $CaCl_2$ or autoclaved secondarily treated sewage effluent. The mixture was stirred vigorously for 5 minutes to break up soil aggregates. Then f2 and poliovirus I viruses prepared as in Schaub and Sorber (6) were combined and added to the mixture to provide a final concentration of 10^4 pfu/ml each. The samples were stirred intermittently for a 30 minute adsorption period. Soil adsorbed and free virions were then separated by centrifugation of 10 ml aliquots of each sample at 12,000 x G for 10 minutes. The centrifuge supernatants were plaque assayed for residual virus on BGM cell monolayers and on <u>E. coli</u> strain K-13 lawns. Additional samples were taken from the soil-virus mixture prior to centrifugation and were plaque assayed on the cells directly. The percent of virus adsorption to the soils was determined to be the difference between direct assay and centrifuge supernatant samples. Virus controls were provided by identical testing of the two virus seeded test waters without soil addition to detect virus losses due to other effects.

RESULTS

The examination of the fate of viruses in wastewater during renovation by the overland runoff mode of land treatment provided some interesting observations.

Tracer Virus Migration and Clearance on Overland Runoff Plots

A definitive picture of tracer virus migration through the plots is seen in Figure 1 which illustrates the time required for the wave of applied tracer to travel down the 36 m long overland runoff slopes. The averaged data from each type wastewater plot shows that the time required for virus to equilibrate in the plot effluents after initiation of tracer seeding was rapid, occurring within 60-90 minutes. Only small differences in virus migration to the plot effluents occurred at various hydraulic loading rates. The tracer virus seeding experiments revealed essentially plug flow characteristics although plot 3 (20 cm/wk) and plot 4 (15 cm/wk) tended toward more disperse flow.

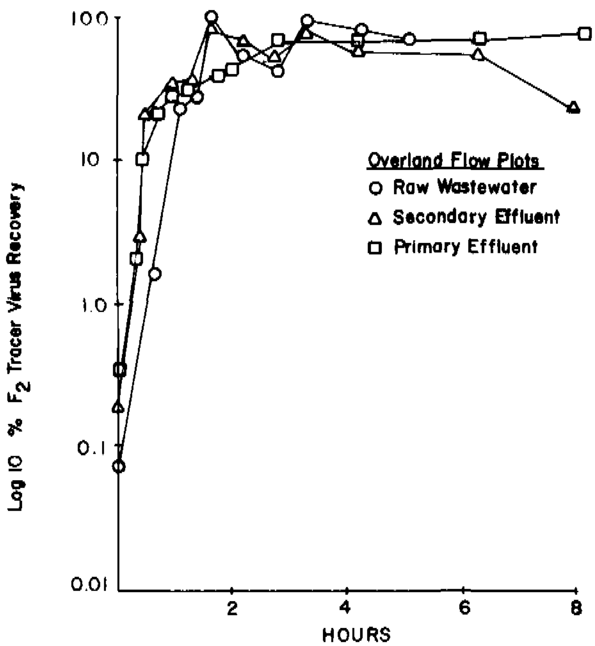

Fig. 1. Tracer f2 Virus Recovery from Overland Runoff Plot Effluents after Initiation of Seeding. The Percent of Virus Recovery from Plot Effluents was Calculated from Seeded Influent Levels.

Intensive sampling was also conducted to establish the time required for applied tracer virus to be flushed from the overland runoff plots after 1 1/2 days of virus application in the wastewater (Fig. 2). In this figure 0 hr is the time at which the tracer virus dosing was terminated. The average virus recovery in plot effluents for each type wastewater is indicated as a percent of applied virus. This figure illustrates the steady rate of clearance of f2 tracer from the plot effluents after seeding was terminated. The flushing of f2 virus from the plots was not nearly as rapid as the response to virus seeding of the plots and in excess of 4 hr was required to achieve a new background equilibration of virus in the plot effluents. The new background coliphage (a nondifferentiated combination of tracer f2 and other indigeneous coliphage) level was approximately 0.1 percent of the applied tracer levels but was 10-fold greater than the original background indigeneous coliphage concentrations. The virus flushing rates for the plots receiving the three type wastewaters were somewhat different, however, the differences were not attributable to differences in hydraulic loading rate or age of the plots. The primary wastewater treatment plots took longest to equilibrate after virus seeding was terminated although the wastewater application rate was approximately that of the raw wastewater treatment plots.

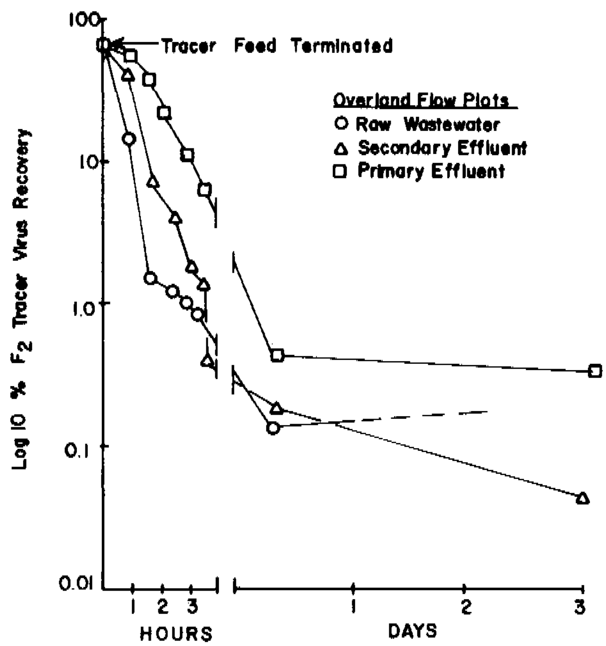

Fig. 2. Tracer f2 Virus Clearance from Plot Effluents after Termination of Wastewater Seeding. The Percent of Virus Recovered from Plot Effluents was Calculated from the Average Influent Level Attained During Seeding.

Tracer f2 Coliphage Treatability

Throughout the overland runoff study f2 coliphage samples were taken simultaneously from the migrating surface waters at 18 m plot midpoints and plot effluents. A comparison of the plots capabilities to remove the viruses can be seen in Figure 3 where the average percent f2 virus removal at both sampling points was calculated from virus application levels. The f2 virus removals generally ranged from 27 to 58 percent of input levels. The f2 coliphage was most actively removed (treated) in the upper half of the plots. Only plot 4 receiving primary treated wastewater at a low hydraulic loading rate effectively removed virus in the lower half of the plot. Treatment plots 1, 3, 6, and 7 exhibited negligible virus removal capabilities on the lower half of their slopes. Plot 3 receiving raw wastewater was poorest in overall virus removal capabilities at around 27 percent.

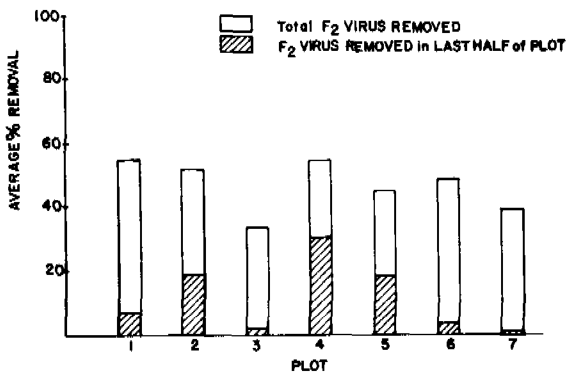

Fig. 3. Tracer f2 Virus Treatability Profile on Overland Runoff Plots.

Tracer f2 Coliphage Interaction within Soils

Soil samples were taken at plot midpoints to determine the extent that coliphage was able to infiltrate the soil and also to detect virus buildup within the soil through filtration or adsorption. Soil cores from the surface, 2 cm and 8 cm depths were examined for total coliphage on days 2, 4, and 13 after tracer seeding. The percent of virus capable of infiltrating beneath the soil surface on each sampling day is calculated in Table 1. This data taken from plots receiving primary and secondary treated wastewaters was combined and averaged. It reveals that an average of only 12.7 percent of f2 coliphage detected on the surface was able to penetrate to the 2 cm level and only 1.5 percent reached the 8 cm level. On day 2, after 1 day of tracer virus seeding, only 8.1 percent of the virus on the surface penetrated to 2 cm and only 0.94 percent was found at 8 cm. After tracer seeding was terminated, recoveries of virus in subsurface samples compared to surface concentrations was much higher. However, this reflected decreased surface virus levels and not a buildup at lower soil depths. Possible contamination of lower soil profile with virus from the surface during coring cannot be ruled out completely, but if so, these subsurface virus recoveries represent a conservative virus removal picture.

Table 1. Translocation of f2 Tracer Virus in Soils After Seeding in Wastewaters

Soil Depth	% Virus Translocation[a] Day 2	Day 4	Day 13	Avg.
2 cm	8.1	14.8	15.2	12.7
8 cm	0.9	1.7	1.7	1.4

[a] $\frac{\text{pfu of f2/gm soil in subsurface sample}}{\text{pfu of f2/gm soil in surface sample}}$ x 100.

Enteric Virus Treatability

Indigeneous enteric viruses were sampled and concentrated from the sprayed wastewaters and overland runoff plot effluents. Virus samples were concentrated to provide a final volume of 18 ml for tissue culture assay which provided a 55-fold concentration factor for the sprayed wastewater and 220-fold concentration for plot effluents. The enteric virus levels in the sprayed wastewater influents averaged 28 pfu/l, 36 pfu/l and 138 pfu/l for secondary and primary treated effluents and raw wastewater, respectively. Table 2 illustrates enteric virus removals for each plot effluent sample as a percent of virus detected in concurrent applied wastewater concentrates. In certain instances enteric virus removals could not be accurately assessed because the virus levels were below detection limits for the volumes sampled and assayed. However, all the secondary treated wastewater plot effluents revealed >85 percent removal. Surprisingly the raw wastewater plot also removed >85 percent of the applied enteric virus (although sample size was limited). Primary treated wastewater plots exhibited more variability and relatively less virus treatability.

Table 2. Percent Enteric Virus Removal by Overland Runoff[a]

Sample Day	Raw Wastewater	Primarily Treated Effluent			Secondarily Treated Effluent		
	Plot 3 (20 cm)[b]	Plot 2 (20 cm)	Plot 4 (15 cm)	Plot 5 (20 cm)	Plot 1 (40 cm)	Plot 6 (20 cm)	Plot 7 (40 cm)
1	-	86.0	>94.0	90.8	94.0	88.0	>94.0
2	-	39.4	76.8	76.8	94.0	82.3	64.7
3	-	81.3	95.3	76.8	>94.0	94.0	>94.0
4	-	72.1	95.3	72.1	88.3	94.0	94.0
8	-	>94.0	>94.0	72.1	>94.0	94.0	>94.0
11	77.0	67.3	58.1	86.0	94.0	82.3	88.3
15	-	72.1	58.1	53.4	64.7	88.3	76.0
24	91.5	21.0	53.4	62.9	82.3	94.0	88.3
25	95.1	76.8	95.3	25.4	88.3	94.0	76.0
Avg. % Removal	87.9	>67.8	>80.0	68.5	>88.2	90.1	>85.5

Average pfu/liter at plot influent: Raw wastewater - 137.8; Primarily treated wastewater - 35.8; Secondarily treated wastewater 28.3

[a] Calculated from virus in plot effluents as a % of plot influent virus levels.

[b] Centimeters of wastewater applied to plots weekly.

Virus Batch Adsorption Tests on Ada Overland Runoff Plot Soils

Representative soil cores from mature overland runoff plots receiving primarily and secondarily treated effluent and raw wastewaters were tested in the laboratory for their capability to adsorb f2 virus and poliovirus I in various water or wastewater suspension batch studies. After 20 minutes adsorption to the various soils the mixtures were split. One portion was assayed directly on appropriate host cells while the second was centrifuged to pellet soil particles and the centrifuged supernatant was then assayed for residual virus on the appropriate host cells. A comparison of virus adsorption to soils contained in secondarily treated wastewater effluent and in distilled deionized water containing 10^{-3} M $CaCl_2$ can be seen in Table 3. It is obvious that poliovirus I adsorption was greater than f2 virus regardless of the suspending medium, depth of the soil sample or quality of wastewater treated on the soil plots. It is also noted that poliovirus adsorption increased in either suspending medium with depth of the soil sample. On the other hand f2 virus data does not have any definite trend toward increased adsorption with sample depth in either suspending medium. F2 virus adsorption was not improved in any of the three soil systems. Surprisingly, poliovirus I adsorption was better when contained in the wastewater. It was determined that soil from the plot receiving secondarily treated wastewater had superior virus adsorption capabilities followed by the soils treating the primary effluent (except for the surface soils) and finally the soils which had treated raw sewage had poorest virus adsorption capabilities.

DISCUSSION

Overland runoff utilizes land treatment mechanisms somewhat differently than used by rapid and slow infiltration modes of treatment. Because of an impermeable clay subsurface layer the contact of various chemical or biological wastewater components with deeper soil layers is prohibited. Virus treatment in overland runoff is provided by the mechanisms of surface settling, straining, filtration, biological metabolism, predation by soil microflora, toxicity of the soil, photochemical reactions and thermal inactivation. Most of these processes take place at or near the soil surface as the wastewater migrates down the surface of the treatment site.

The f2 tracer virus applied to the soil plots was able to migrate to the overland runoff bottom drains (effluents) a distance of 36 m and stabilize throughout the application cycle within 50-90 minutes after tracer application began. No obvious differences in migration of

Table 3. Soil Adsorption Capability of Overland Flow Soils

Soil Depth	% Virus Not Adsorbed to Soil[a]			
	H_2O[b]		Wastewater[c]	
	f2	Polio	f2	Polio
Surface				
Plot 1[d]	126.3	1.3	160.0	0.35
Plot 2	129.4	60.7	74.1	29.3
Plot 3	100.0	6.0	166.8	17.8
2 cm				
Plot 1	65.4	0.37	105.3	0.24
Plot 2	80.0	2.8	88.1	0.92
Plot 3	77.8	3.4	121.1	2.6
8 cm				
Plot 1	129.4	0.096	95.2	0.023
Plot 2	106.0	0.54	128.2	0.35
Plot 3	107.4	3.3	76.6	1.4

[a] Calculated from virus in centrifuged supernatant as a % of mixture prior to centrifugation.
[b] Distilled deionized + 10^{-3} M $CaCl_2$.
[c] Secondarily treated.
[d] Wastewater pretreatment: Plot 1 = secondary; Plot 2 = primary; Plot 3 = raw.

the tracer virus could be matched with age (maturity) of the plots, quality of the wastewaters applied or nature of the flow conditions. On plots receiving the least hydraulic loading (plots 3 and 4) a trend toward more disperse flow characteristics were observed. The time required for tracer virus migration to the treatment plot effluents was very similar to that observed during a tracer study with Rhodamine WT dye performed at the site by EPA staff personnel (7). They observed that 45-125 minutes was required for the tracer dye migration process. In that study the plot receiving the least hydraulic loading (plot 4) also had disperse flow characteristics.

After termination of tracer virus seeding to wastewater the virus was cleared from the plots slowly and more than 4 hr of flushing were required before f2 virus levels in the effluent became equilibrated at new background levels. This indicates that some virus were retained through the effects of surface pooling, flow channelization, temporary adsorption or filtration on the soil surface. Because the stabilized coliphage levels in plot effluents remained approximately 10 times greater than levels prior to tracer introduction it is possible that there was some regrowth of indigeneous coliphage or that some slow virus releasing mechanism was operating. In fact in 18 m midpoint soil cores it was noted that the surface concentration of coliphage dropped slowly (approximately one order of magnitude) over a 13 day period. Subsurface penetration of f2 coliphage would appear limited, since only an average of 12.7 percent of surface virus concentration was observed at the 2 cm depth and only 1.4 percent of the surface concentration was demonstrated at the 8 cm depth on plots receiving primary or secondary treated effluent. It is possible that greater retention within the soil strata occurred in the upper portion of the treatment plots where it was determined that greatest coliphage f2 removals occurred. In general, well over half of the total coliphage removals occurred in the upper portion of the slopes regardless of plot age or type of wastewater applied. Only plot 4 which had the lowest hydraulic loading accomplished greater than 50 percent of the total coliphage removal on the lower portion of the treatment slope but it is uncertain if the low hydraulic loading rate was a factor.

The laboratory virus adsorption studies with f2 coliphage indicated that this virus poorly adsorbed to any of the test soils regardless of depth or suspending medium. Previous studies by Schaub and Sorber (6) and by Moore et al., (8) have revealed that f2 is a poor adsorber on either soils or organic solids at intermediate pH, therefore, the adsorption data from the current study are not surprising. It is likely that f2 coliphage adsorption on the field plots was not a major factor in their removal.

Enteric Virus Removal

Enteric virus removal by overland runoff averaged about 20 to 30 percent greater than f2 coliphage as determined on the plot effluent samples. Exact enteric virus measurements were not possible in certain instances because of limitations in virus sample volume concentration and assay procedures which were imposed. It was determined that the plots receiving secondarily treated wastewater effluent and raw wastewater provided most efficient enteric virus removal while plots receiving primary effluent provided less enteric virus removal albeit better than f2 coliphage removal. The higher enteric virus removals noted on the plots may be in part a reflection of the good adsorption of a typical enteric virus, poliovirus I, to soils observed in the laboratory adsorption studies. In the lab studies soils from the plots receiving secondarily treated wastewater and soil samples at greater depths were more effective in virus adsorption than surface soils from the other wastewater treatment plots. The differences in adsorption may be due to factors (such as soluble organic materials) in the wastewaters which tie up virus adsorption sites on the soil particles. Removal of these entities during secondary wastewater treatment prior to soil application or through removal of the entitites on the upper soil fractions upon application may explain the poliovirus adsorption results in the laboratory studies.

Comparison of Chemical, Virological and Bacteriological Treatability by Overland Runoff

The virus treatability data from this study and chemical and bacteriological indicator data gathered by USEPA personnel (7) on the Ada overland runoff site during the same test period were comared for each type wastewater application. The collective chemical wastewater parameter removals were greatest on the raw wastewater treatment plot followed by primarily and then secondarily treated effluent plots. Collectively the biological parameters (coliforms and viruses) behaved differently with secondarily treated effluent plots providing highest removal efficiency followed by primarily treated effluent plots and then the raw wastewater plot. Possibly the biological removal capabilities of overland runoff are linked to the degree of wastewater pretreatment.

None of the nonvirologic parameters appeared capable of providing enough correlation to be used to monitor virus treatability on the overland runoff system.

In conclusion, coliphage tracer studies revealed that virions generally accompanied wastewater through the treatment slopes at the same flow rate in essentially plug flow characteristics. Tracer virus clearance was somewhat slower but there was little adsorption or penetration of virus into subsurface soil layers; therefore, pooling, flow channelization, or straining may be occurring on the plots. Most of the tracer virus removal was on the upper half of the treatment plot slopes but total removals ranged between approximately 30 to 60 percent of applied virus. Indigeneous enteric virus removals were greater than tracer virus but their removals on soil plots receiving wastewaters of different qualities did not conform to coliphage removal characteristics. The indigeneous enteric group of viruses may adsorb to the various soils more readily than coliphage. Chemical wastewater parameter removal characteristics did not parallel virus removal. More studies are required to fully assess and optimize enteric virus removal by overland runoff.

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HEALTH CONSIDERATIONS

RELATIVE HEALTH FACTORS COMPARING ACTIVATED SLUDGE SYSTEMS TO LAND APPLICATION SYSTEMS

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ABSTRACT

The comparative health factors relative to activated sludge and land application systems are presented. Removal mechanisms and their effectiveness are reviewed. Factors needed to provide an evaluation method, including contact with a susceptible host, and concentration of health affecting wastewater components are then addressed. Land application systems provide equal or greater protection against various wastewater components. Both treatment systems are effective in minimizing health risks.

INTRODUCTION

Land treatment of municipal wastewaters has been practiced at various locations in the United States for over 100 years. The passage of PL 92-500 and PL 95-217, with emphasis on land treatment and recycling of resources, has resulted in a renewed interest in the subject. Further, guidelines promulgated by the Environmental Protection Agency (EPA) have required evaluation of such systems in all facilities plans.

Many public health and water resources regulatory agencies were unprepared for this level of emphasis on an unconventional approach to treatment of municipal wastewaters. Having little or no data on which to make judgments, they felt compelled to take a conservative approach to management and preapplication guidelines until a body of reliable data could be developed to support a less stringent position. A similar approach has been taken by a major portion of the environmental engineering profession.

Purpose

The purpose of this paper is to present a limited comparison of some of the health factors associated with two treatment systems. Although such a comparison must be somewhat subjective at this time, it should provide an approach or framework under which more objective comparisons can be made as data are developed.

Basis of Comparison

For purposes of this paper, conventional treatment will be limited to a standard rate activated sludge plant with discharge to surface waters. A slow rate system will be used for the example assessment as the most common land treatment option.

To reduce the dissimilarities between conventional and land treatment systems, the comparison will be made on a land treatment system with effluent recovery and discharge to a water body. Consideration of aerosols produced from either sprinkling of effluent or aeration of wastewater is not included.

REMOVAL MECHANISMS

Conventional Treatment

In conventional treatment, reductions in infectious agents occur in primary treatment (sedimentation), secondary treatment (aeration and sedimentation), and chlorine disinfection, and as a result of dilution and die-off following surface water discharge. The reductions in number of the infectious agents are presented in Table 1. Although the overall removals vary from negligible values to greater than 99% removals, highly effective mechanisms are required to produce the 4 to 8 log reductions in numbers that are needed to reduce pathogen counts to low numbers. Conventional wastewater treatment has relied heavily on chlorine disinfection to accomplish this die-away of infectious agents.

Table 1. Enteric Microorganism Reduction By Conventional Treatment [1-4]

Microorganism	Primary treatment removal, %	Secondary treatment removal, %
Total coliforms	<10	90-99
Fecal coliforms	35	90-99
Shigella sp.	15	91-99
Salmonella sp.	15	96-99
Escherichia coli	15	90-99
Virus	<10	76-99
Entamoeba histolytica	10-50	10

Conventional wastewater treatment generally discharges the treated effluents into surface bodies of water. Since these waters occasionally serve as recreation areas and sources of potable water, the effect of the receiving waters on the infectious agents is of interest. The survival of enteric microorganisms is less favorable in environments other than the human body. The die-off mechanisms in surface waters combine with dilution to promote reduction in populations (Table 2).

Conventional activated sludge treatment does not remove nitrogen, but modifications of the process can oxidize organic and ammonium forms to lessen oxygen demand of the treated effluents. Nitrate concentrations meeting drinking water standards are usually achieved by dilution within the receiving surface body of water.

Table 2. Survival of Infectious Agents In Surface Waters [5]

Infectious agent	Time for 99.9% removal, d	Removal after 2 d, %
E. coli	5-7	86-94
S. fecalis	8-18	54-83
Enterobacter aerogenes	8-18	54-83
Echo 12 virus	5-12	68-94
Polio I virus	13-20	50-65

The concentrations of trace metals occurring in strictly municipal wastewaters are usually low, since industrial inputs are generally the major source. As shown in Table 3, the concentrations occurring in untreated municipal wastewaters are, at times, less than the EPA drinking water standard. The removals from solution by primary and secondary conventional treatment are also presented in Table 3, as well as the removal by slow rate land treatment. However, the concern about trace metals does not cease with removal from solution, since they are concentrated in sludges as a result of conventional treatment.

Activated sludge treatment with surface water discharge provides the comparison to land treatment. The activated sludge treatment sequence consists of unit processes for grit removal, primary sedimentation, aeration, secondary sedimentation, chlorination (as an option), and surface water discharge. In cases where a flow is required, a 11.4 m^3/s flow is used for example purposes.

Land Treatment

Land application systems provide reductions in infectious agents by some, or all of the following methods: primary treatment, aerated lagoon secondary treatment, storage, application method, retention on soil surface, retention within soil profile, die-off in soil environment by predation, and dilution in groundwater or surface waters.

Table 3. Summary of Trace Metal Information, Concentrations, and Removals [6-8]

Component	EPA drinking water standard, mg/L	Raw municipal wastewater concentration, mg/L	Primary treatment removal, %	Secondary treatment removal, %	Removal by slow rate land treatment
Cadmium	0.01	0.004-0.14	30	60	$\lessgtr$drinking water standards
Chromium	0.05	0.02-0.7	40	40-80	<drinking water standards
Copper	1.0	0.02-3.4	40	60-70	<drinking water standards
Lead	0.05	0.05-1.3	50	50-60	$\lessgtr$drinking water standards
Manganese	0.05	0.11-0.14	30	20	--
Mercury	0.002	0.002-0.05	--	70-80	<drinking water standards
Zinc	5.0	0.03-8.3	50	60	<drinking water standards

Aerated lagoons have been used as preapplication treatment methods since their low cost and low maintenance requirements are generally compatible with land treatment systems. Some reported maximum removal capabilities for indicator organisms and infectious agents are presented in Table 4.

Table 4. Maximum Removal of Enteric Microorganisms by Lagoon Systems [3,6,9]

Enteric microorganism	Removal, %
Coliforms	60-99.99
Fecal coliforms	99
S. typhi	99.5
Virus	99.99[a]

a. Laboratory study.

The capabilities of the soil to remove infectious agents from solution have been investigated under many conditions for microbiological agents. Essentially complete removal of fecal and total coliforms were shown for an experimental slow rate system [10]. Coliform removals to below detection have also been reported [11].

Virus removal in soil systems was reported to be greater than 99% at high flowrates up to 91 cm/d using a sand media [11]. Lesser flowrates can be expected to produce greater removals. A summary of observed travel distance through soil for virus from wastewater effluents ranged from 20 to 117 cm [11].

The die-off of wastewater microorganisms when applied to the soil surface or vegetation is of interest due to potential vector transport, site worker contact, or survival on harvested crops used for food [12]. Microorganism die-off occurs due to sunlight effects of desiccation and ultraviolet radiation. Some reported values are summarized in Table 5.

Table 5. Reported Survival Times of Enteric Microorganisms on Soils and Vegetation [7,13]

Enteric microorganisms		Survival time, d	Estimated die-off after 7 d, %[a]
Coliforms	Fodder	6-34	98
	Vegetables	35	90
	Soil surface	38	88
Shigella sp.	Fodder	<2	Below detection
	Leaf vegetables	2-7	Below detection
	Orchard crops	6	Below detection
Salmonella sp.	Fodder	12-<42	94
	Soil surface	15-46	93
	Leaf vegetables	1-40	98
	Orchard crops	0.75-<2	Below detection
Enterovirus	Leaf vegetables	15-60	89
E. histolytica	Leaf vegetables	2	Below detection

a. Calculated from median survival time.

Control of nitrate-nitrogen discharge to groundwater by land application systems involves system design based on nitrogen balance. Nitrogen removal occurs by crop uptake and microbial denitrification in a slow rate system [14]. The state-of-the-art of land treatment system design for slow rate systems can ensure nitrate concentrations satisfying drinking water requirements (10 mg/L as NO_3-N).

Land application practices reduce heavy metal concentrations to low levels. However, the cumulative application to the soil over the design life of the project should be within acceptable values [15].

Staffing Requirements

Staffing requirements were estimated for the alternative systems to determine degree of worker exposure to health affecting components. According to the methods published by the EPA [16] and an assumed flow of 11.4 m^3/s, both systems (slow rate and activated sludge) would require staffing of about six men per year (approximately 9,000 man-hours). Sludge disposal and vegetation planting and harvesting were not included.

ASSESSMENT

The initial step was to identify the significant wastewater components affecting health. The infectious agents of concern are Salmonella sp., Shigella sp., parasites, Escherichia coli, and viruses. Fecal and total coliforms are included since they are generally accepted for use as indicator organisms. Principal inorganic chemicals of concern are nitrates and trace metals.

Wastewater Treatment Interactions

The process steps for each treatment sequence were identified as to their potential (1) to remove a wastewater component from solution, or (2) to provide an opportunity for contact in the agent-host transmission cycle. The decrease in wastewater components through the treatment sequences were discussed previously. The potential for a health effect to occur was based on type of contact (intensity) and frequency of contact (duration).

Contact Interactions

The assessment method is presented based on contact intensity factors and contact duration factors. Contact intensity describes the relative potential of a contact producing illness. Contact duration estimates the frequency of these contacts based on annual exposure. The two factors are chosen to illustrate the difference between low and high intensity of contact and long and short durations of contact.

Contact Intensity. Incidental physical contact is the potential physical contact between a host body and an object that has been contacted by wastewater or the wastewater itself. This contact is the least likely to cause infection or illness.

Aerosol inhalation is the potential intake of aerosolized wastewater droplets through normal breathing. The factors for infection or illness require the transport of a viable wastewater pathogen in a proper size aerosol particle that will carry beyond the wetted perimeter of a site and still be proper size for potential inhalation.

Accidental ingestion is the potential of ingesting small volumes of water, as typified by recreational activities, such as swimming and water skiing.

Potable ingestion is the supply of water for domestic purpose which includes the daily ingestion for drinking, as well as food preparation, bathing, and other uses. This contact has the greatest potential to produce infection or illness based on contact factors alone.

Contact Duration. Contact duration factors were used to assess the exposure of the total population to various contact opportunities with the health affecting components of wastewater. The method is a compromise between estimating ingested volume of treated or untreated water, and incidental contact. The contact duration is considered to be the total annual man-hours of exposure (see Table 6). During the treatment sequence, the man-hours of contact are estimated from staffing requirements to satisfy operation and maintenance. For general public contact through recreation, municipal surface and subsurface water supply, and private subsurface water supply the following annual estimates were made:

Municipal water supply:

$$\frac{1\ h}{d} \times \frac{365\ d}{yr} \times 30{,}000 \text{ persons} = 1.1 \times 10^7, \text{ say } 10^7$$

This example assumes that the entire community (30,000 persons) has contact through municipal supply.

Private water supply (adjacent owners to land application):

$\frac{1\ h}{d} \times \frac{365\ d}{yr} \times 20$ persons $= 7.3 \times 10^3$, say 10^4

This example assumes that five families (four persons each) have contact from private wells.

In this example, activated sludge treatment preceded chlorine disinfection. Since the die-off by chlorination of various microorganisms varies according to an exponential relationship [17], the percent kill varied for a given chlorine dose and contact time. The estimated removals of other wastewater microorganisms based on 99.99% kill for E. coli and published K values are as follows: Salmonella sp., 99.99%; Shigella sp., 99.99%; poliovirus, 99%; hepatitis virus, 94%; and E. histolytica,, 20 to 45%.

Table 6. Estimated Contact Duration Factors, Annual Man-Hours

	Activated sludge		Land treatment	
Process step	Process	Contact duration	Process	Contact duration
0	Collection	Negligible	Collection	Negligible
1	Primary sedimentation	6×10^2	Headworks	2×10^2
2	Aeration and sedimentation	3×10^3	Aerated lagoon	2×10^3
3	Disinfection	2×10^2	Storage lagoon	6×10^2
4	River die-off	Negligible[a]	Field distribution	2×10^3
5	River dilution	Negligible[a]	Soil surface	5×10^2
6	Recreation	10^5	Soil column	Negligible
7	Municipal water supply	10^7	Groundwater flow	Negligible
8	--	--	Private or public water supply	10^4 or 10^7

a. All river contact assumed to occur during recreation.

RESULTS

Wastewater Components

Infectious Agents. The results of the example assessment for land treatment and activated sludge treatment and river discharge are presented for infectious agents in tabular form in Tables 7 and 8.

Nitrate-Nitrogen. The behavior of nitrate can be adequately controlled during system design so withdrawal from a potable water supply would not occur at levels greater than EPA drinking water standard (10 mg/L as NO_3-N). In cases where the groundwater is not potable, the design should not exceed the existing nitrate concentration or the drinking water limit, whichever is greater. For land application systems using an effluent recovery, the percolation of water is minor so groundwater quality changes would be minimal.

Trace Metals. Trace metals should pose no health concerns in the effluent portions of the wastewaters. Since typical values in raw wastewater are low, the removals by either treatment method would produce concentrations less than drinking water standards. The concentration of metals in sludges and their disposition may require additional consideration of health effects.

Trace Organics. The removal of trace organics would not normally occur in activated sludge treatment. The removal method would be dilution with the river water (assumed 20:1 for this example) and adsorption to settleable particles. Land application provides considerable contact with soil particles and opportunities for adsorption. The soil microorganisms provide further opportunities for microbial breakdown. In a site investigation at Muskegon, it was found that 56 organic compounds were detected in applied wastewater but only 5 were detected in the final effluent [18]. The use of chlorine disinfection forms halogenated organic compounds, some of which are thought to be carcinogenic. The capabilities of land treatment to provide high levels of removal without a (chlorine) disinfection step are noteworthy [19]. The elimination of a chlorine disinfection step with conventional treatment and discharge would generally not be accomplished without an increase in health risks.

Risk Evaluation

Site Workers. The annual man-hours of worker contact were estimated to be equal for a slow rate system and an activated sludge system (Table 6). This is not a direct indication of equal risk because wastewater concentrations are variable throughout the process steps. In addition, contact intensities for both systems are limited to only incidental and aerosol contacts. The onsite conditions change with each contact type, so comparisons are difficult to make.

Table 7. Land Treatment - Estimates of Infectious Agent Concentration

Process step	Fecal and total coliforms		Salmonella and Shigella		Virus		Parasites	
	Reduction[a]	Concentration	Reduction[a]	Concentration	Reduction[a]	Concentration	Reduction[a]	Concentration
Sewage collection	Initial conc.	3×10^7	Initial conc.	4×10^3	Initial conc.	1.9×10^3	Initial conc.	1×10^3[c]
Headworks	None	3×10^7	None	4×10^3	None	1.9×10^3	None	1×10^3
Aerated lagoon	99	3×10^5	99.5	20	99.9	1.9	99.7	3
Storage lagoon	99	3×10^3	99	0.2	99	1.9×10^{-2}	99	3×10^{-2}
Field distribution	None	3×10^3	None	0.2	None	1.9×10^{-2}	None	3×10^{-2}
Soil surface[d]	88-98	60-360	93-94	$(1.2-1.4) \times 10^{-2}$	89 on leaf vegetables	2.1×10^{-3}	Essentially complete (*E. histolytica*)	Below detection
Soil column	99.9999	3.0×10^{-2}	99.99	2.0×10^{-5}	99.99	1.9×10^{-6}	99.9999	3×10^{-7}
Underdrain recovery (50% recovery)[e]	50	1.5×10^{-3}	50	1.0×10^{-5}	50	1.0×10^{-6}	50	1.5×10^{-7}
River die-off	86-94	$(9-21) \times 10^{-5}$	54-83	$(1.7-4.6) \times 10^{-6}$	50-94	$(1.7-4.6) \times 10^{-7}$	54-83	$(2.6-6.9) \times 10^{-8}$
River dilution	95	$(4.5-11) \times 10^{-6}$	95	$(8.5-23) \times 10^{-8}$	95	$(8.5-23) \times 10^{-9}$	95	$(1.3-3.5) \times 10^{-9}$
Surface water supply or recreation	--	$(4.5-11) \times 10^{-6}$	--	$(8.5-23) \times 10^{-8}$	--	$(8.5-23) \times 10^{-9}$	--	$(1.3-3.5) \times 10^{-9}$

a. Expressed as % removal for each process step.

b. Expressed as No./100 mL.

c. Uses a high literature value; most report less than 50 organisms/100 mL.

d. Estimated after 7 days exposure on soil surface or fodder crop, respectively.

e. Recovery assumes 50% by mass basis, but considered as concentration. A river dilution would produce equal concentrations for 50% reduction by mass or concentration.

Table 8. Activated Sludge and River Discharge - Estimates of Infectious Agent Concentration

Process step	Fecal and total coliforms		Salmonella		Shigella		Virus		Parasites	
	Reduction[a]	Dose[b]	Reduction[a]	Dose[b]	Reduction[a]	Dose[b]	Reduction[a]	Dose[b]	Reduction[a]	Dose[b]
Sewage collection	Initial conc.	3×10^7	Initial conc.	4×10^3	Initial conc.	4×10^3	Initial conc.	1.9×10^3	Initial conc.	1×10^3[c]
Primary treatment	10-35	$(2.0-2.7) \times 10^7$	15	3.4×10^3	15	3.4×10^3	10-15	$(9.5-17) \times 10^2$	10-50	500-900
Secondary treatment	90-99	$(2.0-27) \times 10^5$	96-99	34-140	91-99	34-310	76-99	9.5-410	10	450-810
Disinfection (chlorine)	99.99	20-270	99.99	$(3.4-14) \times 10^{-3}$	99.99	$(3.4-31) \times 10^{-3}$	94-99	0.09-25	20-45	250-650
Surface water supply or recreation	None[d]	--	None[d]	$(2.9-31) \times 10^{-5}$	None[d]	$(3.0-7.0) \times 10^{-5}$	None[d]	$(0.06-30) \times 10^{-3}$	None[d]	2-15
Die-off in receiving water	86-94	1.1-38	54-83	$(5.8-62) \times 10^{-4}$	54-83	$(5.8-140) \times 10^{-4}$	50-94	$(0.06-120) \times 10^{-1}$	54-83	50-300
Dilution in receiving water	95	0.06-1.9	95	$(2.9-31) \times 10^{-5}$	95	$(3.0-7.0) \times 10^{-5}$	95	$(0.06-30) \times 10^{-3}$	95	2-15

a. Expressed as % removal for each process.

b. Expressed as No. organisms/100 mL.

c. Uses a high literature value; most report less than 50 organisms/100 mL.

d. A municipal water supply would normally employ chlorination prior to distribution, although it is not shown here.

Overall, the health risks are considered to be small, with neither producing a notable health risk. This is substantiated by a lack of any reports indicating increased health risk from occupational exposure by sewage treatment plant workers within the United States.

General Public. The relative health risk potential to the general public after the treated wastewater effluents are discharged to surface waters or groundwaters is shown in Table 9. The basis for comparison is the estimated concentration for an activated sludge discharge to surface water and land treatment discharge to groundwater, withdrawal and discharge to surface waters. Based on the estimated concentrations of infectious agents, the land treatment system decreased the relative health risk potential by providing greater removals.

Table 9. Summary Comparison of Health Risk Potentials No. Organisms/100 mL

	Relative concentration at surface water supply[a]	
Wastewater agent	Land treatment	Activated sludge
Coliforms	$(4.5\text{-}11)\times10^{-6}$	0.06-1.9
Salmonella sp.	$(8.5\text{-}23)\times10^{-8}$	$(2.9\text{-}31)\times10^{-5}$
Shigella sp.	$(8.5\text{-}23)\times10^{-9}$	$(3.0\text{-}7.0)\times10^{-5}$
Virus	$(8.5\text{-}23)\times10^{-9}$	$(0.06\text{-}30)\times10^{-3}$
Parasites	$(1.3\text{-}3.5)\times10^{-9}$	2-15

a. Provides water for municipal supply as well as recreation.

A summary comparison of removal mechanisms for slow rate and activated sludge systems is provided in Table 10. The decreased risk with slow rate land treatment from infectious agents was shown in comparing estimated concentrations. The slow rate land treatment removal mechanisms provide removals of nitrates, trace elements, and trace organics. An overall decreased relative health risk occurs under these conditions.

Table 10. Summary Comparison of Removal Capabilities

Wastewater component	Slow rate land treatment	Activated sludge treatment
Bacteria		
Salmonella sp.	+	+
Shigella sp	+	+
E. coli	+	+
Virus, in general	+	-
Parasites		
E. histolytica	+	-
G. lamblia	+	-
Nitrate	+	-
Trace elements	+	0 to +
Trace Organics	0 to +	-

Note: Comparisons based on the following notations:
+ Positive removals with decreased health risks
- Minor removals with unchanged health risks
0 Behavior unknown, but partial removals have occurred.

DISCUSSION

Fail-Safe Aspects

The example assessment presented a series of removal mechanisms that showed with well maintained and operated treatment systems, the health risks are greatly decreased. The reliability of the unit processes in conventional treatment gives an indication of the fail-safe aspects. Mechanical equipment, such as pumps, feeders, and mixers, provide continuous and adequate performance only if maintained and operated properly. Smaller systems generally do not provide continuous staffing. System upsets are most apt to occur. The resulting inefficiency depends on engineering design, but may result in minor or major contamination of receiving waters. Disinfection processes are equally subject to upset with results ranging from overchlorination with potential formation of halogenated organic and chlorine toxicity, to underchlorination with insufficient pathogen kill.

Land treatment systems are also subject to short circuiting in the preapplication treatment such as aeration and storage lagoons, but the overall treatment result after passage through a soil column is essentially unaffected by applied wastewater concentration. In addition, a slow rate system is fail safe in that if too much water is applied, the soil will not accept the water. Further, the water must pass through the soil to reach to groundwater or the underdrains. A properly designed land treatment system should provide reliable treatment comparable to or greater than that provided by a well designed activated sludge plant.

CONCLUSIONS

1. Results indicate that under well maintained and operated conditions, both conventional and land treatment systems provide a large measure of safety for public health. Slow rate land treatment systems offer greater protection against parasites and viruses, trace metals, nitrate, trace organics, and halogenated organics.

2. Adequate removal of parasitic eggs and cysts requires positive measures, such as filtration, or long detention times in ponds or storage lagoons. As such, the land treatment alternative offers greater protection with regard to health risks.

3. Conventional treatment and disinfection is also less effective for viruses than treatment by land application. Treatment processes with longer detention times, such as in ponds and storage lagoons, afford better removals than conventional activated sludge treatment.

4. Land treatment systems are less susceptible to failure or upsets than conventional treatment systems, especially for small systems.

5. Observations of interactions not comparing the treatment processes include:

a. The health hazard to site workers does not appear greater than to the general public. The contact by site workers is generally limited to incidental contact or aerosol inhalation. The health risk to site workers is thus limited to infectious agents.

b. The use of wastewater for irrigation of food crops may provide a greater risk than irrigation of nonfood crops since the agent-host transmission cycle will be shorter and positive removals by soil are not used. Die-off varies according to type of crop and infectious agent. However, present application of wastewater to crops as suggested in California regulations, provides a safety factor of 10^8 to 10^{13} over the last reported incidents (early 1900s) of disease transmittal using "night soil" on food crops.

c. Trace metals are difficult to evaluate as to health effects. Intake with water sources should not pose problems because the typically low values in municipal wastewaters are generally reduced to below drinking water standards by wastewater treatment.

d. No estimation can be made of the health hazards resulting from low level of exposure to trace organics. The dose response and health effects are unknown at this time.

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HEALTH CONSIDERATIONS

A SEROEPIDEMIOLOGIC STUDY OF WORKERS ENGAGED IN WASTEWATER COLLECTION AND TREATMENT

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Exposure to wastewater used in land treatment may have adverse health effects but there are no definitive studies of persons living near or working at land application sites. However, studies of wastewater collection and treatment workers who have direct contact with wastewater may suggest the potential hazards for those with indirect contact.

The University of Cincinnati Medical Center is currently engaged in a comprehensive study of wastewater treatment industry workers in the U.S. This U.S. EPA-supported study is a prospective seroepidemiologic investigation of wastewater workers in three cities. The focus of the study is on possible health hazards due to the viral and bacterial content of the wastewater.

The study involves collection of blood specimens and throat and rectal swabs every three months for laboratory analyses, annual health examinations, enviromental monitoring and collection of illness information. The central feature of the laboratory studies is a battery of tests for the determination of the levels of viral and bacterial antibodies and immunoglobulins in serum specimens. The testing is designed to detect changes that may occur after a person becomes occupationally exposed to wastewater.

Initial results have revealed higher levels of several viral antibodies in sewage-exposed workers in comparison with a control group. Some of these results may be indicative of subclinical infections. These and some of the other results obtained during the first three years of this four year study are presented. Efforts are made to relate airborne bacterial levels at wastewater treatment plants to those at a land application spray irrigation site.

Introduction

The purpose of this study is to obtain an assessment of the health risks of populations occupationally exposed to wastewater collection and treatment. Information regarding the human health risks associated with contact with wastewater and related materials brought about by occupational exposure, by residing near wastewater treatment plants and land application facilities, or by recreational pursuits in water receiving wastewater effluents is limited. However, assumptions concerning these risks are providing motivation for the promulgation of state and federal standards designed to protect populations from these various exposures. The growing emphasis on the land application of wastewater and sludges as a viable method of waste utilization increases the need for reliable and up-to-date information on the health risks, if any, involved. The focus of this study is on the pathogenic bacterial and viral agents in wastewater.

The methodological tool for this assessment is a prospective seroepidemiologic study of populations that are intimately exposed to wastes by virtue of their employment as municipal sewer maintenance workers or as activated sludge wastewater treatment plant workers. Specific objectives of the project are:

1. To determine whether wastewater workers develop specific bacterial, viral, and parasitic infections due to occupational expsoure to sewage.

2. To determine the immunologic response among workers, presumed to be exposed to a high level of antigenic stimulation, i.e., wastewaters.

3. To determine whether wastewater workers serve as a reservoir of certain infections and, if so, whether members of the workers' families are affected.

4. To determine the effect of exposure to aerosols generated by the activated sludge treatment process.

5. To determine the concentration of bacterial aerosols at wastewater treatment plants.

An underlying objective of the study is the determination of the sensitivity of various elements of the seroepidemiologic approach for the detection of wastewater-related health effects. Such determination would permit an assessment of the potential application of this methodology to the study of health risks associated with other population exposures to wastewaters. The latter would include persons working with or living near wastewater land-disposal facilities, persons living in the vicinity of wastewater treatment plants and persons engaged in recreational use of bodies of water receiving waste effluents.

Related Studies

Recent literature searches have revealed little evidence of occupational health problems associated with wastewater pathogens (1,2). The few situations where adverse health effects have been found were associated with sewage farming with untreated wastes or with other contact with essentially raw sewage (3,4,5). A comprehensive retrospective study of Berlin sewer workers by Anders failed to yield any health problems associated with sewage pathogens (6). Residents of Israeli kibbutzim where crop irrigation with sewage is practiced experience certain gastrointestinal diseases at rates 2-4 times higher than people in kibbutzim not practicing such irrigation (7). From the information available in that report it is not possible to ascertain the pathway(s) of the infections. The unique nature of kibbutzim with their closely-knit living patterns complicates transferal of the kibbutzim study results to U.S. situations.

In Sweden, Rylander and his associates describe what they term a "sewage workers syndrome" in workers in a sludge drying operation producing dust mostly in the respirable size range (8). Symptoms of this syndrome include eye discharges and fever usually developing several hours after the end of the work day. Laboratory testing revealed elevated levels of the immunoglobulin IgA and of white blood cells. Their investigation is being extended to other wastewater treatment plants. Layton has reported that new employees in the wastewater industry sometimes experience dysentery during the first year of employment but appear to have no unusual ill effects thereafter (9). A preliminary report of a study of wastewater treatment workers in Copenhagen, Denmark suggests that the workers experience elevated immunoglobulin levels and elevated levels of antibodies to Weil's disease (10). Earlier reports in Germany (5) and Ceylon (4) also found elevated levels of antibodies to Weil's disease which can be transmitted by the urine of infected rats and which in its earlier stage may resemble influenza. The U.S. Environmental Protection Agency is currently supporting two studies aimed at determining possible health effects to persons living in neighborhoods surrounding activated sludge treatment plants (11,12). The lack of information about whether neighbors of wastewater treatment plants are themselves at risk has provided further impetus to determine the risk, if any, to wastewater workers.

Overall Study Design

The study consists of an intensive serologic survey correlated with epidemiologic and clinical data on the study populations. The central feature of the design is an evaluation of the effects of occupational exposure to wastewater over at least a 12-month period based on the levels of specific viral and bacterial antibodies in sera collected over that time period. In order to help separate the effect of the occupational exposure from that associated with other possible disease pathways, appropriate control groups are utilized and in addition, many of the wastewater workers are recruited into the study at the time of their initial employment in the wastewater treatment industry. Results of the serological, epidemiological and clinical aspects of the study are correlated with data from an environmental monitoring program. The study is scheduled to be completed in the spring of 1979.

On a quarterly basis, blood samples, throat swabs and fecal samples are collected from participants in the study. Monthly health diaries are maintained by the workers and their families. At times of illness additional throat swabs and/or fecal samples are requested. On a monthly basis a project staff member, generally a nurse, contacts the family by telephone regarding illnesses. Periodically job locations of participants are visited to determine types and levels of exposure and other work conditions. Work site air is monitored periodically for bacterial concentrations and wastewater samples at the wastewater treatment plants are analyzed for bacterial and viral concentrations. On a yearly basis study volunteers are given a medical evaluation that includes hematology, urinalysis, pulmonary function testing, blood chemistry (including tests of liver and kidney function) and examination by a physician.

Epidemiological Aspects

Selection of Study Populations

As initially conceived the study goals were limited to the first three specific objectives listed above. In Cincinnati, the occupational group thought to have greatest exposure to wastewater was sewer maintenance workers who were thus chosen as the exposed population. These workers have been maintaining the combined sanitary and storm sewer system which predates by many years the wastewater treatment plants, the larger of which were built since 1950. Highway maintenance workers of the Cincinnati Public Works Department were selected as the control group.

About one year after this research began its goals were expanded to include a determination of the health effects associated with the dispersion of aerosols generated by the activated sludge wastewater treatment process (Specific Objectives 4 and 5). At this time the study design was expanded to include two additional exposed populations groups: (1) fifty (50) men at the Cincinnati Mill Creek Sewage Treatment Plant which was in the process of being expanded from primary wastewater treatment to include the activated sludge process; (2) and a total of one hundred (100) men newly employed at activated sludge treatment plants. The latter group were recruited from Cincinnati, Ohio; Chicago, Illinois; and Memphis, Tennessee. As additional control groups, water treatment plant workers were selected in Chicago and in Memphis, workers at the Memphis Light, Gas and Water Division.

Personal Data

Illness data are obtained from: monthly family health diaries, telephone contacts and on-the-job contacts. For each participant, the age and sex of each other person in the household is obtained in addition to household income, number of years of school for the worker, age, race, job classification, and previous occupational exposure to wastewater, if any. At the time of joining the study a family medical history questionnaire is administered by a member of the research team. In addition, at the time of the annual health evaluation, a more detailed medical history questionnaire is given.

Job Exposure

Each worker is visited on his job several times during the study to determine type of job, frequency and intensity of wastewater contact and aerosol exposure contact, and other related work conditions. Results of these observations, together with results from the aerosol monitoring, are used to categorize the worker on a relative exposure scale for direct wastewater contact and one for aerosol exposure.

Serologic Survey

Results of a serologic survey of sera collected quarterly provides the basic core of data for this study. The sera are analyzed for antibodies to 33 viruses or groups of viruses and 9 bacteria, and for 3 classes of immunoglobulins. The purpose is to determine two things: a) whether there are differences in antibody levels between test and control groups and b) whether there are significant increases in antibody levels within a group of workers over a period of time, indicating infection. In particular, we are interested in the relationship between an increased antibody level in a volunteer and the presence of illness symptoms. The absence of these symptoms indicates a subclinical infection.

Antibodies are immunoglobulins

which have specificity for particular antigens such as specific viruses and bacteria. Immunoglobulins (Ig) are one of the body's defenses against infecting agents. IgG, the most abundant immunoglobulin class, carries the bulk of the burden of this defense. IgA is the predominant Ig in various seromucous secretions, where it defends various internal and external surfaces of the body against invasion by microorganisms. IgM antibody develops early in an infection and later decreases in concentration followed by an increase in antibodies of the IgG class.

Environmental Monitoring

The monitoring program consists of viral and bacterial analyses of wastewater samples and bacterial analyses of aerosols. The purpose of the aerosol sampling procedure is to provide data for the estimation of worker exposure to airborne bacteria. The viral analyses of wastewater will be correlated with the virus serologic survey results. The virus assay of wastewater in all three cities is being performed under a contract with the Metropolitan Sanitary District of Greater Chicago.

Six-liter twenty-four hour composite samples for virus assay are concentrated using an aluminum hydroxide-continuous flow centrifuge technique (13). Virus content is estimated using a plaque assay procedure using three cell cultures: (1) BGM (Buffalo Green Monkey Kidney Cells), (2) WI-38 (Human Diploid Cell Strain), and (3) PMK (Primary Monkey Kidney Cells).

Six-stage Andersen samplers are used to collect the aerosols from about 0.4 M^3 of air (14). Each sampler contains six molded Andersen glass petri dishes containing 27 ml of plate count agar. A replicate plate method is used for determining the total coliform (TC), fecal coliform (FC) and fecal streptococcus (FS) counts on the petri dishes from the Andersen samplers. Bacterial analyses of wastewater and aerosols are performed in Chicago by the Metropolitan Sanitary District of Greater Chicago, in Memphis by the Memphis State University, and in Cincinnati by the bacteriologist on the study staff. Bacterial analyses are performed according to Standard Methods (15).

Results

A total of about 400 workers are currently involved in the study. Over 100 of these volunteers were recruited at the time they began employment in the wastewater industry. Preliminary results will be presented on the environmental monitoring, viral and bacterial serology and immunologic findings. Clinical data have not been analyzed.

Environmental Monitoring

Monitoring for bacterial aerosols has been conducted in all three cities. A comparison of bacterial aerosol levels at wastewater treatment plants in this study with levels at land application projects will assist in relating potential human exposure levels at these two types of facilities. In order to accomplish this, a summary of airborne fecal coliform data to date from this study is compared in Table 1 to data from an environmental survey at a project involving the spray irrigation of wastewater (16). Results of wastewater analyses are also included to reflect the differences in the wastewater being aerosolized.

Median fecal coliform levels over basins and downwind from the activated sludge plants were 7-14/M^3 while those at comparable distances downwind of the spray irrigation site were only 1.0/M^3. The median fecal coliform level in the wastewater at the activated sludge plants was 160 x 10^4/100 ml, compared to that from the spray irrigation site, 8.1 x 10^4/100 ml. A number of factors must be considered in comparing these data. Rates of aerosolization of wastewater at the activated sludge plants are much lower than that at the spray irrigation site. Wastewater treated varies from 30 MGD to about 1.0 BGD at the activated sludge plants, while that at the spray irrigation facility averaged 1.4 MGD. In addition, the high volume air samples used at the spray irrigation site yield higher fecal coliform values than 6-stage Andersen samplers used in this study. Considering all of these limitations it is concluded that airborne fecal coliform levels appear to be higher at the activated sludge plants than at the spray irrigation facility tested.

Virus Serology Survey

Serologic data are based on concentrations of antibody in serum and expressed as a titre. This titre is the highest dilution level which still produces a reaction, and is expressed as the reciprocal of the dilution. Test procedures used were complement fixation

(CF), hemagglutination inhibition (HI), and microneutralization (MN) (17).

Of interest in the serological survey are differences in antibody titre levels for an individual over a period of time and differences between worker groups at a point in time. Both of these comparisons depend on the laboratory procedures being done at the same time. Testing of at least two sera for the Cincinnati experienced sewer maintenance and highway maintenance workers, and in some cases Cincinnati experienced sewer treatment workers, has been completed and statistically analyzed for 33 viral antibodies and for Hepatitis B antigen.

Tests were performed for antibodies to Polio 1, 2, 3, Coxsackie B1 - B6, adenovirus, reovirus, cytomegalovirus, and Herpes Simplex by CF; Echo 7 and 12 by HI; and Coxsackie A7, A9, A16, Echo 1-6, 8, 9, 11 - 14, 19, 21, 24 and 30 by MN. Because a very high number of negative results were obtained for Polio 1, 2, 3, Coxsackie B1 - B6 by CF, these viruses were retested by MN. For Polio 1, 2, and 3, the retesting gave a wide range of titre levels, but no significant differences between groups. For Coxsackie B1 - B6, the retesting resulted in generally more positive results except for Coxsackie B6. However, there were no significant differences among groups for results by MN, whereas the CF method had resulted in significant differences for Coxsackie B1 in October 1975. An example of a set of sera analyzed by both methods for antibodies to Polio 1 is shown in Table 2.

The data for all sera were tabulated in a contingency table of worker groups vs. titre levels, as shown in the example in Table 2, and statistically analyzed by the chi-square test (18). Negative laboratory results were reported as being less than the minimum detection level (<2 for MN, <4 for CF and HI) and were so tabulated in the contingency table. When significant differences were found, a geometric mean of the titre levels was used to facilitate interpretation. For negative results, an arbitrary value of one-half the minimum was assigned; for example, a result of <4 was assigned the value of 2.

Antibody levels were found to be significantly different at the $p \leq .02$ level between groups tested for four viruses. A listing of these viruses, the dates of the sera collection and the geometric mean are presented in Table 3. In no case in our testing were the titre levels of highway maintenance workers significantly higher by the chi-square test than those of sewer maintenance workers. For Coxsackie B3 virus, antibody levels for the sewage treatment workers were significantly higher than those for sewer and highway maintenance workers. The latter two groups were not significantly different from each other. Significant differences had previously been reported for additional viruses (19) but since these differences were not evident in later testing they were deleted from Table 3. When available, results of wastewater viral assays will be compared with the virus serology data.

No differences have been found among the three groups with regard to family size, number of school-age children, or illness rates. Therefore, the control and test groups appear to be adequately matched for these factors. If so, the higher virus antibody levels in the sewer maintenance workers presumably are the result of occupational exposure to sewage. An acquisition of viral antibodies in response to occupational exposure to sewage has been hypothesized (2); data from this study are the first to support this suggestion.

Results of the serologic survey were also analyzed to determine changes in titre level for each worker over a period of time. Titre level increases of 4-fold or greater (referred to as seroconversions) usually indicate infection. So far, we have not found any significant differences in seroconversion among groups.

Bacterial Serology

Tests for antibodies to Salmonella A, B, C, D and E, and Leptospira Pool 1, 2, 3 and 4 were performed on sera for three groups of experienced Cincinnati workers for October 1976, and two groups of experienced Chicago workers for February 1977. The test methods used were passive hemagglutination for Salmonella (20) and slide agglutination for Leptospira (21). All Leptospira test results were negative. No significant differences were found between groups, in either city, for Salmonella antibodies.

Immunology

Levels of IgG, IgA, and IgM were measured by radial immunodiffusion (22) in sera obtained from sewer maintenance, highway maintenance, and sewage treatment workers in Cincinnati during July of 1975 and 1976 (Table 4). Levels of IgG in the highway workers were significantly higher than those in the sewer workers during both years. No consistent differences in IgA or IgM levels were observed when the sewer and highway maintenance groups were compared. In 1975 IgM levels were higher in the sewer maintenance workers than in the highway maintenance group. This difference was not evident in 1976. IgM levels were found to be lower in both groups during the second year. Another difference observed was that the IgA level in the sewage treatment workers was significantly lower than that of the sewer maintenance group. In contrast, IgG and IgM levels in this group were not significantly different from the other two groups.

For Chicago workers, immunologic analyses were performed on sera collected in January 1977 from inexperienced and experienced sewage treatment workers, and the water treatment workers. No significant differences were found.

Conclusions

Based on the results thus far it appears that sewer maintenance workers have experienced more viral infections than the highway maintenance workers. Although more limited data have been presented for sewage treatment plant workers, increased infection rates are not evident as they were for sewer maintenance workers. Median fecal coliform levels over the activated sludge aeration basins and up to about 100 feet downwind were 7-14/M^3. In this study there appears to be no significant increase in Ig level in individuals exposed to sewage in comparison to individuals exposed to a highway environment.

Testing of sera collected over about a one-year time period beginning with the time of initial employment are now being analyzed. Job evaluations and environmental monitoring for bacterial aerosols will be used to assign relative wastewater and aerosol exposure levels to the sewage-exposed workers. An effort will then be made to determine if dose-response relationships can be established.

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Table 1

Comparison of Total Fecal Coliform Levels at Activated Sludge Plants[a] and a Wastewater Spray Irrigation Facility[b]

	AEROSOL LEVELS #/M^3 [c]			WASTEWATER [d]
	Activated Sludge Treatment Plants [a]			#/100 ml x 10^4
	Upwind (12)	Over Basins (13)	Downwind (11) [e]	
Maximum	28	146	141	8000
Median	0	14	7	160
Minimum	0	0	0	1.1

	Wastewater Spray Irrigation Facility		
	Upwind (16)	Downwind [e]	
Maximum	<1.0	12.2	18.6
Median	below detection	1.0	8.1
Minimum	below detection	below detection	2.4

(a) Chicago (West Southwest and Calumet), Cincinnati (Mill Creek), Memphis (North)

(b) Pleasanton, California (Reference No. 16)

(c) Number in parenthesis indicates number of aerosol samples

(d) 18 samples from activated sludge plants, 17 from spray irrigation facility

(e) Up to 30 meters

Table 2

Contingency Table for Antibodies to Polio 1 October 1976

Comparing Test Methods

SM = Sewer Maintenance Workers
ST = Sewage Treatment Workers
HM = Highway Maintenance Workers

Complement Fixation Test

	Titre Level				
	<4	4	8	16	32
SM	39	0	0	0	0
ST	49	0	0	0	0
HM	47	0	0	0	0

Microneutralization Test

	Titre Level								
	<2	2	4	8	16	32	64	≥128	Total
SM	3	0	7	6	4	7	9	6	42
ST	3	5	5	5	4	8	9	3	42
HM	6	2	4	4	11	9	7	6	49

Table 3

List of Viruses for Which Antibody Levels Were Found to be Different ($p \leq 0.02$) in Cincinnati

Work Groups*

Virus**	Date	Difference	Geometric Mean SM	HM	ST
Echo 4	July 1975	SM > HM	2.7	†	n.t.
Coxsackie B1	October 1975	SM > HM	†	†	n.t.
Echo 30	July 1976	SM > HM	4.1	†	n.t.
	January 1977		3.9	†	n.t.
Coxsackie B3	October 1976	ST > (SM & HM)	†	†	†

* SM = Sewer Maintenance, ST = Sewage Treatment and HM = Highway Maintenance

** Test Types: Complement Fixation for Coxsackie B viruses
Microneutralization for all others

† Geometric mean is below minimum detection level

n.t. Not tested

Table 4

Immunoglobulins (mg/dl) Among Controls and Sewage-Exposed Workers (19)

Group	No.	IgG	IgA	IgM
1975:				
Highway Maintenance	69	2857* (716)	281 (1.6)	95† (1.9)
Sewer Maintenance	41	2176* (728)	327 (1.6)	126† (1.7)
1976:				
Highway Maintenance	53	2620‡ (704)	233 (1.5)	70 (1.7)
Sewer Maintenance	33	2185‡ (707)	271§ (1.4)	68 (1.4)
Sewage Treatment	51	2496 (715)	171§ (2.5)	76 (1.8)

Figures in parenthesis represent standard deviation (S.D.) for IgG and geometric S.D. for IgA and IgM. Geometric means and S.D.s are given for IgA and IgM because calculations were done on log-transformed data.

*$p < 0.001$; †p=0.018; ‡p=0.007; §p=0.002 (many other differences were significant, but only those for data relating to the same year are shown).

HEALTH CONSIDERATIONS

MICROBIOLOGICAL AEROSOLS FROM A FIELD SOURCE DURING SPRINKLER IRRIGATION WITH WASTEWATER

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Measurements were made of the strength and dispersion of bacterial aerosols resulting from land application of chlorinated, ponded wastewater by spray irrigation. An approximately square 2.1 hectare (5.2 acre) area was covered by 96 Rain Bird impact sprinklers, thus creating a multi-point or field aerosol source. Andersen viable-type and large volume electrostatic precipitator air samplers were deployed upwind and on 3 m centers in each of three downwind transects: 30 m (or 21 m), 50 m (or 41 m) and 200 m. In some runs, sampler locations at 4.6 m and 9.0 m elevation, 50 m downwind, were used in lieu of the 200 m location. In four runs, water to be sprayed was seeded with fluorescent dye to characterize the aerosol cloud without the effect of biological decay. During aerosol studies, continuous on-site meteorological measurements were made, and wastewater chemical parameters were monitored.

Bacterial aerosols (standard plate count) at 30 m ranged from 46 to 1582 colony forming units (cfu)/m^3, mean = 485/m^3). This value was reduced by 15% at 50 m and by 92% at 200 m. Aerosol strength was always greatly reduced at the elevated sampler stations; mean reduction from 1.7 m to 9.0 m elevation, 95%. A two- to three-fold variation among simultaneous samples within transects was observed, although in laboratory tests using wastewater aerosols sampler variability was within statistical expectations. The median diameter of bacteria-beaing particles was 2.6 μm at 30 m downwind. Seventy-five percent of such particles fell within the range of efficient pulmonary deposition, 1 to 5 μm. Dye aerosols were two- to three-fold greater in relation to source strength than bacterial aerosols, thus providing an estimate of bacterial aerosol dieoff.

INTRODUCTION

Land treatment of wastewater by spray irrigation is an important alternative to advanced wastewater treatment. From a public health standpoint, this practice raises the question of the potential dispersal of pathogenic bacteria, viruses and other organisms, especially through downwind travel of infectious aerosols. Some aspects of this problem have been addressed in several studies, in which data were obtained on the travel of bacterial (1-7) or viral (6,7) aerosols. The present effort was undertaken to provide measurements of the strength, dispersion, decay and particle size characteristics of bacterial aerosols generated by a field source spray irrigation process. A concomitant objective was to provide simultaneous source strength and meteorological data adequate for predictive modeling of aerosol plume dispersion, thereby facilitating development of design criteria for protection of human health.

This study forms part of a comprehensive evaluation of a demonstration land wastewater treatment site for U.S.

Army Corps of Engineers recreational areas. The study, described in more detail elsewhere (8), was undertaken July-August 1976 as a joint effort of the U.S. Army Cold Regions Research and Engineering Laboratory, Hanover, NH, and the U.S. Army Medical Bioengineering Research and Development Laboratory, Ft. Detrick, Frederick, MD. Application to land by spray irrigation is used as the final step in treatment of a relatively weak domestic wastewater.

The study differs greatly from the earlier experiments in the nature of the spray source. Here, 96 spray heads were used, whereas in other studies only a single spray head or a line source was used.

In this study, most sampling equipment was massed at two close-range distances downwind, so as to provide multiple observations at these critical points. This provided better assessment of sampler-to-sampler variability and better definition of the early part of the curve of aerosol strength vs. downwind distance.

MATERIALS AND METHODS

Site Description and Treatment of Wastewater

The project studied treats wastewater from a campground at Deer Creek Lake State Park, OH, situated 56 km (35 miles) southwest of Columbus, OH, and 8 km (5 miles) south of the town of Mt. Sterling, OH. The spray fields are in an open, fairly level, area.

Raw wastewater is pumped to a stabilization lagoon with a mean depth of 1.7 m (5 ft) and a mean residence time of 90 days. The effluent then passes through a chlorine contact chamber (1 hr contact time, 2 ppm total residual chlorine) into a holding basin with a mean residence time of 4-6 days. It is then pumped to the spray field where a grid of spray heads was used covering an approximately square 2.1 hectare (5.2 acre) area (Fig. 1). This area is served by eight lateral wastewater distribution pipes, each bearing 12 Rainbird spray heads. The distance between laterals is 18.3 m and between spray heads on each lateral, 12.2 m. Sprinkler heads are 0.6 m above ground level, with a nozzle size of 0.79 cm (5/32 inch) internal diameter. Pressure measured at the spray nozzles was 4.1-4.2 kg/cm^2 (56-59 psi) and remained constant across the spray field during testing. The flow was about 18.9 liter (5 gal)/min/nozzle for a total of 1910 l/min (580 gal/min). The area wetted by spray from a single head averaged 13 m (42.5 ft) in radius. while, at maximum, the spray arc reached a height of 2.7 m (9 ft) above ground level.

Field Methods

Meteorological Instrumentation and Measurements. The following parameters were measured at both 2 m and 20 m elevation: wind speed and direction, air temperature, relative humidity. Evaporation, precipitation, solar radiation and atmospheric pressure were also measured, and visual observations were made of sky conditions and cloud cover.

An atmospheric stability assessment was made at the start and end of each test run according to basic procedures used in making plume dispersion estimates as suggested by Turner (9) and Pasquill (10).

Sampler Positioning and Operation. Aerosol sampling runs were not attempted under unstable atmospheric conditions nor during periods of precipitation, shifting wind direction or wind speeds above 6.7 m/sec (15 mph).

Airborne bacteria were collected both with Andersen (11) viable-type stacked-sieve samplers, loaded with standard methods agar (BBL) in disposable plastic Petri dishes, and with high-volume electrostatic precipitator samplers (Litton Model M) through which 100 ml 1/2-strength m-plate count broth (Difco) was continuously recirculated at 10-12 ml/min. Airflow through Andersen samplers was regulated at 28.3 l/min by critical orifice. Litton samplers were operated at 1000 l/min and at an electrostatic potential of 13 to 14 kV. Exposed sampling fluid was held in ice until testing.

Sterilization of the collection discs and tubing of the Litton samplers was accomplished using 0.1% NaOCl, pH 6.5, followed by sequential flushing with 1.0% sterile sodium thiosulfate and sterile water.

During each sampling run, samples of effluent wastewater were taken at the spray nozzle at several points in time and composited.

A typical downwind sampler array is illustrated in Figure 1. The first, or near, row of five aerosol samplers was positioned 21 or 30 m from the most downwind row of sprinkler heads. A second

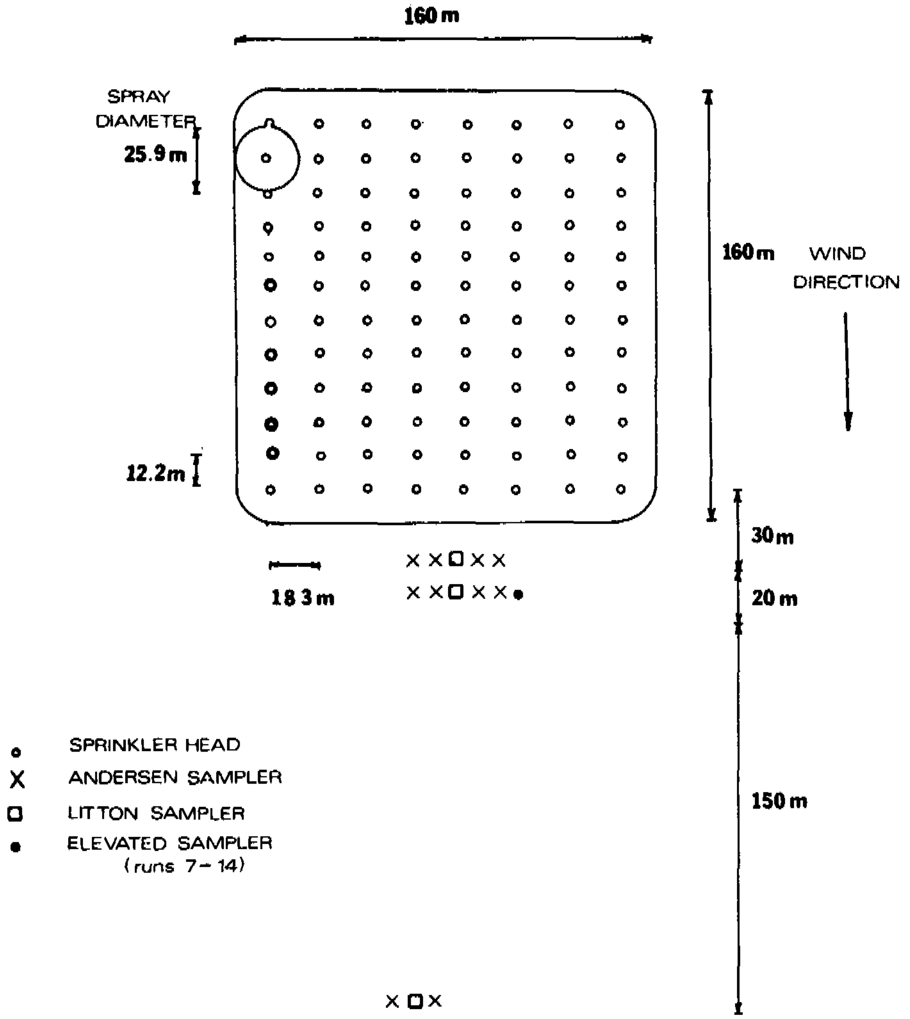

FIGURE 1: Typical downwind sampler array and spatial relations of operating spray heads.

or intermediate row of five samplers was located 20 m downwind of the first row. All sampler rows were aligned perpendicular to the average wind direction. Samplers were elevated on 1.7 m stands and set 3 m apart. A Litton sampler occupied the center of each row, and it was flanked at the 3 m intervals by either four Andersen (microbiological runs) or four all-glass impinger (AGI-30, dye runs) samplers. Two Andersen or AGI samplers were stationed upwind and two at an additional downwind site. To attain a vertical aerosol profile, during runs 7-14, these two small samplers were raised to 4.6 and 9 m elevations on a tower located at one end of the 50 m downwind sampler row. In all other runs, they were located 200 m downwind. Additional Litton samplers were located at the upwind and 200 m stations. The positions of individual samplers and pumps within the above-described pattern were rotated each sampling day according to a random sequence.

During runs using tracer dye, disodium fluorescein (uranine) solution, 5% in distilled water, was injected at a constant rate, into the line leading to the spray field. Air sampling for recovery of fluorescein aerosol was delayed for an interval of time determined by prior experimentation to allow the dye to reach the most distant spray heads and to optimize dye aerosol distribution throughout the field. During dye runs, Andersen samplers were replaced by AGI's. Distilled deionized water was used as collection fluid in both AGI and Litton samplers. Grab samples were taken at the spray nozzle at 2 to 3 minute intervals. These were not pooled, to permit a check on possible fluctuations in fluorescein level during the injection period.

Laboratory Methods

Estimation of Standard Plate Count and Aerosol Particle Sizes. Standard methods agar was used for all standard plate count determinations performed on the Litton and composite wastewater grab samples, using conventional plating methods. Plates used for this purpose and exposed Andersen sampler plates were incubated at 35°C for 48 hours.

From each Andersen sample, a median particle diameter (12) and quartile diameters were derived.

An estimate was made of the percent of each sample falling within the range 1.0 to 5.0 μm, as this has been shown to be the range of efficient deposition in the human pulmonary system (13).

Other Organisms. Total and fecal coliform and fecal *Streptococcus* levels in composited wastewater samples were estimated by the Millipore membrane filter method using m-Endo (Difco) and M-FC (BBL) broths and m-Enterococcus agar respectively, utilizing Standard Methods (14). The total coliforms were incubated at 35°C for 22-24 hours, fecal *Streptococcus* for 48 hours, and fecal coliforms, in a 44.5°C water bath for 22-24 hours.

Fluorometry. Following sampling runs using sodium fluorescein, all wastewater and aerosol samples were assayed fluorometrically against known sodium fluorescein standards in a Perkin-Elmer fluorescence spectrophotometer, Model 204.

Physical and Chemical Wastewater Parameters. Samples of wastewater leaving the second lagoon for the spray field, were tested for the following parameters (samples were taken simultaneously with aerosol test runs): hardness, chemical oxygen demand (COD), total organic carbon (TOC) and total and free chlorine.

Standard Methods (14) was followed in performance of all tests; total chlorine determinations were performed using the orthotolidine (Hach) method.

RESULTS

Meteorology

The majority of runs were made under stable atmospheric conditions. Runs 3, 6, 21 and 24 were performed under less stable conditions. Air temperatures during runs ranged from 20°C to 32.6°C (68°F to 90.6°F), relative humidities from 43 to 95 percent and mean wind speeds from 1.3 to 6.7 m/sec (3 to 15 mph). Runs 9 and 19 were conducted at dusk and runs 5, 14 and 25 under conditions of low solar radiation.

Wastewater Characteristics

Total residual chlorine levels in water used for spraying ranged from 0.1 to 0.4 mg/l. The water was moderate in hardness, 163-198 mg/l, and alkaline, pH 7.0 to 8.7.

Standard plate counts varied from 5.8×10^3 to 6.6×10^4/ml. Total coliforms averaged 1.8×10^2/ml, exceeding fecal coliforms by more than two orders of magnitude. Of a sample of 57 presumptive coliforms, 11 percent were classified as *Klebsiella*.

Aerosol Determinations, Standard Plate Count

Upwind estimates of the background density of airborne bacteria-bearing particles ranged from 23 to 403/m^3 with a mean of 110.8/m^3.

Mean downwind estimates obtained at the near and intermediate distances, i.e., 21-50 m, were significantly higher than corresponding upwind values in all runs.

At the 21-30 m distance, total aerobic bacteria-bearing particles, as measured by Andersen samplers, varied from 46-1582/m^3 above background (Table 1). The mean value for all runs at this sampling distance was 485/m^3 above background. At the 41-50 m distance, values ranged from 0-1429/m^3 above background with a mean for all runs of 417/m^3 (85% of 21-30 m mean). At the 200 m distance, observations were not always above background levels, and the highest value observed was 223/m^3 above background. The mean observation above background for all samples at this distance was 37/m^3 (7.6% of 21-30 m mean). There was a consistent reduction in aerosol strength with elevation, with bacterial levels at the 9 m height being only slightly above background counts.

A notable feature of the Andersen sampler data is the great variability observed between "replicate" samplers located in the same row and operated simultaneously. Two- to three-fold variation was common. The overall standard deviation within rows of samplers in the same run was 178, yielding a coefficient of variation (= standard deviation ÷ mean) of 0.37. In contrast,

Table 1. Bacterial Aerosol Levels (Andersen Sampler): Standard Plate Count Colony Forming Particles (cfp)/m^3 at the Indicated Sampler Locations, and Net Aerosol Levels (= Observed Minus Simultaneous Mean Upwind Value)

Run	First Row Means		Second Row Means		200 m				Elevated			
					Obs.			Net	4.6 m		9 m	
	Obs.	Net	Obs.	Net	First	Second	Mean	Mean	Obs.	Net	Obs.	Net
1	534	481	477	424	126	71	99	46				
2	393	359	532	498	118	246	182	148				
3	446	403	485	442	83	95	89	46				
4	373	286	196	109	99	85	92	6				
5	979	820	658	499	62	129	96	-63				
7	1,362	1,228	1,292	1,158					463	329	163	29
8	477	387	483	393					201	111	--	--
10	1,049	980	832	763					224	155	92	23
11	285	242	274	231					133	90	117	74
12	234	157	267	190					170	93	76	-1
13	239	147	255	163					134	42	92	0
14	192	169	164	141					81	58	19	-4
15	169	104	121	56	69	80	75	9				
16	938	822	880	764	295	274	285	169				
17	967	564	800	397	311	323	317	-86				
19	584	395	520	331	362	412	387	198				
20	420	335	481	396	48	51	50	-35				
21	492	337	280	125	124	209	167	12				
22	1,314	1,202	1,128	1,016	249	260	255	143				
24	535	458	380	311	180	78	129	52				
25	585	299	443	157	108	210	159	-127				
Mean	596	485	519	408			170	37	201	125	93	20

the same samplers operated simultaneously in the laboratory aerosol chamber produced agreement within Poisson expectations.

Estimates of median particle diameter and the percentage of particles with diameters in the range 1 to 5 µm are presented in Table 2. In addition, upper and lower quartile values are presented for each distance or elevation at which Andersen samplers were deployed. Approximately 66 to 78 percent of bacteria-bearing particles were between 1 and 5 µm in diameter, the approximate range of efficient pulmonary deposition.

The data clearly indicate that the background bacterial aerosol was larger in particle size than the aerosol arising from applied wastewater. In samples from locations of low wastewater aerosol concentration, i.e., 200 m and the elevated samplers (Table 1), particle size characteristics were closer to wastewater aerosol values, as estimated by first and second row samplers, than to background (upwind) values. Thus, for example, the percent particles between 1 and 5 µm was 67 at 200 m downwind. The value for close-in samples was 78

Table 2. Median and Quartile Particle Diameters and Percentage of Particles 1 to 5 µm in Diameter, Compared to Mean Net Aerosol Concentration

	Mean Net Aerosol Conc. cfu/m^3	Quartiles			% 1-5 µm
		Lower	Middle	Upper	
Runs 1-5, 15-25					
1st Row	490	1.58	2.46	4.12	78
2nd Row	395	1.55	2.44	4.15	76
200 m	37	1.60	3.01	5.50	67
Upwind	0	2.03	4.15	7.32	48
Runs 7-14					
1st Row	473	1.82	2.82	4.40	75
2nd Row	434	1.88	2.87	4.74	73
4.6 m	125	1.92	3.00	5.10	68
9.0 m	20	1.70	3.03	5.30	66
Upwind	0	1.97	4.59	7.50	49

percent and for upwind samples, 48 percent. Thus, downwind aerosols that were close to background values in numbers of particles/m^3, nevertheless displayed particle size properties of the wastewater-derived aerosol. This indicates that these aerosols were largely of wastewater origin.

Fluorescent Dye Studies

Sodium fluorescein was used as a tracer dye in four runs. Dye levels at the spray nozzle were fairly constant for all runs, with a mean of 4.3×10^3 ng/ml. Wide variability was observed in aerosol samples between individual samplers, with AGI samplers showing greater variability than Litton samplers. Dye aerosol levels at the first and second row stations were generally in the range 100-200 ng/m^3, with expected decreases occurring both downwind and with sampler elevation.

In Table 3, fluorescein aerosol levels have been normalized to source strength. The hybrid parameter used (ng dye/m^3 air x 1000)/(ng dye/ml effluent) allows comparison with the corresponding ratio for microbiological aerosols (net colony-forming particles/m^3 air x 1000)/(colony-forming particles/ml effluent). The results for the two aerosols (dye and bacteria) are not strictly comparable, since the two were not measured in the same runs. For fluorescein the geometric means of the ratios for first and second row samples were 23.4 to 51.7. The corresponding ratios for bacteria were lower by a factor of 2 to 3, i.e., 9.8 to 15.1. The difference serves as an estimate of biological decay (dieoff) of bacterial aerosols.

DISCUSSION AND CONCLUSIONS

The aerosol field data, averaged for all samplers and all runs, define best the bacterial or dye aerosol level at close-in points, i.e., 21 to 50 m downwind from the nearest (downwind) sprinklers, because variability is evened out by replicate samples. Aerosol estimates at 200 m add an additional point on the curve of aerosol strength vs. distance.

A steep decline in bacterial aerosol density was observed upon elevation of samplers to 4.6 m or 9 m above the ground (Table 1). Values at 4.6 and 9 m elevations averaged 30% and 5% respectively of those at 1.7 m. The aerosol cloud was thus at a height readily respirable by man. One explanation of this result is that the spray field exerted a cooling and, therefore, atmospheric-stabilizing effect, thus reducing upward movement of air masses.

Approximately 75% of the bacteria-bearing aerosol particles at 30 m downwind were within the range of pulmonary deposition (1-5 μm). Pathogens, if present, would have ready access to the human respiratory tract. The particle size properties of downwind bacterial colony-forming particles were quite different from those of the background air (Table 2). Such differences may serve as indicators of the source of aerosol components. Thus, the failure of 200 m and elevated station aerosols to exhibit background particle size characteristics despite near-background aerosol counts, suggests a lower background aerosol component than that indicated by upwind sampler data.

It is suggested that for spray systems such as this one, covering a

Table 3. Mean Fluorescein Aerosol Levels Normalized with Respect to Concentration at the Source, (ng/m^3 Air x 1000)/(ng/ml Effluent)

Run	Sample Positions						
	First Row		Second Row		200 m	Elevated 4.6 m	Elevated 9 m
	AGI	Litton	AGI	Litton	AGI	AGI	AGI
6	17.8	53.3	124.0	60.6	27.9	--	--
9	59.3	62.9	69.8	58.1	--	19.5	26.3
18	13.0	45.2	20.8	--	6.9	--	--
23	21.7	35.2	6.7	39.3	0	--	--
Mean	28.0	49.2	55.3	52.7	17.4	19.5	26.3
Geom. Mean	23.4	48.1	33.1	51.7	13.9	19.5	26.1

large area, a significant washout of ambient airborne bacteria takes place resulting in a greatly reduced background microbiological aerosol downwind. The downwind aerosol, even though comparable in overall density to the upwind aerosol, would then be largely wastewater-derived and would be expected to include enteric pathogens and sewage indicator organisms in proportions characteristic of wastewater aerosols.

At 20-50 m downwind from the spray field, aerosols of seeded fluorescein dye were 2-3 times greater than bacterial aerosols in relation to source strength. With all other factors being equal these differences are estimates of biological decay (dieoff).

The total bacterial aerosol concentrations (standard plate count) encountered at the Deer Creek land waste treatment site were quite comparable to those observed at Ft. Huachuca, in earlier studies (4-6). However, in the present study, standard plate count values at the nozzle prior to aerosolization were significantly lower, 3×10^4/ml as compared to 4×10^5/ml. Compensating factors in this study may have been the presence of a field of many sprinklers, rather than one, and the generally higher relative humidity levels experienced.

Coliforms were a small part, less than 1.0 percent of aerosol populations, presenting scarcely measurable increases above background levels. For the treated wastewater at this site, coliforms would not be suitable indicators of the presence of pathogens in the aerosol state. The Klebsiella group comprised approximately 0.1 percent of wastewater bacteria. Thus, a mean aerosol level of 485 cfu/m^3 at 30 m downwind suggests a Klebsiella aerosol strength of 0.5 cfu/m^3.

Further evaluation of the data from the Deer Creek study, based on predictive mathematical modeling of aerosol behavior, is forthcoming.

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HEALTH CONSIDERATIONS

EFFECTS OF MUNICIPAL WASTEWATER IRRIGATION ON SELECT SPECIES OF MAMMALS

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Population densities and concentrations of heavy metals in tissues of small mammals on effluent irrigated and nonirrigated sites were investigated. Populations of white-footed mice (Peromyscus leucopus) were significantly higher on irrigated versus nonirrigated sites during fall but not spring. Numbers of cottontail rabbits (Sylvilagus floridanus) trapped were too small to formulate conclusions concerning the effects of wastewater irrigation on their populations.

Analyses of heavy metals in liver and kidney tissues from white-footed mice suggested that Pb and Cd were accumulating in mice inhabiting the irrigated forested sites. Only concentrations of Cu in kidneys were significantly higher in cottontail rabbits for effluent irrigated versus control sites. Heavy metals had not reached hazardous levels in herbivorous small mammals inhabiting areas irrigated with sewage effluent.

INTRODUCTION

Research at The Pennsylvania State University Wastewater Facility has demonstrated the feasibility of terrestrial wastewater renovation. The major benefits of this practice are: (1) cessation of stream and lake eutrophication, (2) recharging the groundwater, and (3) the "fertilization effect" of high nutrient levels in discharged wastewater. Past research has emphasized the influence of wastewater renovation on soil characteristics, groundwater geology and hydrology and plant ecology (Sopper and Kardos 1973, Parizek 1974).

The effects of wastewater irrigation on wildlife and wildlife habitat have not been thoroughly researched. Modification of terrestrial ecosystems by wastewater renovation may have significant effects on the health status of wildlife. If these effects are detrimental, mortality of wildlife may be increased. Beneficial effects on wildlife may exist also, and these effects should be documented, particularly if wastewater renovation can be used to enhance wildlife populations (Anthony 1976).

A more recent and justified concern with terrestrial wastewater renovation is the introduction of heavy metals into local ecosystems. The presence of heavy metals in municipal effluent has been documented by Konrad and Kleinert (1974) and Mytelka et al. (1973). Increases in Cu and Zn concentrations and total uptake in reed canarygrass (Phalaris arundinacea) have been documented (Sidle et al. 1976), but concentrations of Cu, Zn, Cd, or Pb were not high enough to be toxic to plants. Sidle and Sopper (1976) have shown differences in Cd concentrations between different vegetative species in sprayed and control areas. These findings substantiate the potential hazards to wildlife regarding heavy metal accumulation of wastewater irrigated areas.

This paper reports on a study designed to compare population densities and concentrations of heavy metals in tissues of small mammals from irrigated and nonirrigated areas. The research focused on two species of small mammals:

the cottontail rabbit (Sylvilagus floridanus) and white-footed mouse (Peromyscus leucopus).

METHODS AND MATERIALS

One rabbit study area and two small mammal study areas were selected on the Pennsylvania State University wastewater renovation facility in Centre County, Pennsylvania. The rabbit study area was located in an aspen-white pine-shrub habitat and consisted of four 1.6 ha enclosures. Two enclosures received 5 cm of effluent per week from May 1971 to November 1974, and the remaining two enclosures (controls) received no effluent spray.

The small mammal study areas were located in forested habitats. The irrigated site was a red pine (Pinus resinosa) plantation which suffered almost a 100 percent blow-down in 1968 and is now dominated by pokeweed (Phytolacca americana) and black raspberry (Rubus occidentalis). The area was irrigated with 5 cm of effluent per week from 1963 to 1970 and 7.5 cm per week from 1970 to present. The nonirrigated site (control) was a mixed oak (Quercus spp.) forest community.

Population Studies

Small mammals were live trapped, marked, and released on control and effluent irrigated sites during September 1972 and 1973, and May 1973. Sherman live traps were set at a density of 34 traps per ha with a grid interval of 20 m. An effective trapping area of .24 ha was calculated per trap using 27.8 m as the mean distance that Peromyscus spp. traveled between recaptures. All traps were checked once a day, and trap periods varied from 7 to 14 days depending on trap success. Population estimates were calculated using capture-recapture techniques and the modified Schnabel estimator (Overton 1965).

Trapping of cottontail rabbits was conducted in November and December of 1971, 1972, and 1973 on control and effluent irrigated sites. Traps were set at a density of 25 traps per ha with a grid interval of 20 m, and population estimates were determined by total capture. Conibear traps were placed in holes and runways to remove trap-shy animals and ensure total capture.

Heavy Metal Accumulation

White-footed mice and cottontail rabbits were collected from control and irrigated forested areas. Liver, kidney, and bone (femur) from cottontail rabbits and liver and kidney from white-footed mice were frozen until samples could be processed. Tissue samples were wet ashed in concentrated nitric and perchloric acid. Concentrations of Cu, Zn, Cr, Pb, Co, and Ni were analyzed by atomic absorption methods using the Perkin-Elmer 306 atomic absorption spectrophotometer. For Cd, an atomic abosrption spectrophotometer equipped with a model 2200 Heated Graphite Atomizer was used for the determinations.

RESULTS AND DISCUSSION

Small Mammal Populations

The population density of white-footed mice on an area irrigated with sludge-injected effluent was not significantly different ($P>0.05$) from the control area in fall of 1972 (Table 1). Population densities on two areas irrigated with effluent and sludge-injected effluent were significantly ($P<0.05$) higher than densities on two adjacent control sites in fall 1973. All spring population densities on both irrigated sites were not significantly different ($P>0.05$) from densities on adjacent control sites. Increase in fall populations was most likely in response to vegetation changes which occur in a mature deciduous forest. The lush undergrowth of herbaceous plants in summer and early fall provides increased food and cover for mice. This vegetation virtually disappears in winter resulting in "normal" populations in the spring.

Other mammalian species captured in smaller numbers on both areas were the short-tail shrew (Blarina brevicauda), meadow vole (Microtus pennsylvanicus), southern flying squirrel (Glaucomys volans), opossum (Didelphis marsupialis), and longtail weasel (Mustela frenata).

Fall populations of cottontail rabbits were slightly higher, but not significantly so, on irrigated plots as compared to nonirrigated plots (Table 2). Numbers of rabbits trapped and size of enclosures were too small to make definitive conclusions concerning the effects of wastewater irrigation on cottontail rabbits. Casual observations indicated that winter ice accumulation

Table 1. Estimates of small mammal (*Peromyscus* spp.) populations on irrigated, nonirrigated, and sludge-injected areas.

Year	Season	Irrigated area (no./ha)	95% CI[a] Lower limit-Upper limit	Nonirrigated area (no./ha)	95% CI[a] Lower limit-Upper limit	Z Value[b]
			Area I: Effluent Treatment			
1972	Fall	8.98	5.77 - 13.59	7.65	4.56 - 12.40	.3669
1973	Spring	2.43	1.68 - 3.51	1.98	1.25 - 3.01	.2546
1973	Fall	9.06*	7.14 - 11.30	4.61*	2.94 - 6.82	.0022
			Area II: Sludge-injected Effluent Treatment			
1973	Spring	6.00	3.79 - 9.11	6.22	2.45 - 13.49	.3897
1973	Fall	16.75*	12.66 - 21.76	6.93*	4.16 - 10.71	.0030

[a]CI is the confidence interval according to Chapman and Overton (1966).

[b]Z value from Chapman and Overton (1966). Indicates the probability that the two means are from the same population.

*Denotes a significant difference at the .05 level.

Table 2. Numbers of cottontail rabbits in irrigated and nonirrigated enclosures during fall of 1971-1973.

Year	Irrigated			Nonirrigated		
	No. of Adults	No. of juveniles	No. of total	No. of Adults	No. of juveniles	No. of total
1971	5	8	13	4	6	10
1972	8	5	13	8	3	11
1973	10	5	15	8	2	10

lowered and broke shrubs and saplings resulting in more food and better winter cover for cottontail rabbits. These vegetative changes plus increase in forage quality (Wood et al. 1973) suggest a potential benefit to cottontail rabbit populations. These relationships between rabbit populations and vegetative changes due to wastewater irrigation should be investigated in future studies.

Heavy Metal Accumulation

Concentrations of Pb in liver and kidney and Cd in kidney of white-footed mice from the irrigated forest area were significantly ($P<0.01$) higher than those from mice from the nonirrigated area (Table 3). Conversely, concentrations of Zn and Ni were significantly ($P>0.01$) higher in liver and kidney of mice from the nonirrigated site. There were no significant ($P>0.01$) differences in Cu, Cr, and Co in liver and kidney between the two forested areas.

Analyses of heavy metals in liver and kidney tissues from white-footed mice suggested that Pb and Cd were accumulating in mice inhabiting the treated forested areas. However, concentrations of Pb and Cd had not reached levels that are considered to be toxic to animals. Concentrations of Zn, Ni, Cu, Cr, and Co showed that these metals were not accumulating in mice on the sprayed forested area. Although concentrations of Pb and Cd had not reached toxic levels in the present study, our data demonstrate the potential hazards of these metals with other sewage irrigation systems particularly from industrial areas where metal concentrations in effluent are higher.

Only concentrations of Cu in kidneys of cottontail rabbits were significantly higher ($P<0.05$) for effluent irrigated than control sites (Table 4). Conversely, Cu in bone, Cd in kidneys and liver, and Ni and Pb in kidneys were significantly higher ($P<0.05$) in rabbits from nonirrigated sites. No significant differences in concentrations of Cr and Zn in rabbit tissues were found between irrigated and nonirrigated sites.

Analysis of heavy metals in liver, kidney, and bone of cottontail rabbits indicated that only Cu was accumulating in rabbits inhabiting wastewater irrigated sites. Results showed that Cd, Cr, Ni, Pb, and Zn were not accumulating in rabbits on irrigated sites.

Sidle and Sopper (1976) documented the presence of heavy metals in liquid and sludge-injected effluent from the Pennsylvania State University Wastewater Renovation Facility. Their data characterize heavy metal concentrations commonly encountered from municipal sewage effluent in nonindustrial areas. Sidle et al. (1976) have investigated heavy metal uptake by plants in wastewater irrigated areas. Results of this study indicate that heavy metals were not accumulating to hazardous levels in small mammals inhabiting wastewater irrigated sites at the Pennsylvania State University Facility. Higher concentrations of Pb and Cd in white-footed mice on irrigated sites do, however, suggest some potential hazards.

Plant uptake of heavy metals is highly dependent on soil pH, organic matter, competing cations, and plant species (Haghiri 1974), as well as the amount of heavy metals applied in the wastewater. All these factors could

Table 3. Concentrations (PPM) of heavy metals in liver and kidney of white-footed mice (*Peromyscus leucopus*) from effluent irrigated and nonirrigated sites. Values are mean ± standard deviation.

	Liver		Kidney	
Element	Irrigated site	Nonirrigated site	Irrigated site	Nonirrigated site
Cu	9.73 ± 3.68	11.87 ± 6.80	7.20 ± 1.45	7.67 ± 1.37
Zn	34.17 ± 9.57**	50.26 ± 20.10**	23.65 ± 4.86**	43.61 ± 28.50**
Cr	2.22 ± .90	2.15 ± 1.57	7.99 ± 10.30	8.04 ± 6.03
Pb	2.00 ± 1.49**	.89 ± .46**	7.11 ± 2.87**	3.71 ± 2.88**
Co	.67 ± .23	.75 ± .43	1.29 ± .88	1.44 ± .71
Cd	.15 ± .10	.10 ± .12	.66 ± .46**	.18 ± .18
Ni	.81 ± .30**	1.35 ± .79**	2.85 ± 1.32**	4.43 ± 2.01**

**Denotes a statistically significant difference between means of irrigated and nonirrigated sites at the .01 level of significance.

Table 4. Heavy metal levels in cottontail rabbit tissues from animals captured from effluent irrigated and nonirrigated enclosures. Values are mean ± standard error of mean (s/n).

		Metal (PPM)					
Tissue	Treatment	Cd	Cr	Cu	Ni	Pb	Zn
Liver	Irrigated	1.209 ± .12*	.878 ± .05	3.602 ± .37	2.501 ± .22	1.157 ± .09	45.904 ± 5.4
	Nonirrigated	1.668 ± .31*	.855 ± .06	3.701 ± .56	2.485 ± .27	1.084 ± .06	40.945 ± 6.4
Kidney	Irrigated	6.719 ± .81*	.483 ± .04	3.687 ± .11*	3.126 ± .30*	1.010 ± .12*	35.664 ± 2.4
	Nonirrigated	9.411 ± .92*	.452 ± .03	3.353 ± .15*	6.567 ± 2.54	1.909 ± .55*	33.417 ± 1.9
Bone	Irrigated	7.828 ± .46	5.301 ± .29	3.439 ± .27	85.853 ± 5.1	50.778 ± 2.7	200.611 ± 13.5
	Nonirrigated	7.918 ± .56	5.521 ± .25	5.095 ± .42*	77.140 ± 4.4	50.243 ± 2.6	179.141 ± 16.4

*Denotes a statistically significant difference between means of irrigated and nonirrigated sites at the .05 level of significance.

explain the lower concentrations of heavy metals in small mammals' tissues from the treated areas as compared to the control areas. Contrasting results for white-footed mice versus cottontail rabbits were probably a result of the different areas they inhabited and corresponding soil-plant relationships in terms of heavy metal uptake. In addition, white-footed mice select different food items than cottontail rabbits; the former are primarily granivorous while the latter are herbivorous. Sidle and Sopper (1976) have documented differences in Cd concentrations between different plant species which strongly suggests that mammals with different food habits would accumulate heavy metals at different rates.

Studies on the effect of wastewater renovation on wildlife and wildlife habitat have been limited to only a few vegetative types and have been hindered by small plot size and short duration of investigations. As larger irrigation systems become operable opportunities to investigate the definitive relationships between habitat changes and wildlife responses will arise. Integral to these studies are investigations on the enhancement, prevalence, and pathogenicity of diseases in wildlife which may present a human health hazard (Anthony 1976).

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HEALTH CONSIDERATIONS

ACCUMULATION OF TRACE ELEMENTS IN SOIL AND PLANTS FROM LAND DISPOSAL OF SECONDARY DOMESTIC WASTEWATER

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ABSTRACT

The accumulation of trace elements in soil and plants from land disposal of secondary treated wastewater has been examined at two locations: Las Virgenes and Orange County Wastewater Treatment Plants in Southern California. Study sites were selected on the basis of past history of land disposal of wastewater, rate of application and manner of applications. Soil, plant and wastewater effluent samples were collected 6 times during the first year of the study. The soil was analyzed for particle size distribution, cation exchange capacity, soil moisture and metal content. A total of 12 elements have been studied. This paper reports on nickel, cadmium, silver and lead. Metal determinations were done using atomic absorption spectometry. The studied indicated that at Las Virgenes cadmium was the only metal which was higher in plant material collected from sites receiving wastewater, while no difference was found in trees studied at Orange County. Silver, cadmium and lead were higher in the surface layer of soil receiving wastewater effluent at Orange County, while the distribution of metals at the Las Virgenes test sites receiving wastewater was not restricted to the surface layer.

INTRODUCTION

The application of treated wastewater to land may serve several beneficial purposes. The soil may act as a filtering system, removing nitrogen, phosphorus, and biochemical oxygen demand (Kardos, et al., 1974; Bouwer, 1973; Kardos, 1970). The use of treated wastewater for irrigation purposes can extend water resources by augmenting current supplies to meet present and future demands. However, concern has been expressed due to the possibility of environmental contamination from prolonged wastewater application to land. The effect on soil and plants from the application of sludges to cropland has been well documented (EPA Report 43019-76-013). The possible effects from the use of wastewater have more recently been studied (Sidle and Sopper, 1976; Sidle et al., 1976; Sidle et al., 1977). Therefore this study was undertaken to determine if nickel, lead, cadmium and silver accumulated in soil and plants from the long-term application of chlorinated secondary domestic sewage effluent.

METHODS

Sampling Locations

The accumulation of trace elements in soil and plants from the land disposal of secondary effluent has been examined at two locations, Las Virgenes Wastewater Treatment Plant and Orange County Wastewater Treatment Plant, in Southern California. Sites were selected on the basis of past history of land disposal of wastewater, rate of wastewater application and manner of application. Three sampling sites were selected at Las Virgenes, two had received secondary chlorinated effluent by spray irrigation from March through November of each year for a period of 4 years previous to the initiation of the study. One site had received no wastewater until March, 1977. In Orange County secondary effluent had been applied by surface flooding to the disposal site, a forested area, for a period of twelve years, and the control site had never received wastewater.

Soil Type

At Las Virgenes the soil at the test site was characterized as sorrento loam. The surface layer (0-67.5 cm) is dark greyish brown with a pH range of 6.1 - 6.5. The subsurface layer (67.6-150 cm) is brown silt loam. The soil at the control site is slightly more basic with the surface pH range of 6.6 - 7.3 and the subsurface pH range of 7.4 - 8.4. The soil at the Orange County study sites is characterized as hueneme fine sandy loam. The surface layer is light brown-grey. The subsurface layer is stratified into grey loamy sand, silty loam, loamy fine sand, fine sandy loam and silty clay loam.

Sample Collection

Six soil sampling holes were established at 12m intervals within each study site. Soil samples were collected at 15cm intervals from the surface to a depth of 120cm. A total of six samples were collected at Las Virgenes and Orange County from August 1976 through July 1977. Plant and soil samples were collected at the same time. Mature healthy plants were selected and whenever possible plants from different test sites were matched to Genus and species.

Metal Analysis

Soil samples were air-dried, pulverized and passed through a 2mm sieve. Two grams of air-dried soil were placed in a 25mm x 200mm test tube and 12.5 ml of 4 N HNO_3 was added. The samples in duplicate were then digested (70 C - 80 C) for 14 to 18 hours. The volume of the samples was adjusted to 50ml with deionized water and the samples were shaken at 300 rev/min for 15-20 minutes. The digest was then filtered through Whatman 42 ashless filter paper, and the filtrate collected and analyzed on a 503 Perkin-Elmer Atomic Absorption Spectrometer using flame ionization, except for Pb, which was determined using an electrodeless discharge lamp. A series of standards for each metal (250ppb, 100ppb, and 50ppb) and a reagent blank were run with each set of digestions.

Plant Analysis

Plant samples were separated and washed. All parts of the plant were washed in deionized water except the root material, which was washed in a 10% HCl acid solution followed by rinsing in deionized water. The plant material was then air-dried for 48 hours and oven dried at 50°C for an additional 48 hours. The digestion procedure referred to above was followed except that 1 gram of plant material and concentrated nitric acid was used. If the plant material was not fully digested after heating, several drops of hydrogen peroxide were added. Plant filtrate was also analyzed using atomic absorption spectrometry. All metals were determined using flame ionization, except for Pb and Ni, which were determined using a graphite furnace.

Water Sample Analysis

Secondary effluent and irrigation water from the test locations was collected at each sampling time. The water samples were immediately acidified to a pH$<$1, using concentrated HNO_3. Water samples in duplicate were digested using the procedure for soil except 9.4ml of water and 3.1ml of concentrated HNO_3 were added to the

TABLE 1

Wastewater Application Rates

Site	Amount applied (acre-ft/acre-yr)
Las Virgenes Control[1]	0
Las Virgenes Control[2]	15
Las Virgenes Low Rate	4.5
Las Virgenes Moderate Rate	8-10
Orange County Disposal	0.2-1
Orange County Control	0.8-1

[1]August 1976 - February 1977

[2]After February 1977

test tubes. After digestion, the water samples were not diluted or filtered. All water samples were analyzed for metal content using atomic absorption spectrometry.

Soil Characteristics

Cation exchange capacity (CEC) and pH were determined using standard procedures (Chapman and Pratt, 1961). The percent of water contained in the soil sample was determined using the method outlined in Standard Methods for the Analysis of Water and Wastewater.

Particle size analysis was determined using a modification of the pipet method. All the soil which was less than 53 m was analyzed using a Hiac particle size analyzer Model 2000.

RESULTS

Wastewater

Wastewater application rates for the two study locations are shown in Table 1.

The mean yearly concentration of the trace elements in wastewater effluent and irrigation water are shown in Table 2.

The metal concentrations in secondary effluents were lower than in the irrigation water, indicating that metal content was increased from transport through irrigation pipes.

General Soil Characteristics

Variation is seen in the particle size distribution at all test locations. An analysis of particle size data throughout the soil column showed variation in particle size distribution among sampling holes at all test locations. The largest differences in particle distribution were among those particles <51 μm. At the Las Virgenes moderate rate disposal site, 20% more particles were in this size range compared to the control site.

The cation exchange capacity of the control sites for both locations was significantly greater than those receiving wastewater. The particle size distribution for the test sites at Las Virgenes were varied between 3.8-7.8% sand, 38-52.5% silt, and 20.4-28.5% clay. At the Orange County site the sand content varied between 5.6-8.3%, silt 42.7-46.7% and clay 15.9-25.6%.

The moisture content of the soil at the Orange County control site was greater than the disposal site throughout the year. At Las Virgenes the effect of moderate-rate spray irrigation on the moisture content of the soil was clearly evident. The amount of water contained in the soil column was similar for the control site (prior to March 1977) and the low rate application field. After the initiation of wastewater application to the control site, a large increase (7-10%) in soil moisture occurred.

Metal Content in Soils

Though the range of nickel content was greater at the Orange County disposal site, the mean concentration for all samples taken below 15cm was less than that found at the control site (Table 3). The pattern of Ni distribution in the soil at Las Virgenes was somewhat more complex. There was wide variation in Ni content for all the locations and depths. Generally, the mean concentration of nickel (over the year study) was lowest at the low-rate application field, while below 45cm the concentration of Ni was approximately equal for the control and moderate application rate sites.

Silver levels (Table 4) were elevated in the 0-15cm section of soil at the Orange County disposal site,

TABLE 2

Concentration of Trace Elements in Secondary Effluent and Irrigation Water

Site	Metal mg/1							
	Pb		Ni		Ag		Cd	
Las Virgenes Secondary Effluent	0.007	0.002	0.029	0.019	0.002	0.0003	0.002	0.001
Las Virgenes Irrigation	0.020	0.004	0.044	0.003	0.004	0.001	0.011	0.013
Orange County Secondary Effluent	0.040	0.013	0.089	0.046	0.006	0.001	0.029	0.006
Orange County Irrigation	0.072	0.027	0.081	0.044	0.008	0.001	0.037	0.013
Orange County Control Irrigation	0.001	0.003	0.040	0.010	0.002	0.001	0.001	0.001

TABLE 3

Nickel Content (μg/g dry weight) of Test Soils

Soil Depth (cm)	SITE				
	Orange County Control	Orange County Disposal	Las Virgenes Control	Las Virgenes Low Rate	Las Virgenes Moderate Rate
0-15					
range	12.50 - 22.68	10.38 - 31.25	14.50 - 53.50	25.40 - 35.87	22.50 - 45.40
$\bar{X}$	16.97 ± 2.58	14.91 ± 3.63	43.18 ± 9.55	30.08 ± 2.23	30.22 ± 4.32
16-30					
range	10.75 - 19.80	8.13 - 25.25	15.30 - 57.50	25.00 - 40.50	22.15 - 59.60
$\bar{X}$	16.56 ± 2.74	11.60 ± 2.35	40.99 ± 8.90	30.25 ± 1.21	34.18 ± 8.75
31-45					
range	14.13 - 25.00	8.50 - 18.38	15.00 - 56.25	22.50 - 30.26	21.70 - 56.80
$\bar{X}$	17.01 ± 2.92	10.66 ± 1.46	42.32 ± 10.34	27.86 ± 3.05	34.57 ± 6.86
46-60					
range	12.50 - 21.00	8.30 - 22.00	13.30 - 4.50	22.50 - 32.18	19.90 - 67.36
$\bar{X}$	16.86 ± 2.37	9.72 ± 1.13	37.24 ± 5.06	27.86 ± 3.05	37.85 ± 13.37
61-75					
range	13.80 - 19.80	5.10 - 19.00	14.00 - 43.50	21.30 - 34.50	18.95 - 68.40
$\bar{X}$	16.71 ± 1.93	8.69 ± 3.30	36.17 ± 5.52	26.46 ± 3.62	35.34 ± 11.90
76-90					
range	11.30 - 20.90	5.30 - 35.00	13.00 - 40.50	19.80 - 41.50	24.60 - 62.80
$\bar{X}$	16.55 ± 3.22	11.08 ± 4.15	37.18 ± 3.55	25.84 ± 3.05	36.30 ± 11.22
91-105					
range	10.00 - 20.90	4.50 - 15.95	13.00 - 46.30	18.30 - 33.50	27.70 - 39.80
$\bar{X}$	16.52 ± 0.91	9.81 ± 3.08	36.76 ± 3.43	24.96 ± 2.56	32.08 ± 3.70
106-120					
range	14.30 - 19.90	9.30 - 30.05	13.50 - 65.40	16.75 - 27.30	24.30 - 43.15
$\bar{X}$	17.20 ± 1.57	13.90 ± 5.49	46.29 ± 17.62	21.69 ± 2.78	33.51 ± 6.37

$\bar{X}$ = Yearly mean

TABLE 4

Silver Content (μg/g dry weight) of Test Soils

Soil Depth (cm)	SITE: Orange County Control	Orange County Disposal	Las Virgenes Control	Las Virgenes Low Rate	Las Virgenes Moderate Rate
0-15					
range	0.83 - 1.50	1.32 - 4.28	1.10 - 2.50	0.38 - 1.08	0.38 - 1.08
$\bar{X}$	1.07 ± 0.17	2.40 ± 1.08	1.44 ± 0.28	1.36 ± 0.54	0.82 ± 0.19
16-30					
range	0.17 - 1.31	0.66 - 1.43	0.85 - 2.05	0.44 - 1.03	0.44 - 1.03
$\bar{X}$	1.05 ± 0.15	1.01 ± 0.28	1.32 ± 0.33	1.38 ± 1.30	0.85 ± 0.16
31-45					
range	0.80 - 1.40	0.82 - 1.08	0.82 - 2.55	0.70 - 1.07	0.70 - 1.07
$\bar{X}$	1.02 ± 0.13	0.96 ± 0.10	1.38 ± 0.44	0.80 ± 0.23	0.88 ± 0.15
46-60					
range	0.85 - 1.35	0.68 - 1.14	0.73 - 2.63	0.43 - 1.13	0.43 - 1.13
$\bar{X}$	1.10 ± 0.09	0.86 ± 0.09	1.31 ± 0.43	0.72 ± 0.19	0.95 ± 0.25
61-75					
range	0.78 - 1.30	0.40 - 2.04	0.68 - 2.22	0.45 - 1.35	0.45 - 1.35
$\bar{X}$	1.06 ± 0.14	0.75 ± 0.31	1.27 ± 0.40	0.69 ± 0.19	0.93 ± 0.28
76-90					
range	0.83 - 1.45	0.55 - 1.23	0.79 - 2.24	0.43 - 0.87	0.45 - 1.35
$\bar{X}$	1.01 ± 0.13	0.76 ± 0.25	1.29 ± 0.37	0.67 ± 0.17	0.82 ± 0.24
91-105					
range	0.63 - 1.83	0.54 - 0.99	1.13 - 2.25	0.53 - 1.28	0.49 - 1.10
$\bar{X}$	1.02 ± 0.21	0.74 ± 0.17	1.50 ± 0.16	0.75 ± 0.12	0.88 ± 0.23
106-120					
range	0.70 - 1.24	0.44 - 1.53	1.20 - 2.15	0.46 - 1.08	0.45 - 1.12
$\bar{X}$	1.02 ± 0.13	0.80 ± 0.19	1.51 ± 0.05	0.74 ± 0.20	0.85 ± 0.26

$\bar{X}$ = yearly mean

while below 15cm the concentration of silver was similar to or lower than that of the control site. At Las Virgenes test sites Ag concentration (Table 4) throughout the soil column was greatest at the control site.

Lead concentration in the soil column at Orange County is shown in Table 7. The surface layer of the soil was higher in lead content at the disposal site than the control field. Overall, however, the mean Pb concentration below 15cm was greater at the control site.

The distribution of lead in the Las Virgenes test fields presented a different pattern (Table 5). Here the low-rate application field had the highest content of surface lead. The location of this field next to a highway is the probable explanation. Below 45cm the amount of lead in the control and low-application rate fields was very similar. The data indicated that the moderate-rate application site had a higher concentration of lead than the other test sites at Las Virgenes.

The distribution pattern of Cd at both Orange County and Las Virgenes is shown in Table 6. At the Orange County medium-rate application site the Cd concentration in the surface layer of the soil was higher than that at the control field. The Cd levels at lower depths were similar for these two sites. Cadmium levels are naturally elevated in soil at Las Virgenes. The control site had mean Cd levels of approximately 3 mg/kg. The low-rate application field had a somewhat higher mean level of Cd throughout the soil column tested, and the moderate-rate application field had still higher concentrations. In order to determine if these Cd levels were elevated due to natural background concentrations in moderate-rate soil samples, an adjacent moderate-rate disposal site and the area above the moderate-application rate field (no wastewater application except due to

TABLE 5

Lead Content (μg/g dry weight) of Test Soils

Soil Depth (cm)	SITE				
	Orange County Control	Orange County Disposal	Las Virgenes Control	Las Virgenes Low Rate	Las Virgenes Moderate Rate
0-15					
range	8.10 - 13.96	6.33 - 41.50	1.70 - 7.50	7.48 - 17.90	5.20 - 15.00
$\bar{X}$	11.63 ± 1.91	22.75 ± 10.82	6.43 ± 1.37	11.32 ± 0.96	9.96 ± 2.29
16-30					
range	6.90 - 12.90	4.00 - 11.50	1.97 - 12.60	5.59 - 12.40	5.97 - 11.30
$\bar{X}$	9.85 ± 0.92	8.21 ± 2.26	5.74 ± 1.40	9.28 ± 1.31	8.56 ± 1.74
31-45					
range	4.90 - 11.70	6.55 - 10.40	2.00 - 7.80	5.40 - 9.80	4.88 - 11.70
$\bar{X}$	8.55 ± 0.80	7.66 ± 0.56	5.86 ± 0.26	6.90 ± 0.92	7.78 ± 2.23
46-60					
range	6.20 - 13.50	3.70 - 11.80	1.40 - 15.80	3.40 - 8.50	4.78 - 11.10
$\bar{X}$	9.27 ± 1.41	7.21 ± 2.38	7.03 ± 1.70	6.03 ± 1.62	7.32 ± 1.60
61-75					
range	6.70 - 16.40	1.70 - 10.10	1.40 - 10.37	3.70 - 8.90	4.44 - 9.60
$\bar{X}$	9.50 ± 1.93	5.28 ± 3.05	6.04 ± 1.38	6.00 ± 1.50	7.62 ± 1.54
76-90					
range	6.40 - 16.00	2.40 - 14.40	1.60 - 6.20	3.00 - 7.90	4.40 - 10.70
$\bar{X}$	8.97 ± 1.87	7.13 ± 4.31	5.71 ± 2.03	5.39 ± 1.41	6.72 ± 1.80
91-105					
range	4.00 - 22.10	2.30 - 9.60	1.60 - 8.10	2.70 - 10.60	4.80 - 10.50
$\bar{X}$	10.66 ± 3.71	5.10 ± 2.22	5.59 ± 1.03	5.35 ± 2.20	6.92 ± 1.88
106-120					
range	7.00 - 17.50	3.30 - 8.90	3.20 - 8.08	2.80 - 8.00	4.50 - 11.80
$\bar{X}$	10.64 ± 2.68	5.27 ± 1.69	5.52 ± 0.34	5.20 ± 2.03	7.08 ± 1.47

$\bar{X}$ = yearly mean

aerial carryover) were also analyzed for Cd content at the 0-15cm depth. The cadmium levels were 12.46 μg/g and 5.34 μg/g respectively.

Metal Content in Plants

The metal content of the trees studied at Orange County is shown in Table 7. Of the tree species sampled only the genus *Pinus* was found at both test locations. The data in this Table indicates that somewhat higher concentrations of Pb and Ag were found in the pine trees at the disposal site. The high amount of lead in the needles at the disposal site may be due to incomplete removal of surface Pb during washing of the plants. There appeared to be little difference with regard to the metals tested between *Pinus* spp. located at the disposal site.

The trace-element content in plant species (Tables 8 and 9) collected from Las Virgenès indicated that there is little difference in metal uptake among the test sites except for Cd. The only agricultural crop tested was *M. sativa*, which was only present at the low-rate application field. A comparison between different species of this genus indicated that *M. hispida* found at the moderate-rate disposal field had a greater concentration of Cd than *M. sativa* growing in the low-rate field. Further, *M. lupulina* taken from the control site had lower Cd levels than those species growing on sites receiving wastewater. Comparisons of Cd levels in coniferous foilage from Cd polluted and nonpolluted areas have shown levels of 0.05 - 1.0 μg/g and 0.01 - 0.9 μg/g respectively (Schacklette, 1972). Tyler (1972) found that *Pinus strobus* L. had Cd levels of 0.9 μg/g in untreated soils. This study has also shown that there is little difference in Cd levels in *Pinus* spp. between the control and disposal sites. Therefore, in soils such as

TABLE 6

Cadmium Content (μg/g dry weight) of Test Soils

Soil Depth (cm)	SITE: Orange County Control	Orange County Disposal	Las Virgenes Control	Las Virgenes Low Rate	Las Virgenes Moderate Rate
0-15					
range	0.90 - 1.53	1.20 - 10.80	1.90 - 4.10	4.10 - 5.80	5.90 - 13.70
$\bar{X}$	1.16 ± 0.18	5.47 ± 2.92	3.35 ± 0.74	4.75 ± 0.37	10.44 ± 2.28
16-30					
range	0.90 - 1.30	1.00 - 1.90	1.80 - 4.10	4.10 - 6.10	6.60 - 14.00
$\bar{X}$	1.14 ± 0.17	1.13 ± 0.17	3.46 ± 0.37	4.67 ± 0.51	11.50 ± 1.65
31-45					
range	0.90 - 1.95	0.90 - 1.70	1.90 - 4.10	4.00 - 6.10	5.70 - 16.90
$\bar{X}$	1.12 ± 0.19	1.28 ± 0.28	3.46 ± 0.32	4.29 ± 0.54	11.88 ± 1.80
46-60					
range	0.90 - 1.50	0.80 - 1.40	1.80 - 4.20	3.40 - 5.00	4.60 - 15.10
$\bar{X}$	1.21 ± 0.15	1.08 ± 0.19	3.42 ± 0.33	4.23 ± 0.46	12.62 ± 2.05
61-75					
range	0.90 - 1.40	0.60 - 1.20	1.50 - 4.10	1.40 - 4.70	3.80 - 17.80
$\bar{X}$	1.13 ± 0.18	0.85 ± 0.18	3.42 ± 0.59	4.06 ± 0.38	11.81 ± 0.81
76-90					
range	0.95 - 1.60	0.70 - 1.80	1.60 - 4.40	2.90 - 4.30	8.60 - 17.80
$\bar{X}$	1.25 ± 0.19	1.01 ± 0.26	3.48 ± 0.32	3.81 ± 0.26	11.80 ± 0.85
91-105					
range	0.80 - 1.60	0.70 - 1.30	1.90 - 3.80	3.80 - 4.30	9.00 - 19.70
$\bar{X}$	1.20 ± 0.20	0.96 ± 0.04	3.26 ± 0.47	3.99 ± 0.16	12.03 ± 3.67
106-120					
range	0.95 - 1.40	0.90 - 1.50	1.80 - 4.30	3.30 - 4.20	6.60 - 16.10
$\bar{X}$	1.15 ± 0.18	1.08 ± 0.09	3.58 ± 0.44	3.70 ± 0.07	12.03 ± 3.18

$\bar{X}$ = yearly mean

that found in Orange County where accumulation occurs only in the surface layer of soil, no increase in Cd levels would be expected in plant species with root zones below 15cm. It should be noted that some of the data are based on only one collection time and therefore should be regarded as preliminary.

DISCUSSION

Previous investigators have shown that zinc and copper can accumulate in the soil from the long term application of wastewater (Sidle et al., 1977). However, these investigators could not demonstrate a definite accumulation patter for lead, nickel or cobalt. Further, Sidle and Sopper (1976) were only able to demonstrate increased Cd levels at 0-5cm depth in soil which had received 5-15cm per week wastewater over a 10 year period.

The major findings of the present investigation was that long-term application of wastewater by flooding or spray irrigation can cause an accumulation of certain trace elements in both soil and plants. Silver, cadmium and lead were shown to accumulate in the hueneme loam soil (Orange County), while no definite accumulation pattern was found in the sorrento loam (Las Virgenes) soil. There was some evidence to suggest that Pb also may accumulate in this soil. In huenema loam the accumulation of trace elements was confined to 0-15cm depth. Though the elements under investigation in this study were different from those in the Sidle et al. (1977) investigation, these workers also found accumulation in the upper layer of the soil. In sorrento loam Cd was found to be evenly distributed throughout the soil column. Plants growing at the moderate-rate application site at Las Virgenes (sorrento loam) also were found to have elevated Cd levels compared with similar species from the other test sites. The relatively high cation exchange capacity of the soil and the

TABLE 7

Trace Element Concentration in Plant Species from Orange County

Plant Species	Pb	Ni	Ag	Cd
	Metal (ug/g) dry weight			
		Orange County Disposal		
Pinus Pinea				
Needle				
range	4.6 - 21.38	0.95 - 2.23	0.11 - 0.22	1.35 - 2.5
$\bar{X}$	11.69 ± 7.18	1.54 ± 0.55	0.16 ± 0.04	1.87 ± 0.49
Stem				
range	4.95 - 6.68	0.88 - 2.55	0.16 - 0.22	1.95 - 2.05
$\bar{X}$	5.92 ± 0.87	1.69 ± 0.84	0.21 ± 0.06	2.00 ± 0.07
Cone				
range	6.55 - 10.7	2.17 - 2.56	0.15 - 0.53	1.65 - 1.80
$\bar{X}$	8.63 ± 2.93	2.37 ± 0.27	0.29 ± 0.17	1.73 ± 0.11
Pinus halpensis				
Needle				
range	5.50 - 15.96	1.20 - 1.88	0.27 - 0.70	1.41 - 2.0
$\bar{X}$	10.73 ± 7.40	1.54 ± 0.48	0.48 ± 0.30	1.71 ± 0.42
Stem				
range	0.05 - 19.5	1.08 - 1.60	0.20 - 0.90	1.65 - 3.40
$\bar{X}$	9.78 ± 13.75	1.34 ± 0.37	0.55 ± 0.50	2.53 ± 1.24
Melaleuca leucadena				
Leaf[1]	3.88	3.68	0.26	1.35
Stem				
range	6.1 - 6.9	1.28 - 3.70	0.32 - 0.35	1.4 - 1.82
$\bar{X}$	6.5 ± 0.57	2.49 ± 1.71	0.34 ± 0.02	1.61 ± 0.30
Cupressus semperivens				
Leaf				
range	19.59 - 23.45	3.70 - 5.33	0.43 - 0.75	2.10 - 2.23
$\bar{X}$	21.52 ± 2.73	4.52 ± 1.15	0.59 ±	2.17 ± 0.09
Stem				
range	1.40 - 50.95	--	0.35 - 0.36	2.55 - 2.69
$\bar{X}$	35.03 ± 26.17	3.28[1]	0.35 ±	2.62 ± 0.10
Cone[1]	1.48	1.35	0.18	1.55
Eucalyptus sp.				
Leaf				
range	28.00 - 34.70	3.97 - 4.25	0.29 - 0.60	1.25 - 1.46
$\bar{X}$	31.35 ± 4.74	4.11 ± 0.20	0.45 ± 0.22	1.36 ± 0.14
Stem				
range	4.77 - 6.78	--	<0.01 - 0.19	1.75 - 6.36
$\bar{X}$	5.78 ± 1.42	1.62[1]	<0.10 ± 0.10	4.06 ± 3.26
		Orange County Control		
Pinus sp.				
Needle				
range	2.75 - 11.65	0.20 - 6.0	0.99 - 0.44	1.00 - 1.60
$\bar{X}$	6.68 ± 4.54	2.53 ± 3.06	0.28 ± 0.13	1.27 ± 0.31
Stem[1]	21.23	3.00	0.55	2.20

-- = no sample available
1 - value based on one sampling time
$\bar{X}$ = yearly mean

low pH could explain these elevated levels. Miller et al. (1976) have shown that increased cation exchange capacity and low pH increase Cd uptake from the soil by plants. This is most likely explained in terms of increased Cd solubility.

As in the work done by Sidle et al. (1977) no accumulation of nickel was noted at either of the test locations, and the levels of nickel found in the plant materials were well below the phytotoxicity level of 50 µg/g reported by Cunningham et al. (1975).

It is important to note that difference in lead levels in the surface soil tested could be attributed to aerial fallout as well as wastewater application. One indication that this is not the case at the Las Virgenes sites is related to Pb concentrations within the depth profiles. Below 60cm the control and low-rate application fields contained similar lead concentrations. These concentrations are lower than those found at similar depths in the moderate-rate application site. Because the moderate-rate application site is removed from any roadways and Pb levels remain elevated to a depth of 120cm, wastewater application could account for this increase.

TABLE 8

Concentration of Lead and Cadmium in Plant Species from Las Virgenes Test Sites

Plant Species	Site: Las Virgenes Control		Las Virgenes Low Rate		Las Virgenes Moderate Rate	
	Metal ug/g					
	Pb	Cd	Pb	Cd	Pb	Cd
Medicago sativa						
leaf	--[1]	--	1.94 ± 1.45	1.79 ± 0.34	--	--
stem	--	--	1.43 ± 0.79	1.80 ± 0.35	--	--
root	--	--	1.33 ± 1.08	2.38 ± 0.41	--	--
Medicago lupulina[2]						
leaf	0.7	--	--	--	--	--
stem	1.1	--	--	--	--	--
Medicago hispida[2]						
stem	--	--	--	--	2.6	3.6
seed	--	--	--	--	2.6	2.5
Brassica nigra						
leaf	1.65 ± 0.24	1.5 ± 0.56	1.65 ± 0.21	3.25 ± 0.70	2.4 ± 1.56	8.30 ± 4.10
stem	0.78 ± 0.10	1.65 ± 0.77	1.20 ± 0.42	1.96 ± 0.25	1.3 ± 0.71	8.10 ± 2.96
root	0.40 ± 0.26	3.70 ± 2.12	0.6 ± 0.42	1.96 ± 0.60	0.9 ± 0.42	7.65 ± 1.20
flower	0.60	--	--	2.1	1.7 ± 0.00	4.30
Raphanus sativus						
leaf	3.15 ± 1.77	3.45 ± 0.36	--	--	2.0	7.2
stem	1.50 ± 0.00	2.70 ± 0.14	--	--	2.8	5.6
root	1.40 ± 0.14	2.95 ± 0.21	--	--	0.6	6.4
flower[2]	1.0	2.2	--	--	0.2	3.0
Avena fatua[2]						
whole plant	1.3	1.6	--	--	--	--
Bromus sp.[2]						
leaf	2.7	1.4	--	--	0.4	5.6
root	2.8	1.6	--	--	1.1	4.2
Rumex crispus[2]						
leaf	--	--	--	--	2.1	--
stem	1.5	1.7	--	--	0.9	3.3
root	0.5	2.3	--	--	--	4.1
flower	2.3	1.6	--	--	1.2	2.7

1 - no sample available
2 - value based on one sampling time

Silver is seldom studied because of its insolubility and low translocation rates into plants. However, it does serve as a good model for the distribution of insoluble elements in soil. Therefore, as would be expected Ag accumulated in the uppermost portion of the soil (0-15cm) at the Orange County disposal site, and its translocation into forest vegetation appears to be minimal.

In conclusion, the data from this study indicate that the manner and amount of accumulation of trace elements is likely to differ with the pattern of wastewater application, the amount of wastewater applied, the type of soil to which the wastewater is being applied, and the concentration of other trace elements. In the case of Cd, the application of wastewater containing low concentrations of this metal may well result in an increased Cd content in the soil and eventually increased levels in plants growing on that soil. Further study is needed to fully understand different verticle accumulation patterns of elements in different soil types.

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TABLE 9

Concentration of Silver and Nickel in Plant Species from Las Virgenes Test Sites

Plant Species	Site: Las Virgenes Control		Las Virgenes Low Rate		Las Virgenes Moderate Rate	
	Metal ug/g					
	Ag	Ni	Ag	Ni	Ag	Ni
Medicago Sativa						
leaf	--	--	0.48 ± 0.32	2.03 ± 0.72	--	--
stem	--	--	0.27 ± 0.10	1.55 ± 0.56	--	--
root	--	--	0.42 ± 0.11	4.45 ± 1.14	--	--
Medicago lupulina						
leaf	0.3	--	--	--	--	--
stem	0.1	--	--	--	--	--
Medicago hispida						
stem	--	--	--	--	0.2	2.3
seed	--	--	--	--	0.5	1.3
Brassica nigra						
leaf	0.63 ± 0.14	1.55 ± 0.05	0.35 ± 0.07	1.00 ± 0.71	0.4 ± 0.14	0.85 ± 0.07
stem	0.30 ± 0.10	0.73 ± 0.21	0.30 ± 0.00	1.20 ± 0.14	0.6 ± 0.14	1.65 ± 0.07
root	0.20 ± 0.00	1.3 ± 0.72	0.30 ± 0.14	1.20 ± 1.13	0.5 ± 0.14	2.20 ± 0.07
flower	0.3	1.1	--	--		
Raphanus sativus						
leaf	0.70 ± 0.14	2.55 ± 1.06	--	--	0.9	1.5
stem	0.95 ± 0.92	1.90 ± 0.8	--	--	0.3	0.9
root	0.40	2.05 ± 1.9	--	--	0.2	0.4
flower	0.40	--	--	--	0.3	0.6
Avena fatua						
whole plant	0.13	2.13	--	--	0.2	2.5
Bromus sp.						
leaf	0.2	1.1	--	--	0.2	1.0
root	0.3	1.0	--	--	0.2	1.4
Rumex crispus						
leaf	--	--	--	--	0.4	2.1
stem	0.2	1.4	--	--	0.4	1.2
root	0.5	2.1	--	--	--	--
flower	0.4	2.2	--	--	0.3	1.4

1 - no sample available
2 - value based on one sampling

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HEALTH CONSIDERATIONS

THE FATE OF SEWAGE EFFLUENT HEAVY METALS IN LAND WASTEWATER DISPOSAL SITES

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The fate of heavy metals applied to land in secondary treated sewage effluent was reviewed. Concentrations of many metals have been reported to approach, and in some cases exceed, those recommended for continuous irrigation. About half of the metals in secondary treated sewage effluent are associated with the suspended solids. Metals applied to land are generally concentrated in the soil very near to the surface by both adsorption and precipitation mechanisms. The sewage effluent applied metals appear to be more readily extractable than those found in untreated soils. There is no evidence of leachate losses of metal from soils which have been used for sewage effluent disposal.

While effluent applied metals are taken up by plants, the uptake rate is very small compared to application rates, and is too small to significantly reduce the metals in the soil. Metals may leave disposal sites with wind and water eroded soil particles. This can be reduced by maintaining vegetative cover and by minimizing runoff and erosion. With proper management, the metal constituents of sewage effluent should not have a detrimental effect on the environment.

INTRODUCTION

Increasingly, secondary treated sewage effluent is either being disposed of on land or used to irrigate crops. Although the concentrations of heavy metals are generally low in properly treated effluent, long term applications of wastewater which has not been properly treated could result in the accumulation of certain metals to undesirable levels. Much information is available on the fate of sewage sludge applied metals (see Page (1974) for a review), but very limited work has been reported on the fate of metals applied in effluent.

Metals applied to soils may be transformed or lost through a variety of pathways. They may be adsorbed in the soil on mineral or organic surfaces, they may precipitate, or they may be taken up by vegetation. Metals could be lost from the soil by crop removal, or under certain conditions, by leaching through the soil profile. Wind and water borne soil particles may carry the adsorbed metals from the surface.

CONCENTRATIONS OF METALS IN EFFLUENT

The concentration of heavy metals in secondary treated effluents reported by various authors are given in table 1. As would be expected, all reported values are far below those commonly found in sewage sludge. In general, lead and cadmium are found in the lowest concentrations, and zinc and copper are found in the greatest concentrations in effluents.

The extreme high values of all metals are one to two magnitudes greater than the means reported. When the grand mean of all data is compared with acceptable concentrations for continuous irrigation, we find that both Cd and Cu concentrations are greater than the recommended limits. The means of all metals are below those recommended for ir-

Table 1. Mean and range of metal concentrations in secondary sewage effluent reported by various sources. The FWPCA recommended concentrations for irrigation water are also given.

All Volumes ppm.	Page (1974) Dissolved Mean	Page (1974) Dissolved Range	Page (1974) Total Mean	Page (1974) Total Range	Mytelka (1973) Total Range	Menzies/ Chaney (1975) Total Range	Bradford (1975) Total Mean	Bradford (1975) Total Range	Sopper (1973) Total Mean	Spyridakis/ Welch (1976) Total Mean	Grand Mean Total	Extreme Range Total	20 year Irrigation[1]	Continuous Irrigation[2]
Cd	.016	< .005-.04	.027	< .005-.15	< .02-6.4	< .005-6.4	.01	< .005-.22	.009	.1	.0365	< .005-6.4	.05	.005
Cr	.08	< .01-1.0	.20	< .005-1.46	< .05-6.8	< .5-6.8	.007	< .001-.10	.023	.2	.1075	< .001-6.8	20.0	5.0
Cu	.07	< .01-.55	.14	.01-1.3	< .02-5.9	< .02-5.9	.021	.006-.053	.109	.2	.47	.006-5.9	5.0	.2
Hg	-	< .02-.3	.37	< .02-1.0	< .0001-.125	< .0001-.125	-	-	-	-	-	< .0001-1.0	-	-
Ni	.24	< .02-.86	.43	< .02-5.4	< .1-1.5	< .02-5.4	.040	.003-.60	.093	.2	.1908	.003-5.4	2.0	.5
Pb	.038	< .02-.08	.14	< .02-1.3	< .02-6.0	< .2-6.0	.023	.003-.35	.104	.1	.0918	.003-6.0	20.0	5.0
Zn	.21	.01-1.7	.44	.03-4.7	< .02-20.0	< .02-20.0	.05	.004-.350	.211	.3	1.001	.004-20.0	10.0	5.0

[1]FWPCA (1968) for fine textured soils only and when using irrigation practice is for a maximum of 20 years.

[2]FWPCA (1968) for any soil with no time limit on irrigation practice.

rigation limited to 20 years. The extreme high concentrations of all metals reported exceed the recommendations for continuous irrigation, and in most cases exceed those recommended for limited irrigation. It is apparent that irrigation with secondary treated sewage effluent will result in metal applications which do approach, and in some cases exceed, levels recommended for irrigation water. It is thus evident that irrigation and disposal systems utilizing secondary treated sewage effluent should be managed to prevent long term buildup of undesirable metals.

Blakeslee (1973) reported both dissolved and total concentrations of those metals for which he analyzed. Typically 45% of the metals are water soluble. This is a much greater fraction than would be expected for sewage sludge (Page 1974). Thus since about half of the metals in sewage effluent are associated with the solids suspended, even if all the solids were removed the mean cadmium concentration in solution would still exceed the recommended level for continuous application. The averages of all available data were utilized to calculate the amounts of metals which may be expected to be applied per year to land irrigated with sludge effluent. Calculations were done assuming irrigation of 60 cm per year which would normally be used to irrigate a crop and applications of 2.4 M per year which might be applied to permeable soils where the objective is to dispose of the waste rather than to utilize it. The results are shown in table 2. It is evident that rather large amounts of certain metals may accumulate on land used for long term disposal.

Table 2. Anticipated application rates of heavy metals in kg/ha as a function of irrigation rates calculated assuming mean concentrations in sewage effluent.

Metal	Rate of Irrigation 60 cm ha/yr.	240 cm ha/yr.
Cd	0.219	0.876
Cr	0.645	2.58
Cu	2.82	11.28
Hg	2.22	8.88
Ni	1.1448	4.5792
Pb	0.5508	2.2032
Zn	6.006	24.024

ADSORPTION AND LEACHING OF METALS THROUGH SOILS

The mechanisms of adsorption of Cu, Zn, Fe, and Mn have been reviewed by Ellis and Knezek (1972). They suggest that although solubility of oxides of naturally occuring compounds of these metals are extremely low, adsorption and binding of these metals by soil particles is sufficiently strong to play an important role in their availability. Bingham et al. (1964) reported in fact that montmorillonite clay is capable of adsorbing both Zn and Cu beyond its cation exchange capacity, particularly at neutral to alkaline pH's. They explained this by a combination of adsorption and precipitation. Sharpless et al. (1969) reported that Zn added to soils reverted to unavailable forms which can be extracted only by strong acids. There is little information available, however, on the rate of reversion of metals to forms which are less available to plants and less likely to leach through the soil. A listing of solubility product constants of common forms of Cd, Cu, Hg, Fe, Mn, Ni, and Zn is given by Ellis (1973). The hydroxides of all are quite insoluble, as are many of the other commonly found compounds. Thus once the exchange sites are satisfied, the excess ions should be readily precipitated. It should be noted, however, that natural organic matter chelation maintains amounts of metals in solution at greater concentrations than would be expected whether or not precipitation is considered.

Griffin et al. (1976) have reported that the adsorption of cationic heavy metals - Pb, Cd, Zn, Cu, and Cr (III)-increases as pH increases, while the adsorption of anionic heavy metals - Cr (VI), As, and Se - decreases as the pH increases. Precipitation of cationic heavy metals becomes an important mechanism of removal from solution at pH values of 5 and above. They observed no precipitation of the anionic heavy metals between pH 1 and 9. The anions are adsorbed most strongly at pH's of 4 or below.

A year long study of the mobility of five metals applied to the soil in secondary treated sewage effluent was conducted by the author. The sewage effluent was spiked to raise the concentration of each of 5 metals to 1 ppm. Applications of 2 cm of effluent were made per week to the soils enclosed in lysimeters. The movement and distribution of metals in the soil was traced during the study. No metals were detected in the leachate collected from the bottom of the 1.4 meter deep lysimeters during the study. Profiles of the total and DTPA extractable metal concentrations from one of the soils are shown in figures 1, 2, 3, 4, and 5. In all cases, the metals applied accumulated at or very near the soil surface. In most cases, increases in metal concentrations did not penetrate past 10 cm in depth. A large fraction of the applied metals were located in the top 2 cm and could have been removed by scalping this thin layer of soil.

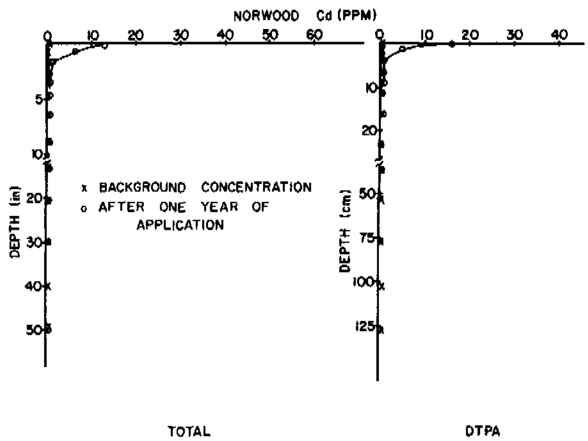

Figure 1. The distribution of total and DTPA extracted cadmium in Norwood sandy clay loam soil following 1 year application of secondary treated sewage effluent containing 1 ppm. Cd.

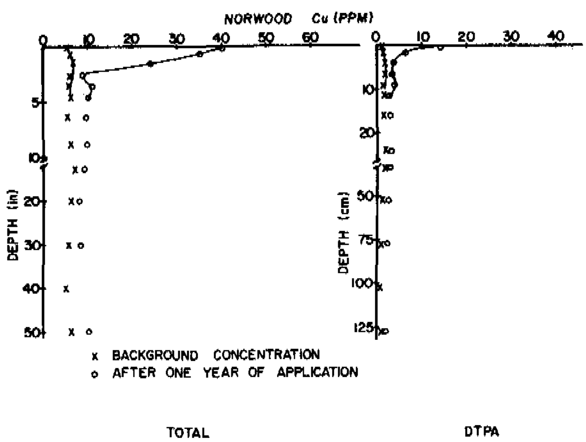

Figure 2. The distribution of total and DTPA extracted copper in Norwood sandy clay loam soil following 1 year application of secondary treated sewage effluent containing 1 ppm. Cu.

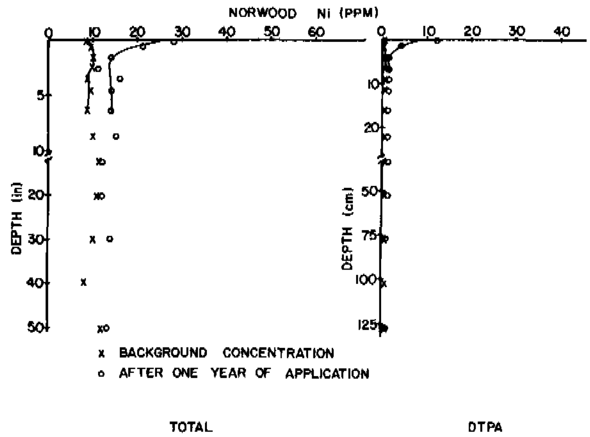

Figure 3. The distribution of total and DTPA extracted nickel in Norwood sandy clay loam soil following 1 year application of secondary treated sewage effluent containing 1 ppm. Ni.

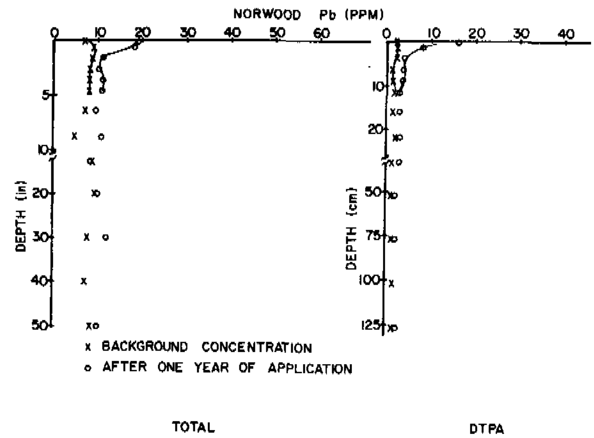

Figure 4. The distribution of total and DTPA extracted lead in Norwood sandy clay loam soil following 1 year application of secondary treated sewage effluent containing 1 ppm. Pb.

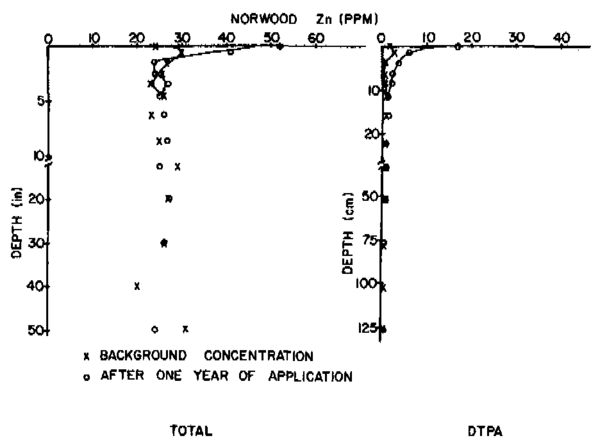

Figure 5. The distribution of total and DTPA extracted zinc in Norwood sandy clay loam soil following 1 year application of secondary treated sewage effluent containing 1 ppm. Zn.

A large fraction of the metals added were extractable with DTPA. For some metals, particularly Ni, Pb, and Zn, it is evident that the effluent metals were more easily extractable from the soils than were the metals indigenous to the soil. Studies are continuing to determine the length of time required for the readily extractable metals to revert to less available forms.

The results shown here are representative of those for all four of the soils studied. There was little evidence that differences in clay content, or initial pH influenced the movement of metals. The pH of the water was alkaline, and slowly shifted all the soils to higher pH levels. This may have masked any differences which might have resulted from the initial pH levels.

LOSSES OF HEAVY METALS ON ERODED SOILS

Since most metals are adsorbed or precipitated very close to the soil surface, the potential for losses with eroding soil is great. The potential is particularly great when repeated applications are made without tilling the soil. Tillage would incorporate the metals throughout the plow layer and thus lessen concentrations at the surface from which erosion takes place. Stevenson and Welch (1977) have reported wind transport of metals from treated plots to adjacent fields. Although no data are available on the concentration of metals in the soil eroded from areas treated with sewage effluent, it stands to reason that the concentrations would be similar to those found in surface soils.

With proper management, metals should not accumulate on disposal sites to undesirable levels. Therefore, even if the soil erodes, it should not cause unduly elevated metal concentrations on adjacent lands. However, if eroded soil accumulates in an aquatic environment, it is possible that certain metals could be released under anaerobic or acid conditions and constitute a hazard. Therefore, normal soil conservation practices including the maintenance of a good ground cover and prevention of undue runoff should be employed.

PLANT UPTAKE OF HEAVY METALS

The literature abounds with reports on the uptake of heavy metals from waste treated soils. Much of the information dealing with the uptake of metals from sludge treated soils has been summarized by Page (1974). Much less information exists, however, on the uptake of metals for soils treated with sewage effluent, but there is little reason to suspect that the mechanism of plant uptake from effluent treated soils would differ much from those treated with sludge.

The fraction of metals taken up by a bermudagrass crop grown on soil treated with 5 cm of secondary treated sewage effluent containing 1 ppm. of Cu, Cd, Ni, Pb, and Zn is given in table 3. It is evident that only small quantities of the applied metals can be removed by crop uptake. The grass used would require 50 to 1000 years to remove the metals applied during 1 year if removal continued at the same rate as given here. Excess amounts of nearly all metals have been shown to be toxic to plants, while some metals, particularly Cd, may be taken up in concentrations that could be transmitted in toxic concentrations to animals. However, if metals are not allowed to accumulate to excessive amounts, the hazard of transmission from areas used to dispose of sewage effluent should be minimal.

Table 3. Percent of metal taken up by bermudagrass during one year's growth from four soils integrated with 5 cm of secondary treated sewage effluent per week.

Metal	Nacogdoches clay pH 5.6	Norwood sandy clay pH 7.7	Bastrop sandy clay loam pH 6.9	Lakeland sandy loam pH 6.4
Cd	0.4%	0.3%	0.1%	0.1%
Cu	0.9%	0.5%	0.2%	0.3%
Ni	0.6%	0.4%	0.2%	0.3%
Pb	0.6%	0.5%	0.2%	0.2%
Zn	2.0%	1.3%	0.6%	0.5%

Contribution of the Texas Agricultural Experiment Station. Their work was supported in part by the U. S. Army Medical Research and Development Commission. Contract N. DAD 17-73-C-3102.

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HEALTH CONSIDERATIONS

SORPTION OF CADMIUM ONTO TWO MINERAL SOILS

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Sorption of cadmium in low range concentrations (5-50 ppb in solution) onto two mineral soils (< 0.5 mm fraction) has been studied by means of batch experiments. Data on sorption isotherms, the kinetics, the influence of pH, competitive effects of calcium, and the reversibility of the sorption process are presented.

INTRODUCTION

The attentuation of heavy metals in land treatment systems is of great concern, in particular where the effluents recharge aquifers or discharge into surface waters used for water supply. In this connection, cadmium is of special interest due to its relatively high mobility in soil and its high toxicity. The current average cadmium intake is approaching the recommended limits of recommended maximum tolerance levels with contact through plants as the major source of cadmium.

The soil is generally an effective filter for attenuation of Cd (although typically less effective for Cd than for several other cationic metals, e.g. Pb). Several investigations on pilot land treatment systems and filtration systems (/6/, /7/, /23/, /29/, /32/) indicate that the major portion of Cd applied is retained within the upper 1 m of soil. Except for industrial waste waters, the efficiency of the soil filter to attenuate Cd may only be of concern in the long term. The long term problems are: (1) possible depletion of the attenuation capacity of the soil, which can limit the lifetime of the land treatment site, (2) possible migration of previously retained metal due to infiltrating precipitation after close down of the site. To evaluate these aspects the attentuation mechanism per se must be studied.

Several mechanisms probably are involved in the attenuation of Cd in land treatment systems. (A recent, general discussion of possible attenuation mechanisms is presented in /10/). Attenuation by physiochemical sorption processes at the soil-water interface is considered most important and hence selected for research, because of the variable signigicance of some of the mechanisms involved (e.g. microbial assimilation) and the complexity of the superimposing mechanisms. This research approach probably results in conservative estimates on the attenuation, but at the present state-of-the-art it is the most feasible approach for studying the long term aspects of the attenuation mechanisms.

Sorption of Cd from solutions containing high concentrations of Cd (more than 1 ppm Cd, > 10^{-5} M Cd) onto soil or common soil constituents has been frequently reported in the literature (/1/,/2/,/3/,/4/, /5/,/8/,/9/,/14/,/16/,/17/,/18/,/19/,/20/, /21/,/22/,/23/,/24/,/26/,/27/,/28/,/33/, /34/). The investigations have generally included determination of sorption isotherms and the influence of pH on sorption capacities. The competitive effect of calcium and other ions (/2/,/3/,/8/,/22/,/26/, /34/) and the reversibility/irreversibility of the sorption process (/1/,/3/,/16/, /19/,/21/,/27/) have been subject to research, only in a few cases. The literature is somewhat confusing, in particular concerning the desorbability. To some extent, this may be due to misinterpretation of experimental results. These findings will not be discussed further because extrapolation of the results to a concentration range pertinent for land treatment systems (5-50 ppb Cd; $5 \cdot 10^{-8}$M-$5 \cdot 10^{-7}$ M Cd) is probably not acceptable.

Very few investigations have dealt with sorption from low range Cd concentrations (/11/,/12/,/13/,/25/,/31/). Garcia-Miragaya & Page (/11/,/12/) studied the influence of Cl-complexation and ionic strength on the sorption by montmorillonite in the 15 to 120 ppb Cd range. Gardiner (/13/) applied partition coefficients to Cd sorption on river muds, clays, silica, and humic material and found decreasing partition coefficients for increasing Cd concentrations (2-177 ppb Cd). Furthermore, the experiments showed that the sorption-desorption process was fast and that the presence of EDTA decreased the sorption. Levy & Francis (/25/) studied the influence of salt saturation, Fe/Al-oxide coatings, and organo-clay complexes on the sorption by montmorillonite. They found less sorption when Ca^{2+} was present compared to e.g., Na^{+}. They also found that organo-clay complexes caused appreciable difference in Cd sorption only when montmorillonite was coated with oxides. Only the oxide coated clays exhibited significant irreversibility in the desorption studies.

Sidle & Kardos (/31/) reported on the kinetic aspects of Cd sorption onto a forest soil using an empirical rate equation. Equilibrium conditions were approached within 10 hours, and at equilibrium the sorption was expressed in terms of Freundlich and Langmuir isotherms.

OBJECTIVES

Because insufficient information is available on the attenuation of Cd in soils, the objective of the research presented in this paper is to obtain quantitative information on the soil sorption mechanism at low range concentrations of dissolved Cd (5-50 ppb Cd). This involves: (1) determination of isotherms, (2) competitive effects of Ca^{2+}, (3) the kinetics of the sorption process, (4) the effect of pH on sorption capacities, and (4) evaluation of the reversibility/irreversibility of Cd sorption by means of sorption-desorption studies employing varying bracketing conditions with respect to Ca^{2+} and H^{+}.

MATERIALS AND METHODS

Soils

Two soils were selected from the Danish Soil Library: 16C (loamy sand; C-horizon: 50-100 cm) and 67C (sandy loam, C-horizon: 65-100 cm), on the basis of their low organic content, fairly high content of sand and moderate pH. The soils selected are considered to be typical for land treatment infiltration systems. Because of the need for improved experimental reproducibility, the coarse fractions were removed by dry sifting in a 0.5 mm mesh metal free sieve. The preliminary characterization of the soils is presented in Table 1.

Table 1: Charaterization of the soils employed.

	16 CS	67 CS
Type: < 0.5 mm fraction of C-horizon (0.5-1 m depth)	Loamy sand	Sandy loam
Texture of < 0.5 mm fraction:		
% coarse sand (200-500 μ)	31.6	16.0
% fine sand (20-200 μ)	54.7	53.5
% silt (2-20 μ)	6.8	12.0
% clay (< 2 μ)	6.2	18.0
% organic matter	0.7	0.5
pH (10^{-2} M $CaCl_2$)	6.0	6.5
CEC (meq/100 g)	7.5	17.6
Mn (μg/g):		
Dithionite-citrate	82	55
Hot 1:1 HNO_3 for 8 hours	74	110
Fe (μg/g):		
Dithionite-citrate	2790	5560
Hot 1:1 HNO_3 for 8 hours	5800	15100
Zn (μg/g):		
Hot 1:1 HNO_3 for 8 hours	18	29
Cd (μg/g):		
Hot 1:1 HNO_3 for 8 hours	0.03	0.035
Mineral composition of clay fraction (X-ray analysis of sodiumcarbonate and dithionite + citrate pretreated clay fractions).	The clay fraction of both soils is dominated by the 14-Ångström-minerals (montmorillonite, vermiculite, chlorite). Both contain some hydrous micas and kaolinite, and a little quartz and feldspar. The relative differences are: 16 CS: - more chlorite - some more amphibole 67 CS: - more hydrous mica - more interstratified vermiculite	

Salt Matrix

The soils were generally exposed to a specially cleaned 10^{-3} M $CaCl_2$ salt matrix ($Ca^{2+} \simeq 40$ ppm). Cd, stored as acidified 10 and 100 ppm stock solutions obtained from $CdCl_2$ standards, was added to the salt matrix by calibrated, fixed volume micropipettes yielding the initial Cd concentrations (< 5% of Cd is present as $CdCl^{+}$ and higher Cl-complexes, /15/). The pH was adjusted by addition of microamounts of analytical grade HCl or $Ca(OH)_2$. In some experiments, the salt matrix consisted of 10^{-2} M $CaCl_2$.

Experiments

The research was conducted as batch experiments. The kinetic experiments employed a mechanically stirred 2000 ml polyethylene (PE) reactor. The pH *equilibration* (pH = 6.00 ± 0.03) of the soil-salt matrix slurry was accomplished before the addition of Cd. The pH adjustment was conducted continuously after the start of the experiment. At varying time intervals, 5 ml aliquots were sampled, by a pipette with disposable tips, and transferred to centrifuge tubes for separation. All other experiments employed 100 ml PE reactors (0.5 - 6 grams of soil; 50-100 ml salt matrix) agitated in a rotating mixer for 20 hours and manually adjusted three to six times to the proper pH. The 100 ml PE reactors were specially fitted into the centrifuge tube holders, and after completion of the agitation the slurry was separated by centrifugation without transferring the sample. All samples were separated at a theoretical particle size of 0.2 µm. The supernatant was removed by pipetting (less than 1% of liquid remained in the reactor).

In some of the desorption studies, the desorbing salt matrix was added instantaneously after separation. In other desorption studies, the soils were cautiously dried in the reactor under occasional mixing with a PE rod. The dried soil was homogenized by means of the rod and aliquots of the soil transferred to several other reactors for desorption. The desorption experiments were conducted analogously to the sorption experiments. All experiments were conducted at room temperature. The liquid samples were transferred to PE bottles, acidified (conc. HNO_3) and stored at 1^oC.

Cd analysis

The Cd content of the samples was determined by flameless atomic absorption spectrophotometry (Perkin-Elmer 300 S supplied with a HGA-70 graphite furnace and deuterium background compensator). In order to obtain a safe and high-quality Cd-analysis, all samples and standards were subject to solvent/solvent extraction: A sample volume was transferred to a glass test tube, buffered with purified citrate, and extracted by 0.5% DDDC (Diethylammonium-Diethyl-Dithio-Carbamate, $C_9H_{22}N_2S_2$, Mercks No. 3408) dissolved in xylene (Dimethylbenzene, $C_6H_4(CH_3)_2$, May & Baker) during agitation in a rotating mixer for 30 minutes. Three to five ppb Cd in the xylene phase is optimal. Twenty µl of the organic phase was injected twice into the graphite furnace.

Precision and accuracy

Several calibrations with industrial standards indicate that the analytical method employed does not cause any systematical deviation from accurate determination of Cd. Estimates on the precision of determination of Cd in a liquid sample are about 6% (standard deviation), and total precision of the sorption experiment, based upon independently conducted replicational sorption experiments and Cd determinations, is within 10%.

Mass balances of the experiments show that no unaccounted, significant sources or sinks of Cd were present. Thus a decrease in dissolved Cd will be considered balanced by an increase in sorbed Cd, and vice versa.

RESULTS AND DISCUSSION

Sorption isotherms

The amount of Cd sorbed per unit of soil (µg Cd per g of soil) as a function of salt matrix Cd equilibrium concentration is presented in Figure 1 and Figure 2 for the respective soils. The salt matrix is 10^{-3}M $CaCl_2$ and pH is 6.0. Both soils exhibit slightly *nonlinear* sorption

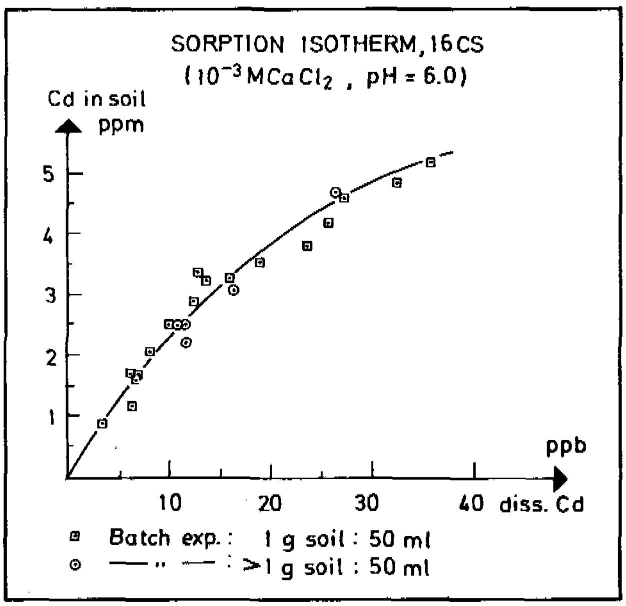

Figure 1: Isotherm for sorption of Cd by a loamy sand (16 CS) from a 10^{-3} M $CaCl_2$ saltmatrix at pH = 6.0.

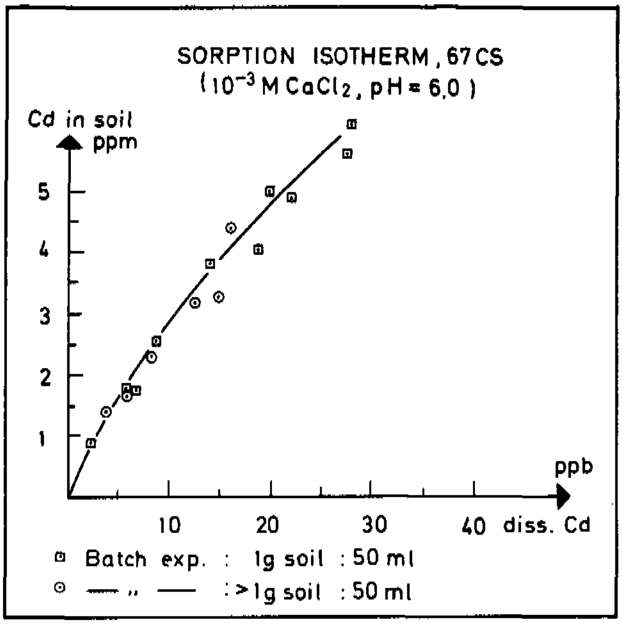

Figure 2: Isotherms for sorption of Cd by a sandy loam (67 CS) from a 10^{-3} M $CaCl_2$ saltmatrix at pH = 6.0.

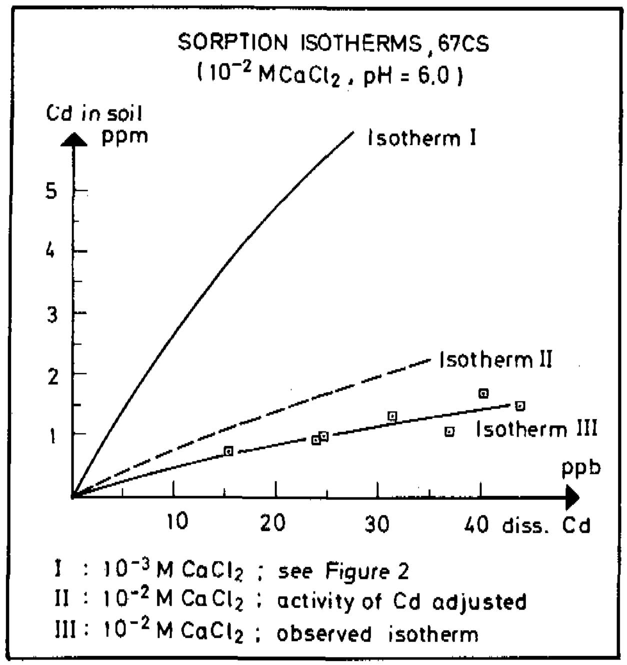

Figure 3: Isotherm (III) for sorption of Cd by a sandy loam (67 CS) from a 10^{-2} M $CaCl_2$ saltmatrix at pH = 6.0. Isotherm I corresponds to the isotherm in Figure 2 (10^{-3} M $CaCl_2$). Isotherm II is calculated from isotherm III and accounts for reduced sorption due to Cl-complexation and the effect of ionic strength.

isotherms with sorption maxima beyond the dissolved concentration range considered pertinent in this study. The sandy loam (67 CS) has a somewhat higher sorption capacity than the loamy sand (16 CS). The data points are obtained from several experimental series and express the total scatter around the isotherm. As indicated on the graphs, not all data points originate from experiments employing the same soil-salt matrix ratio. The fact that these deviating soil-salt matrix ratios do not cause a deviation from the isotherm, implies that the isotherm is not subject to influence from the experimental set-up. Naturally, the soil-salt matrix ratio affects the composition of the liquid with respect to macroions (which may compete for the sorption sites), but not significantly.

Competitive effects of Ca^{2+}

In Figure 3, an isotherm for sorption by the sandy loam (67 CS) from a 10^{-2} M $CaCl_2$ salt matrix at pH = 6.0 is presented. Compared to the 10^{-3} M $CaCl_2$ sorption isotherm, the 10^{-2} M $CaCl_2$ isotherm shows considerably less sorption. This is due to increased Cl-complexation, increased ionic strength and increased Ca^{2+} competition. The Cl-complexation and the effect of ionic strength reduce theoretically the Cd activity to 55% of the activity in the 10^{-3} M $CaCl_2$ salt matrix. The dashed isotherm (II) in Figure 3 accounts for this, and the difference between isotherm I (the 10^{-3} M $CaCl_2$) and isotherm II expresses the competitive effect of Ca^{2+}. This experiment shows that even in low range concentrations of Cd, the Ca^{+2} ion is apparently a significant competitor for sorption sites.

Kinetics of sorption

The kinetics of the sorption process are illustrated in Figure 4, where the dissolved Cd concentration is plotted versus time after addition of 60 ppb Cd to the soil-salt matrix slurry (16 CS). The graph shows that the sorption process is very fast: 95% of the sorption takes place within 10 minutes and equilibrium is apparently approached within 1 hour. Analogous experiments with the sandy loam showed corresponding results. The 20 hours contact time employed in the batch ecperiments seems sufficient.

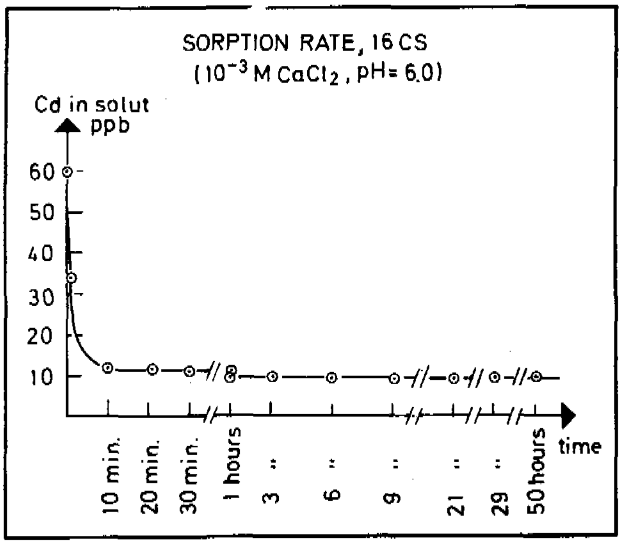

Figure 4: Dissolved Cd (ppb) as a function of contact time for sorption at pH = 6.0 by a loamy sand (16 CS) from a 10^{-3} M $CaCl_2$ saltmatrix holding 60 ppb Cd initially.

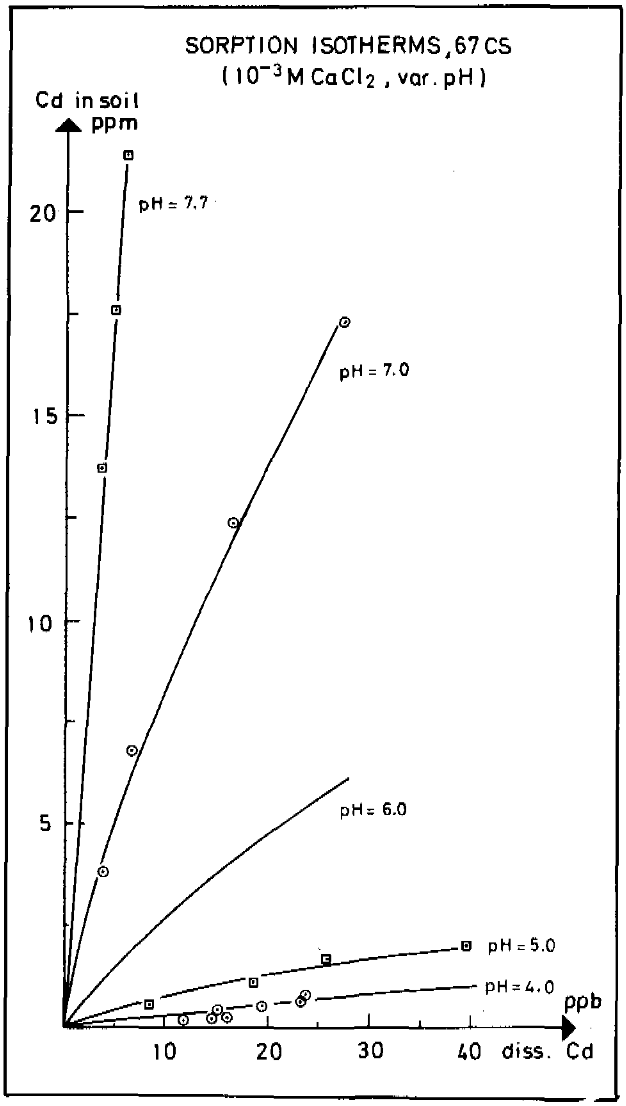

Figure 5: Isotherms for sorption of Cd by a sandy loam (67 CS) from a 10^{-3} M $CaCl_2$ saltmatrix at varying pH (4.0-7.7).

Influence of pH

The influence of pH on the sorption capacity is illustrated in Figure 5, where equilibrium isotherms at varying pH are presented for Cd sorption by a sandy loam (67 CS) from a 10^{-3} M $CaCl_2$ salt matrix. Acid conditions lower than pH 4.0 were considered environmentally unimportant and the upper pH (7.7) was defined by an apparent precipitation, which may result in removal of Cd from solution by other processes than soil sorption. The plot shows that the sorption capability is extremely influenced by pH. In particular above pH = 6, the sorption capacity increases substantially with increasing pH. This phenomenon is probably due to less H^+ competition at alkaline pH and to the formation of readily sorbed $Cd(OH)^+$ - complexes; the latter argument is by James and Healy (/18/). Analogous experiments with the loamy sand (16 CS) showed corresponding results. However, the sandy loam (67 CS) seems to be more influenced by pH than the loamy sand (16 CS).

Reversibility of Cd sorption: 16 CS, 10^{-3} M $CaCl_2$, pH=6.0

The reversibility of sorption by the loamy sand (16 CS) was studied by desorbing previously Cd loaded soil with the same salt martrix: 10^{-3} M $CaCl_2$. The results are presented in Figure 6. The dashed line governed by a Cd mass balance illustrates the Cd-pathway of the sorption-desorption sequence. At the start of the sorption experiments all Cd is present in the salt matrix and at the start of the desorption experiment all Cd is present in the soil. The graph shows that the loamy sand does not exhibit any irreversible sorption of Cd: The desorption concentrations are randomly scattered around the sorption isotherm. A few of the data points are obtained by repeated desorption. Some of the data points are obtained from desorption of dried and stored soils while others are obtained from desorption experiments conducted immediately after com-

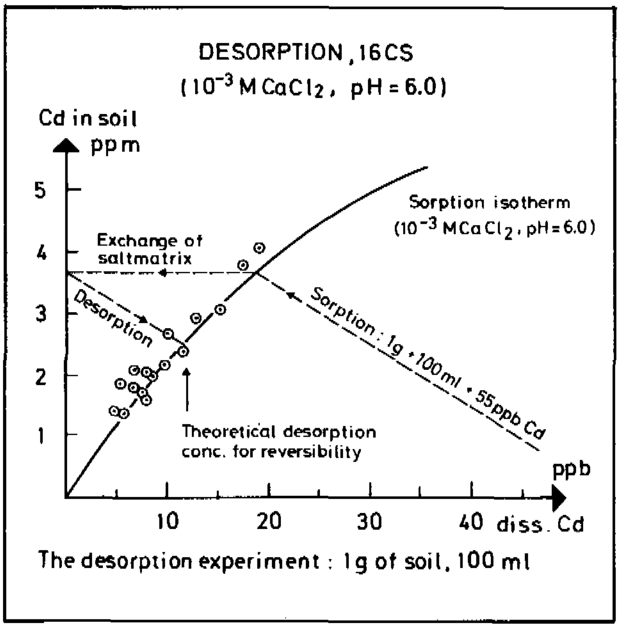

Figure 6: Cd concentrations obtained by desorption of a Cd loaded loamy sand (16 CS) with 10^{-3} M $CaCl_2$ at pH = 6.0.

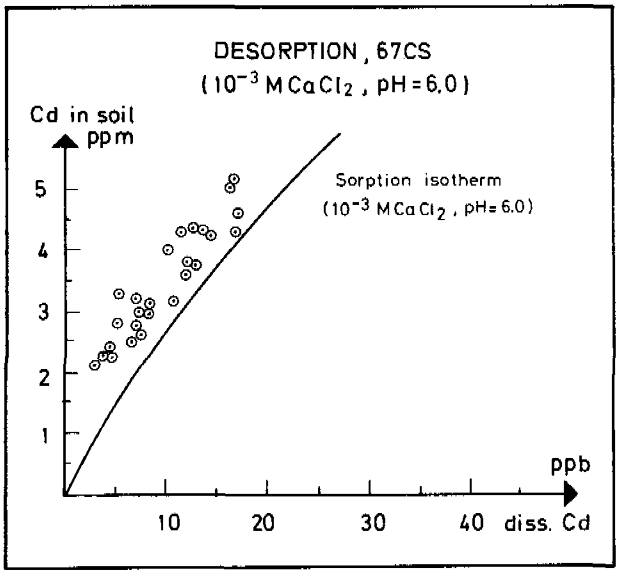

Figure 7: Cd concentrations obtained by desorption of a Cd loaded sandy loam (67 CS) with 10^{-3} M $CaCl_2$ at pH = 6.0.

pletion of the sorption experiments. The experiment indicates that the predominant part of the Cd sorption by the loamy sand is an ion exchange process.

Reversibility of Cd sorption:
67 CS, 10^{-3} M $CaCl_2$, pH=6.0

Corresponding desorption experiments were conducted employing the sandy loam (67 CS). These results are presented in Figure 7. The sandy loam exhibits a slight irreversible sorption of Cd: The desorption concentrations are systematically located above the sorption isotherm. The amount of Cd irreversibly sorbed at the actual conditions is of the order of 1 µg Cd per g of soil.

Reversibility of Cd sorption:
67 CS, 10^{-2} M $CaCl_2$, pH=6.0

Experiments involvind desorption by a 10^{-2} M $CaCl_2$ salt matrix of the sandy loam loaded with Cd available in a 10^{-2} M $CaCl_2$ salt matrix are presented in Figure 8. The sandy loam does not exhibit any significant irreversible sorption of Cd in the experiments employing a 10^{-2} M $CaCl_2$ salt matrix. This emphasizes that the small, but significant irreversible sorption observed in the experiments employing a 10^{-3} M $CaCl_2$ salt matrix is not definite, but dependent on the composition of the salt matrix. The fact that only a 10 fold increase in Ca^{2+} concentration reduces the irreversible sorption to a non-observable level, indicates that the sites irreversibly sorbing Cd from a 10^{-3} M $CaCl_2$ salt matrix possess only low energy.

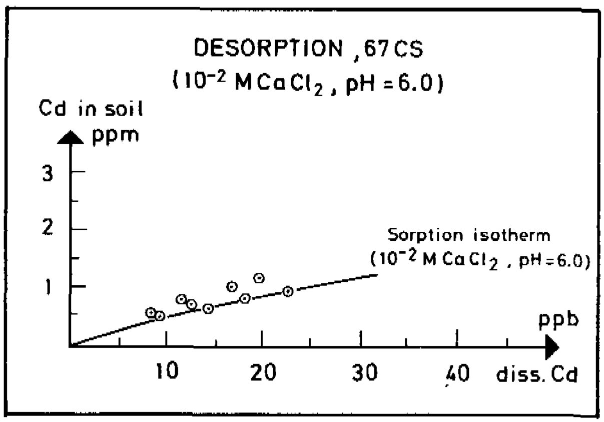

Figure 8: Cd concentrations obtained by desorption of a Cd loaded sandy loam (67 CS) with 10^{-2} M $CaCl_2$ at pH = 6.0.

Reversibility of Cd sorption:
67 CS, 10^{-3} M $CaCl_2$, pH=4.0

The irreversibility of Cd sorption from a 10^{-3} M $CaCl_2$ salt matrix and pH = 4.0 was studied by desorption at corresponding conditions. The results presented in Figure 9 show that no significant irreversibility is present at pH = 4.0.

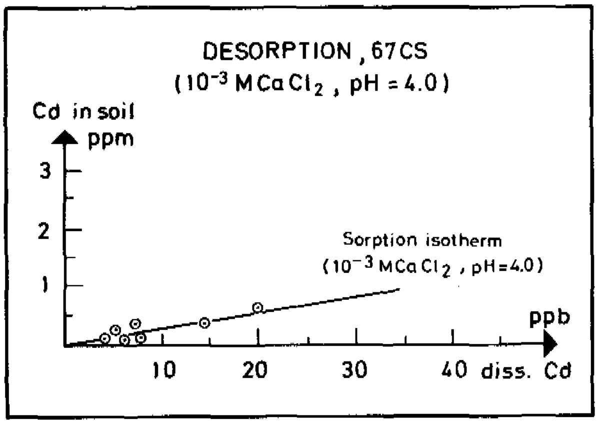

Figure 9: Cd concentrations obtained by desorption of a Cd loaded sandy loam (67 CS) with 10^{-3} M $CaCl_2$ at pH = 4.0.

CONCLUSIONS

Sorption of Cd from low range concentrations by 2 sandy soils (loamy sand 16 CS, sandy loam: 67 CS) was described by slightly non-linear sorption isotherms. Equilibrium was approached within 1 hour of contact time. The Ca^{2+} concentration and, in particular, pH significantly influenced the sorption capacities of the soils. Sorption by the loamy sand (16 CS) was completely reversible indicating that sorption is due to ion exchange. The sandy loam exhibited a small, but significant irreversible sorption of Cd from a 10^{-3} M $CaCl_2$ salt matrix at pH = 6.0. But a 10^{-2} M $CaCl_2$ and at pH = 4.0 no irreversibility was observed, indicating that the irreversibility observed at 10^{-3} M $CaCl_2$ at pH = 6.0 is not specific.

The experiments indicate that the sandy soils will extensively sorb free Cd ions present in land applied waste waters. But the sorption capacities of the soils are significantly influenced by the Ca^{2+} and H^+ (and maybe other ions) concentrations of the soil-wastewater system. The relative small or lacking irreversible sorption of Cd, indicates that Cd may migrate from the land treatment site after the close down of the site.

Further experiments are being conducted concerning (1) the effect of Zn and EDTA on the irreversibility of Cd sorption, (2) the effect of aging on the irreversibility of Cd sorption by the sandy loam, (3) the applicability of the batch experiment retults to column experiments, and (4) the soil constituents causing the irreversible sorption of Cd.

ACKNOWLEDGEMENT

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MONITORING PROCEDURES

A PROTOTYPE PROGRAM FOR MONITORING DOMESTIC WASTEWATER LAND TREATMENT SYSTEMS

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Land treatment of domestic effluent is currently a limited but growing practice at Army installations. Under the authority of the Office of The Surgeon General, a need to provide guidance on public health aspects of land treatment systems has been recognized. This paper presents a comprehensive prototype monitoring program for discharges, surface waters, and ground waters relating to land treatment of domestic wastewater. The program sets forth minimum surveillance requirements based on identified water quality monitoring objectives designed to provide sufficient public health and environmental protection, meet the requirements of state and Federal regulatory agencies, assist the plant manager in controlling effluent quality, contribute to public confidence in the acceptability of land application as a viable and safe treatment technique, and permit the accumulation of data on water quality enabling demonstration of compliance with existing state criteria for public water supply sources. The recommended monitoring program specifically addresses the positioning and design of the sampling station network; selection of appropriate parameters and monitoring frequencies; collection, measurement, and analytical techniques and materiel requirements. Schedules are developed for initial sewage effluent characterization, baseline monitoring of native waters, and operational management and compliance monitoring. Rationale for the design criteria are provided to permit adaptability and appropriate adjustments keyed to existing site configurations and unique conditions.

BACKGROUND

There is an obvious requirement to safely and economically treat or dispose of increasing quantities of domestic liquid wastes. The existence of numerous land treatment systems around the country demonstrates that the land treatment alternative has often been demonstrated to be both a viable and cost effective method of achieving adequate renovation of domestic wastewaters, if properly designed, operated, and monitored. Land application of domestic effluent is currently a limited, but growing practice in the Army. Many installations are currently utilizing, designing, or considering land treatment technology where effluent quality upgrading is required to achieve the water quality objectives established by the Federal Water Pollution Control Act Amendments of 1972 and 1977.

From a survey conducted by Morris and Jewell of Cornell University,[1] it was shown that only 21 out of 50 states have any regulations or guidelines pertaining to the monitoring of a land treatment system. In addition, out of these 21 states, only a few specify more than the provision of a monitoring well in the direction of ground-water flow away from the treatment site. Therefore, under the authority of the Office of The Surgeon General, a need to provide guidance on monitoring and public health aspects of land treatment systems has been recognized. Detailed monitoring guidance to assist military facilities was developed in 1975 and is contained in a technical report prepared by the

US Army Environmental Hygiene Agency,[2] which was distributed by the Office of the Chief of Engineers to major Army commands.

OBJECTIVES

This paper presents a comprehensive prototype monitoring program for discharges, surface waters, and ground waters relating to land treatment of domestic wastewater. The program sets forth recommended minimum surveillance requirements based on the following identified water quality monitoring objectives:

-Provide sufficient public health and environmental protection

-Meet the requirements of state and Federal regulatory agencies

-Assist the treatment plant manager in controlling effluent quality

-Contribute to public confidence in the acceptability of land application as a viable and safe treatment technique.

-Permit the accumulation of water quality data enabling demonstration of compliance with existing state criteria for public water supply sources.

Goals for resultant water quality should derive directly from established state surface water and ground-water criteria or from specific water use designations applied to state water resources. However, where such criteria have not yet been defined, it is necessary to make some basic assumptions concerning existing and potential ground-water uses and appropriate water quality criteria related thereto, in order to select meaningful parameters for the monitoring program. In that regard, it is believed that, after renovated domestic wastewater enters an aquifer and mixes with native ground water, the resultant water quality should approach drinking water standards. This is considered to be a reasonable and attainable goal, unless the native ground water itself is nonpotable.

MONITORING PROGRAM DESIGN

The recommended monitoring program specifically addresses the positioning and design of the sampling station network; selection of appropriate parameters and monitoring frequencies; collection, measurement and analytical techniques and materiel requirements. Schedules are developed for initial sewage effluent characterization, baseline monitoring of native waters, and operational and compliance monitoring. Brief rationale for the design criteria are provided to permit adaptability and appropriate adjustments keyed to existing site configurations and unique conditions.

The recommendations are considered conceptually applicable to all major modes of land treatment. However, modifications may be desired when dealing with systems other than slow rate irrigation. For example, ground-water surveillance may not be necessary in support of overland flow systems on very thick impermeable soils, and it may be advantageous to monitor soil moisture for operational purposes at rapid infiltration sites or for compliance purposes where the water table is exceedingly deep.

SURVEILLANCE STATIONS

Pretreated Effluent Monitoring. A requirement is stipulated to monitor the flow and quality of the liquid waste after pretreatment but before land application. Data obtained from this monitoring station would allow: demonstration of compliance with applicable pretreatment and disinfection criteria, effluent quality control by maintenance of proper pretreatment conditions, and determination of hydraulic and parameter loadings.

Surface Water Monitoring. A requirement is stipulated to monitor the quality of "down-gradient" surface waters, both natural and artificial water bodies, which may consist, in whole or in part, of renovated wastewater derived from land treatment. The positions of several representative surface water sampling stations are depicted on Figure 1: their locations and rationale are discussed below.

Springs and Landlocked Ponds - All springs and landlocked surface water bodies, either perennial or intermittent, which occur between the land treatment site and the installation boundary (or a major flowing surface water body, whichever is closer), in the direction of declining water table elevation, should be monitored. Such monitoring would allow determination of the concentration levels and trends of any contaminants, derived from the applied liquid waste, which may be present in the ground water which is "discharged" to the surface at that point. Although landlocked ponds will not convey any contamination via direct discharge to surface streams, as in the case of springs, all such surface water bodies, which are fed primarily by ground-water seepage, should be monitored to ascertain the absence of contamination levels which could adversely affect the local bio-ecosystem or the existing or potential use of that water.

The lateral limits of surveillance coverage may be approximated by triangular side areas, each defined by a 45 degree angle to the direction of regional ground-water flow, as shown on Figure 1. Because the direction of groundwater flow is often not linear and not static (being constantly modified by changes in aquifer recharge and discharge), and considering the effect of lateral dispersion, the side angle coverage provides a factor of safety to assure the detection of potential contaminants.

Renovated Wastewater Collection Points - Where "renovated" wastewater must be intercepted by ditches along the site perimeter, or where shallow subsurface water must be artificially collected via drains or discharge wells following the land treatment phase, this collected liquid should be routinely monitored prior to final discharge to a surface water body. Such data are required to allow demonstration of compliance with National Pollutant Discharge Elimination System (NPDES) permit requirements.

Ground-water Monitoring. Ground water under and around land treatment sites should be monitored to document the maintenance of a water supply source of adequate quality with respect to public health and environmental protection standards. Figure 1 depicts representative locations of shallow ground-water observation wells for a single contiguous land treatment site.

Onsite Wells - One centrally positioned surveillance well should be constructed within the land treatment site to monitor the height of the ground-water mound generated beneath the area of application as well as the quality of the renovated wastewater upon entering the zone of saturation. Such a well could provide early detection of potential pollution breakthrough should such a situation develop.

When "significantly" nonuniform conditions of soil permeability, wastewater quality, application mode, or hydraulic loading are instituted in the land treatment system, additional onsite wells should be included in the monitoring program to document anticipated variations in renovated wastewater composition from one part of the site to another. Examples of "significant" differences, necessitating the utilization of additional onsite wells, include: year-round application of wastewater to one area of the site and short periodic application to the rest of the site, the use of two or more major modes of treatment (based on soil variations), or segregated applications of wastewaters from very different sources.

Baseline Well - One surveillance well should be positioned outside the treatment site and in the up-gradient direction of general ground-water flow. It should be situated far enough away from the site perimeter to be beyond the influence of the ground-water mound created by wastewater recharge in order to continually monitor the quality of native ground water. This well will serve to document any inflowing ground-water contamination resulting from some activity other than the land treatment operation.

Perimeter Wells - Surveillance wells should also be distributed along the disposal site perimeter in all directions which normally experience ground-water flow away from the site. These perimeter wells would enable monitoring of the resultant ground-water quality after substantial mixing with the treated wastewater had occurred and would serve to demonstrate compliance with regulatory ground-water quality criteria. It is suggested that an "appropriate" number of these perimeter wells should be a function of the length (in meters) of the projection (P) of the egress perimeter onto a line perpendicular to the ground-water gradient. The following formula is suggested, where PW is the number of perimeter wells, rounded off to the nearest whole number: $PW = (P/500)+1$.

PARAMETER SELECTION

Wastewater Characterization. In order to set forth a meaningful list of parameters for routine effluent surveillance, the wastewater must initially be characterized prior to land treatment implementation. Characterization may be accomplished by a one-time sampling and analytical effort. In addition to defining the concentration levels of conventional indicator and health-significant parameters, waste characterization will influence system design by demonstrating the degree of pretreatment required to prevent obvious environmental pollution or system failure. Table 1 presents the list of parameters recommended for initial effluent characterization, based on the following brief rationale.

The 10 constituents termed "health-significant" are included to provide quantification of those substances which bear adversely on public health and which typically are presented in municipal or domestic effluent (although normally at low levels). The indicator parameters: biochemical oxygen demand, chemical oxygen demand, fecal coliform, pH, and dissolved solids serve to characterize the general quality of the wastewater in terms of oxygen demand, dissolved organic content, indicator organisms, hydrogen ion activity, and dissolved ion concentration. High dissolved solids can be potentially deleterious to vegetation, recreational water bodies, and groundwater supplies. The various forms of nitrogen and phosphorus will define the anticipated major nutrient loadings. Nitrogen loading is particularly important from a health viewpoint in that all forms of applied nitrogen have the potential for being transformed within the soil environment to nitrate, a potential source of ground-water pollution. Very high concentrations of oil and grease or suspended solids could potentially lead to a loss of soil infiltration capacity. Sodium, calcium, and magnesium are analyzed in order to calculate the Sodium Adsorption Ratio which, in turn, is an estimate of soil permeability reduction potential. Specific conductance, boron, and copper concentrations are compared to the threshold toxicity levels for vegetation. Many other metals of concern in water used for irrigation purposes do not normally exist in domestic effluent in ionic concentrations at or above plant toxicity threshold levels. Therefore, unless the presence of those substances is suspected (due to geographical prevalence or industrial input), they need not be considered for analysis.

Operational and Compliance Monitoring Program. The routine surveillance program recommended herein is designed to provide the minimum data required for maintenance of satisfactory system performance (including pretreatment), for protection of public health and environmental quality, and for compliance demonstration with any existing state regulatory criteria for discharges, surface waters, and ground waters. The following brief rationale apply to the specific parameters recommended in Table 2.

Biochemical oxygen demand and suspended solids determinations aid in the definition of pretreatment efficiency; pH is a general indicator of water quality. Chemical oxygen demand is an indicator of organic content and serves to document the system's continuing ability to remove oxygen-demanding substances. Fecal coliforms are an indicator of pathogenic microorganisms of intestinal origin. Wherever chlorination is provided, chlorine residual determinations should be performed to document the adequacy of disinfection. Flow measurements enable proper management of the hydraulic load and interpolation of constituent loadings. Nitrate/nitrite-nitrogen is a particularly important parameter to follow throughout the course of domestic wastewater land treatment because of its health implications. Total phosphate analyses will define the potential development of conditions favoring eutrophication in receiving surface waters. Phosphorus analyses are therefore only required where native surface water bodies are in close proximity to the treatment site. Measurement of the fluctuations of the static water level in each monitoring well will permit continual evaluation of ground-water mounding effects, flow patterns, and trends.

In addition to the parameters given in Table 2, any health-significant constituent demonstrated (by initial wastewater characterization) to be in excess of the permissible limit recommended for drinking water should be included in the routine monitoring program. Still other parameters could be designated for mandatory surveillance at any of the sampling stations by responsible state regulatory authorities.

FREQUENCY SELECTION

Initial Surveillance. Initial wastewater characterization should be performed prior to design of a land treatment system. Three to seven days of continuous 24-hour flow-composited samples, supplemented by analyses of discrete samples as required because of preservation requirements and outlying values, should suffice for waste characterization. Also prior to land treatment implementation, all specified native surface water and ground-water surveillance stations should be grab sampled on a quarterly basis at least four times to provide baseline data on the native water quality. All parameters to be included in the operational and compliance monitoring program should be analyzed/measured in these baseline samples.

Operational and Compliance Monitoring. In lieu of more stringent NPDES permit or other state requirements, one measurement or sample and analysis per month, for all specified parameters, should be the minimum frequency needed to meet the stated surveillance objectives of all sampling stations. One exception is that continuous measurement of the pretreated effluent flow is encouraged. Routine surveillance should be continued year-round at all appropriate stations, even if waste application is seasonal or otherwise periodic, to monitor time-lag quality effects upon all water resources. Periodic review of the data generated may indicate areas for adjustments to the monitoring program and will insure that the program remains responsive to the needs of the facility.

PROJECTED PROCEDURAL AND MATERIEL REQUIREMENTS

Sample Collection and Measurement Effort. It is recommended that all samples obtained for analysis be grab samples, rather than composite samples, with the exception of the initial wastewater characterization phase. The use of composite sampling techniques is inappropriate at many sampling points because of station remoteness, power and manpower considerations, and the damping (equalizing) effect of land treatment on constituent concentrations. All samples should be collected in clean, clearly labeled, plastic containers and glass bottles. The use of a truck or trailer-mounted, submersible, low capacity pump, designed to swing off the side of the mount, could be used to obtain samples from all wells.

Three of the recommended parameters lend themselves to field measurement rather than laboratory analysis. Continuous flow records at the pretreatment facility may be obtained directly by utilizing a combination level reading and recording device. Elsewhere, a Parshall flume or similar device may be employed to permit relatively accurate mass flow calculations. pH may be measured onsite using a portable, direct-reading pH meter with internal compensation for temperature variations. The static water level in each well should be routinely measured using a portable, transistorized water level indicator.

Analytical Effort. Immediately following sample collection (and filtration, if appropriate) each sample should be appropriately preserved to retard the chemical and biological changes that inevitably ensue after the sample is removed from the parent source. Further steps in proper sample preparation may involve procedures such as dilution, incubation, addition of chemicals, indicator solutions, etc., to permit relatively accurate, precise, and consistent subsequent analytical determinations. The recommended methodology for the actual performance of analyses is contained in Standard Methods, 14th ed.[3] The analytical skill requirements will vary from parameter to parameter and among sample preparation, actual sample analysis, and analytical evaluation tasks. Duplicate samples and standardized solutions should be processed along with the principal workload to verify the precision and accuracy of the analytical techniques employed. This quality assurance workload generally amounts to about 20 percent of the basic workload.

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Table 1. Guide to Parameter Selection for Initial Effluent Characterization

Health-Significant Parameters:

Arsenic*	Lead*
Barium*	Mercury*
Cadmium*	Nitrogen, Nitrate/Nitrite
Chromium*	Selenium*
Fluoride*	Silver*

Indicator and Operational Management Parameters:

Biochemical Oxygen Demand	Nitrogen, Total Kjeldahl
Boron*	Oil and Grease
Calcium*	pH
Chemical Oxygen Demand	Phosphorus, Total Phosphate
Coliform, Fecal	Sodium*
Copper*	Solids, Dissolved
Magnesium*	Solids, Suspended
Nitrogen, Ammonia	Specific Conductance

* Analyzed from sample which has been passed through 0.45 micron filter.

Table 2. Guide to Parameter Selection for the Operational and Compliance Monitoring Program

Parameters	Pretreated Effluent	Surface Water Sampling Stations*	Ground-water Sampling Stations
Biochemical Oxygen Demand	X		
Chemical Oxygen Demand	X	X	X
Chlorine Residual	X		
Coliform, Fecal	X	X	X
Flow	X	X	
Nitrogen, Nitrate/Nitrite	X	X	X
pH	X	X	X
Phosphorus, Total Phosphate	X	X	X
Solids, Suspended	X		
Static Water Level			X

* If collected renovated effluent is discharged directly to a surface water body, the specific parameters for routine surveillance will be stated in the NPDES permit.

EXPLANATION:

PE - Sampling station for pretreated effluent
CRW- Sampling station for collected renovated wastewater
BW- Baseline well
OW- Onsite well
PW- Perimeter well

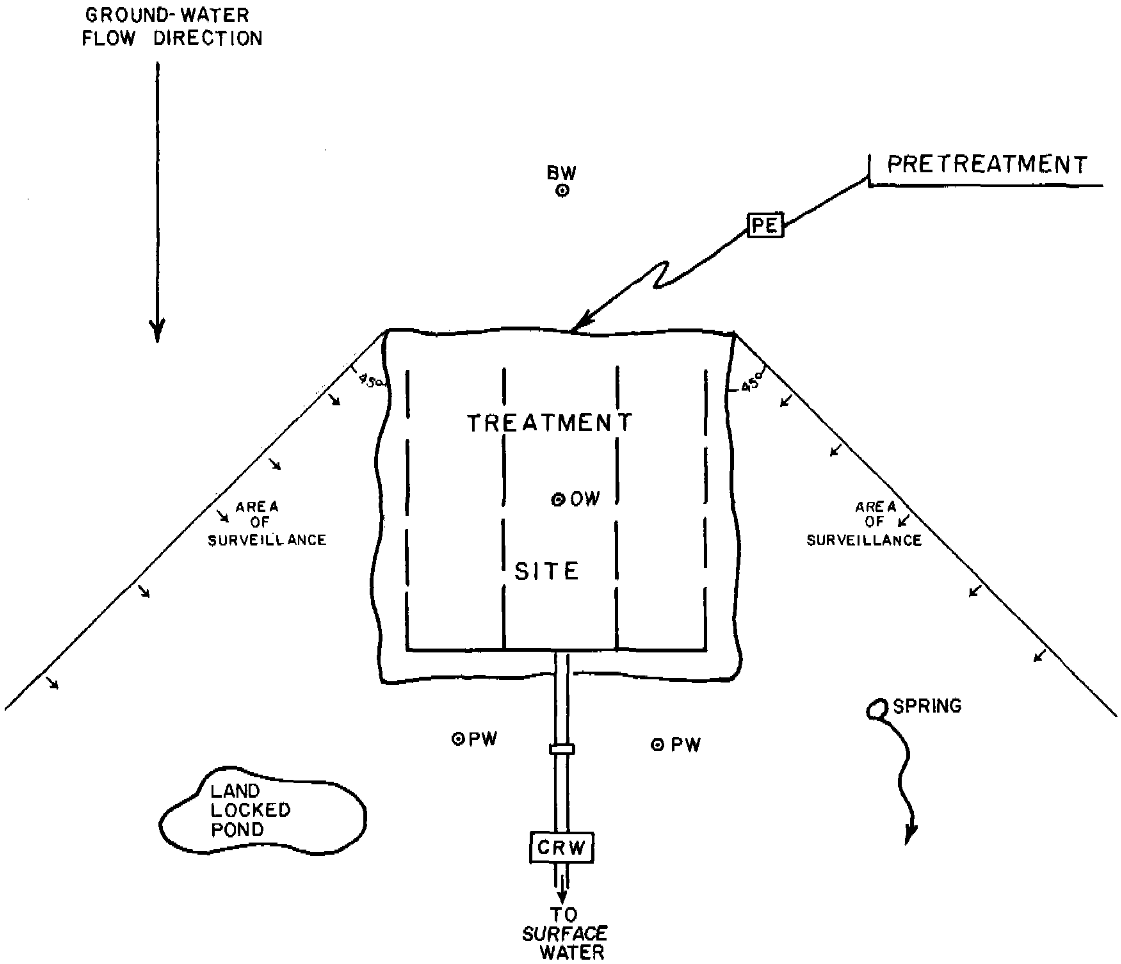

Figure 1. Representative Sampling Stations

MONITORING PROCEDURES

SEPTIC LEACHATE DETECTION - A TECHNOLOGICAL BREAKTHROUGH FOR SHORELINE ON-LOT SYSTEM PERFORMANCE EVALUATION

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ABSTRACT

A comparator fluorometer has been designed to fingerprint wastewater effluent discharges and to monitor concentrations along shoreline areas. Using a stable ratio between fluorescent organics commonly found in effluent and inorganics as conductivity, the instrument is calibrated against a standard effluent. Mounted in a boat, the system draws water from the nearshore lake bottom through the instrument which detects and profiles any plumes of effluent-like substances emerging from lake sediments. With the ENDECO[R] Type 2100 Septic Leachate Detector, the SEPTIC SNOOPER[TM] System, the operator is now able to continuously scan an extensive shoreline at walking pace and spacially locate areas of concern. Discrete laboratory samples can be taken on location for bacterial and nutrient analysis. Expensive laboratory time for normal random taking of samples is substantially reduced and redirected towards identified problem areas.

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The movement of nutrients from lake sediments into overlying waters is often viewed as a passive process rather than an active emergence of groundwater plumes from on site disposal units (Welch et.al. 1977). Huff et.al. (1977) found that 65 percent of the dissolved mineral nitrogen entering Lake Wingra came from subsurface springs and groundwater seepage. The contribution of nutrients from subsurface discharges of shoreline septic units has been estimated at 30 to 60 percent of the total nutrient load in certain New Hampshire lakes (LRPC, 1977).

The lake shoreline is a particularly sensitive area since: 1) the groundwater depth is shallow, encouraging soil water saturation and anaerobic conditions; 2) septic units and leaching fields are frequently located close to the waters edge, allowing only a short distance for bacterial degradation and soil adsorption of potential contaminants; and 3) the recreation attractiveness of the lakeshore often causes temporary overcrowding of homes leading to hydraulically overloaded septic systems. Groundwater inflows frequently convey wastewaters from nearshore units through bottom sediments and into lake waters, resulting in macrophyte growth and algal blooms, particularly in porous soils found on glacial outwash plains (Figure 1).

The ENDECO Type 2100 Septic Leachate Detector, combined with static leachate collectors, can form effective tools with which regulatory agencies can monitor the condition of shoreline septic systems. Profiling of the shoreline with the SEPTIC SNOOPER System indicates the position of malfunctioning septic systems (Figure 2). Installation of static leachate collectors can quantify the rate of concentration of effluent flowing out through located subsurface plumes.

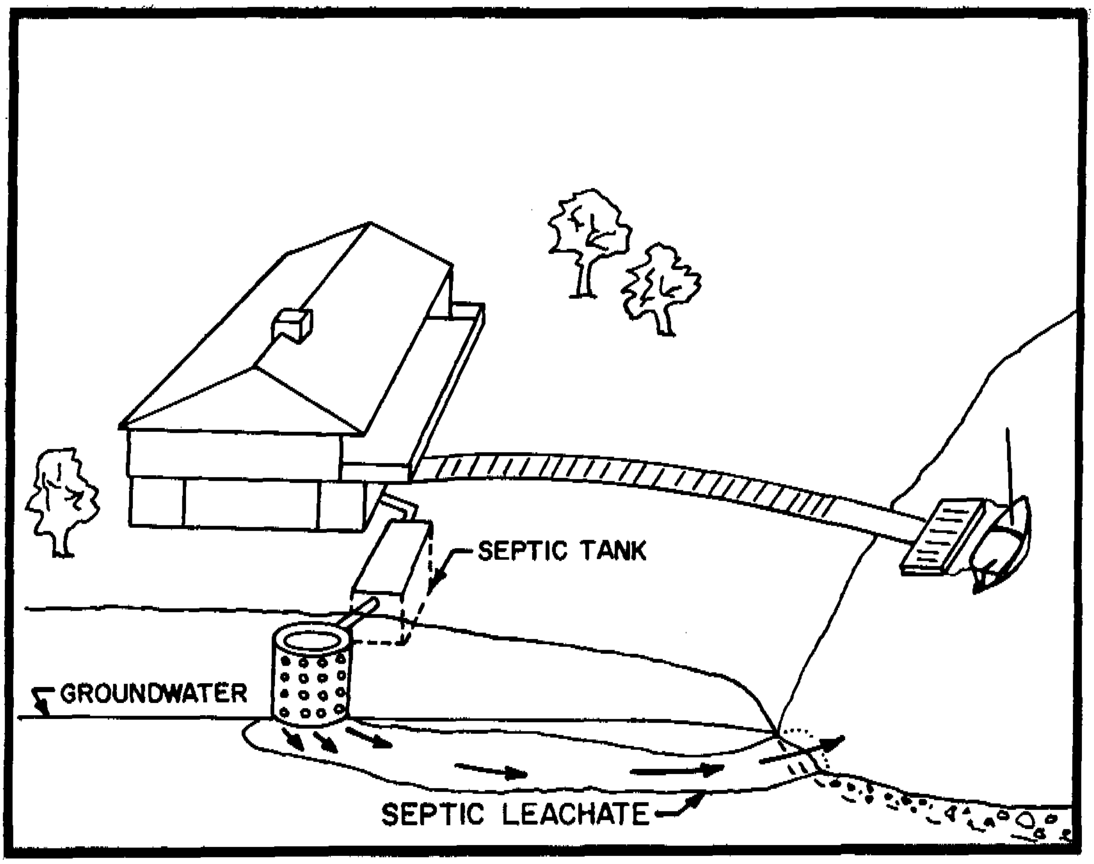


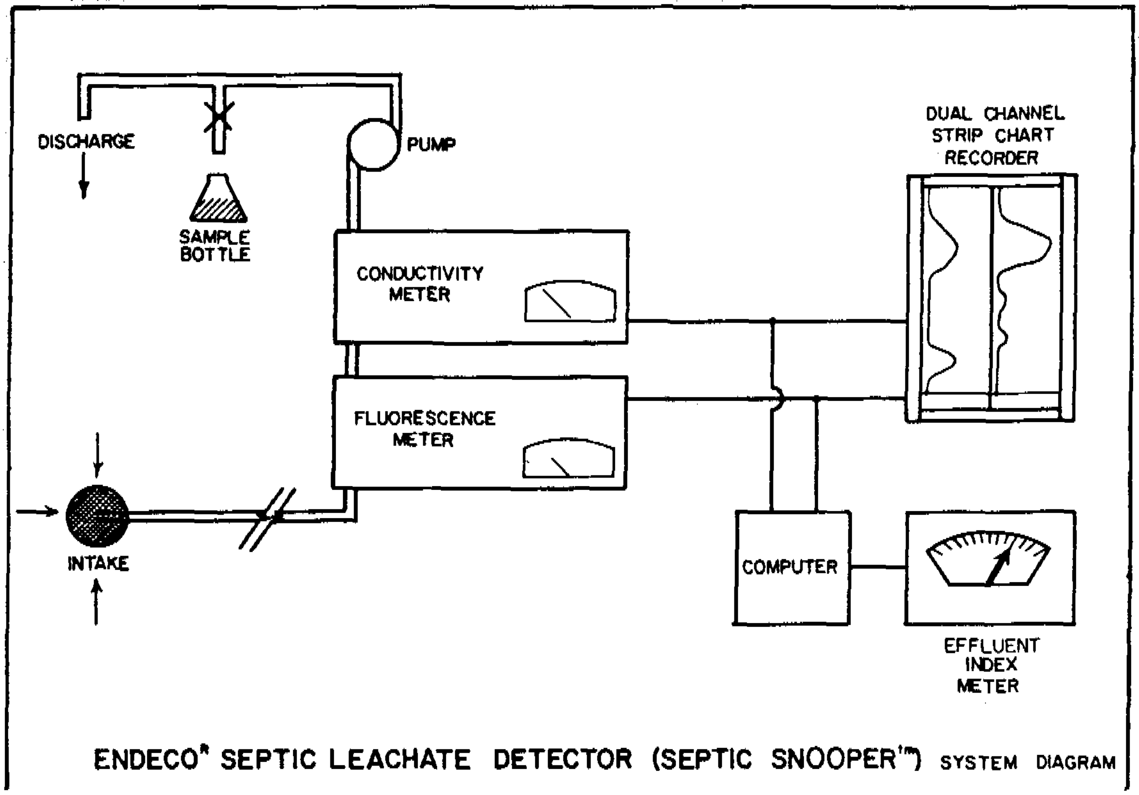
Figure 1. Excessive Loading of Septic System on Porous Soils Causes Plumes of Poorly-treated Effluent Which Move with Groundwater Flow and may Discharge into Nearby Lakes.

Septic Leachate Detector Theory of Operation and System Design

Investigations of wastewater effluent deposited on sand filter beds at Otis Air Force Base, Cape Cod, Massachusetts showed persistent fluorescent organics at distances greater than 100 meters from disposal areas (Kerfoot et.al. 1975). Wastewater effluent contains a mixture of near UV fluorescent organics derived from whiteners, surfactants and natural degradation products which are persistent under the combined conditions of low oxygen and darkness. Figure 3 shows two samples of sand-filtered effluent from the Otis Air Force Base Sewage Treatment Plant. One was analyzed immediately and the other after having sat in a darkened bottle for six months at 20°C. Note that no decrease in fluorescence was apparent, although during the aging process some narrowing of the fluorescent region did occur.

Aged effluent percolating through sandy loam soil under anaerobic conditions reaches a stable ratio between the organic content and chlorides which are highly mobile anions. The stable ratio (cojoint signal) between fluorescence and conductivity allows ready detection of leachate plumes by their conservative tracers as an early warning of potential nutrient breakthroughs or public health problems.

The Septic Leachate Detector consists of the subsurface probe, the water intake system, the analyzer control unit, and the graphic recorder (Figure 2). Initially the unit is calibrated against stepwise increases of a wastewater effluent, of the type to be detected, added to the background lake water. The probe of the unit is then placed in the lake water along the shoreline. Groundwater seeping through the shoreline bottom is drawn into the subsurface intake of the probe and travels upwards to the analyzer unit. As it passes through the analyzer, separate conductivity and specific fluorescence signals are



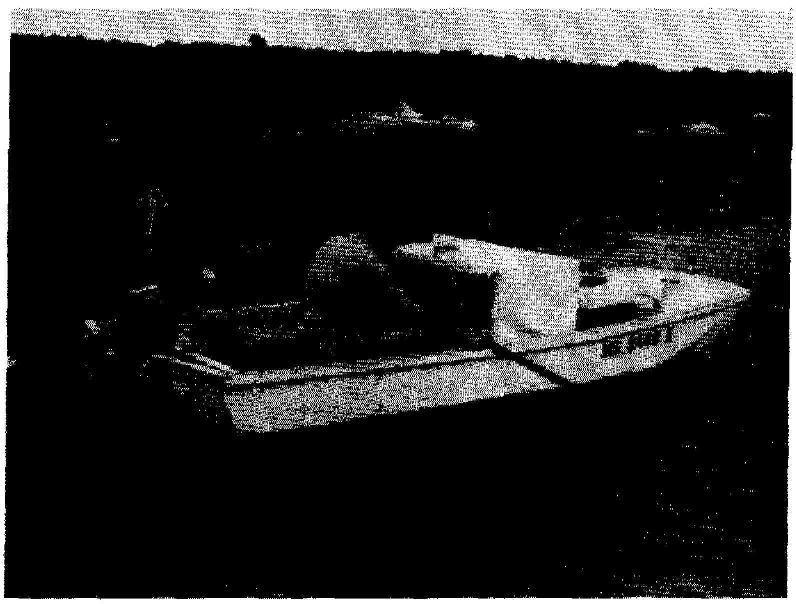


Figure 2. The ENDECO Type 2100 SEPTIC SNOOPER Consists of Combined Fluorometer/Conductivity Units Whose Signals are Adjusted to "Fingerprint" Effluent. The Unit is Mounted in a Boat and Piloted Along the Shoreline. Here the Probe is Shown in the Water with a Sample Being Taken at the Discharge of the Unit for Later Detailed Analysis.

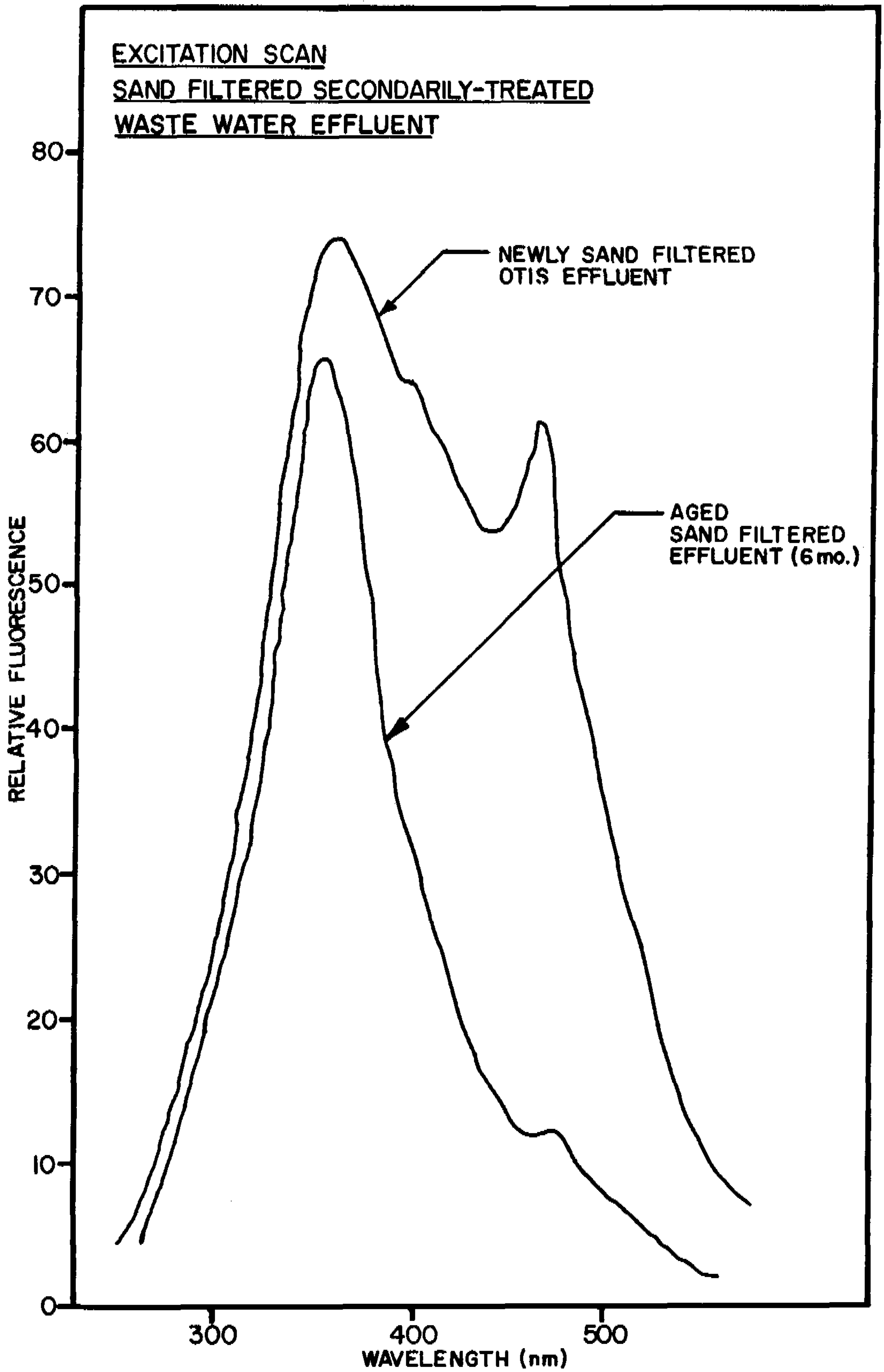


Figure 3. Sand-filtered Effluent Produces a Stable Fluorescent Signature, Here Shown Before and After Aging.

generated and sent to the analog computer circuit which compares the signal against the standard to which the unit was calibrated. The resultant signal, as a percentage of the standard, is continuously documented on a stip chart recorder as the boat moves forward. The analyzed water is continuously discharged from the unit back into the receiving water. Wherever discharges of wastewater plumes are located, water samples are taken at the peak and analyzed by EPA Standard Methods for chemical content of the plume. The unit is powered by a 12 volt automobile battery or a portable generator.

FIELD CAPABILITIES OF SYSTEM

The ENDECO Type 2100 Septic Leachate Detector has three important uses: a) performing rapid sanitary surveys, b) identifying precise sources of nutrients and c) quantifying subsurface nutrient inflows for lake management. The following examples illustrate how the Leachate Detector has provided insight into septic system impact on shorelines, information which would not have been possible with traditional discrete water samplings used alone.

Sanitary Surveys

Some shoreline situations may be marginally suitable for septic systems (Holzer, 1975). The soils may be too shallow, have too high a water table, have coarse gravel, fissure cracks, or an impermeable layer (called "fragipan"). Erosion along beach areas may actually break into boundaries of older drainage fields. The types of malfunctions fall into several categories: 1) subsurface hydraulic overloads in porous soils, resulting in a plume of partially treated wastewater emerging from the bottom sediments into the lake; 2) conduiting - where effluent "short-circuits" through subsurface cracks or highly permeable gravel layers and enters the lake either above or below the water surface; and 3) overflow of septic systems yielding surface runoff of wastewater into tributaries feeding a lake or directly into lake perimeters.

For example, Figure 4 depicts the approximate size of discharges observed by the SEPTIC SNOOPER System along the shoreline of a cove in a New Hampshire lake. The lower two lines present the bacterial and nutrient concentrations observed in water quality samples taken for background locations and at the centers of maximum observed discharges. The concentrations of toal coliform (dark) and fecal streptococcus bacterial (light) are indicated by the height of the histograms. The horizontal dotted line represents a concentration of 1000 counts/100 mL, the present public health safety criteria for recreational-use waters. The discharges of Stations 5, 6, 7 and 8B of Piper Cove exceeded the recreational level. All observed discharges of septic leachate that were sampled, as shown in the lowest line, revealed nitrogen contents significantly above background levels. The total nitrogen and total phosphorus contents of water samples are also plotted as histograms. Total phosphorus values were noticeably elevated at Stations 5 and 7 of Piper Cove.

Sources of Nutrients

Spinnaker Cove is a closed loop canal dredged out of marshlands in sandy soil in the Town of Mashpee, Massachusetts. In 1977, Environmental Devices Corporation conducted a survey to establish that elevated nutrient contents observed in the canal resulted from septic leachate and not drainage from the central peat marsh (EMI, 1975).

A shallow topsoil, consisting of an overburden of mixed medium sand and peat, rested atop a strata of peat, varying from .6 to 2 meters below the surface, which was followed by light medium to fine sand. From accumulation of groundwater in the canal at steady state, the flushing time of the canal was estimated to range between 8.6 and 12.4 tidal cycles, indicating a groundwater infiltration rate of about 3.68 cubic meters per 30.3 meters distance per tidal cycle.

Surveys of the shoreline indicated a series of discharges from individual lots, with the largest plumes observed in the region of least flushing (Figure 5). The elevations of ammonia-nitrogen (NH_4-N) levels corresponded to the location of the effluent discharges, clearly identifying the nitrogen as originating from septic system leachate and not decaying organic matter from the centrally-located marsh. A highly

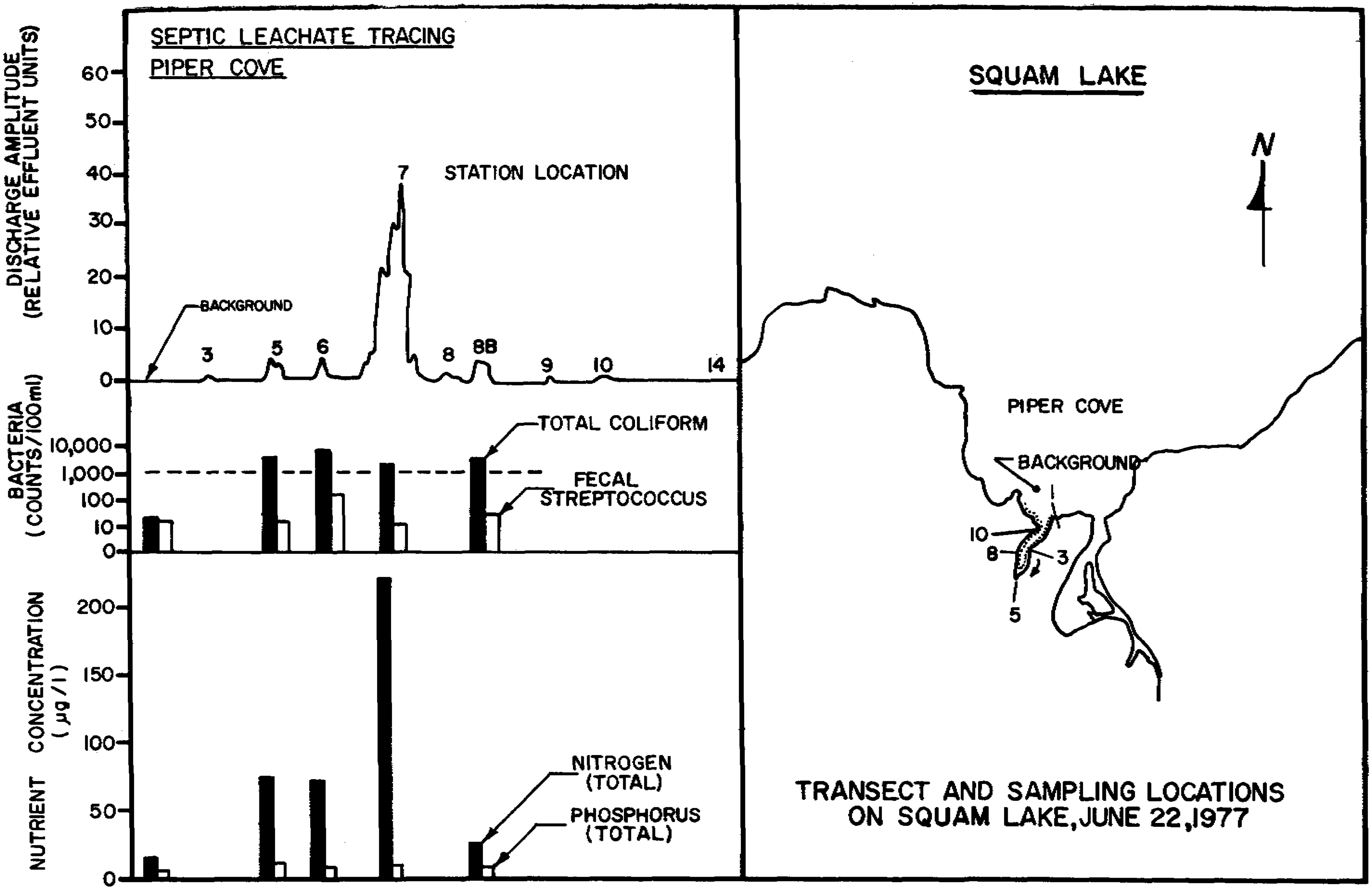


Figure 4. A Typical Survey Shows Results of Bacterial and Nutrient Analyses Taken at Observed Peak Concentrations.

significant correlation (r^2 = +.94) existed between the fluorescent signal from the SEPTIC SNOOPER System and the NH4-N content from water samples taken at peak concentration (Figure 5).

Quantifying Subsurface Nutrient Inflows

To evaluate the extent of nutrient loading under different soil conditions, SEPTIC SNOOPER surveys were conducted along different shorelines of Lake Winnipesaukee for the Lakes Region Planning Commission in New Hampshire. The data was utilized in the Lakes Region Area-wide Waste Water Management Program (208 Program) for the purpose of analyzing septic system nutrient retention and effective life of the units. Nitrogen and phosphorus ; penetration of the soil and lake bottom were measured directly at locations of observed plumes of substantial size and from leachate collected in static seepage collection meters.

The Leachate Detector was used to scan shoreline beaches on Lake Winona. When a small plume was detected, after a water sample was obtained for later analysis, a static seepage collection meter was placed at the peak of the observed plume (Figure 6).

The seepage meter design was based on a sampling device designed by David R. Lee of Waterloo University which consists essentially of a meter drum seepage cylinder sunk into the sediment near the shoreline and vented to a 4 liter plastic bag. The drum

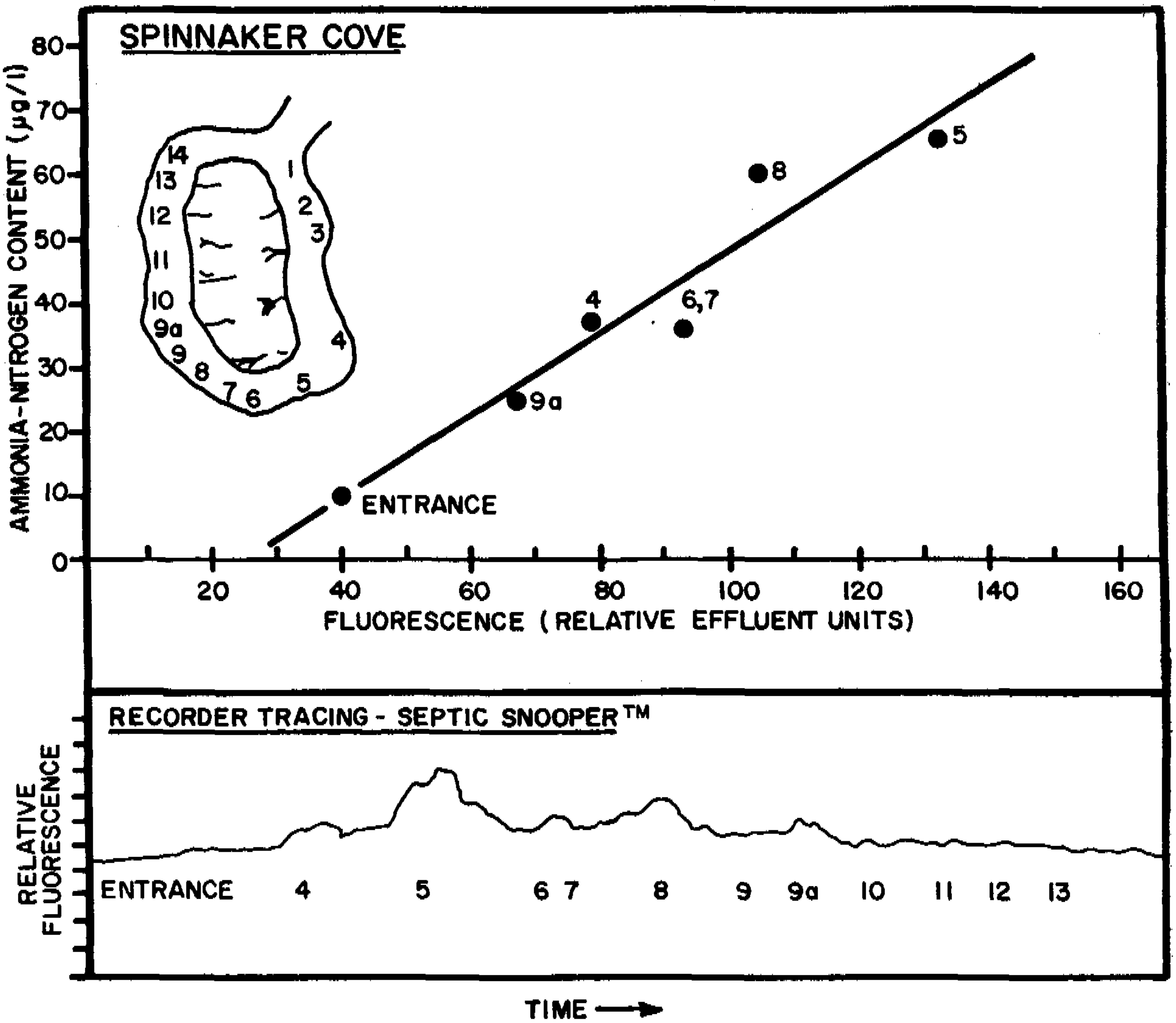
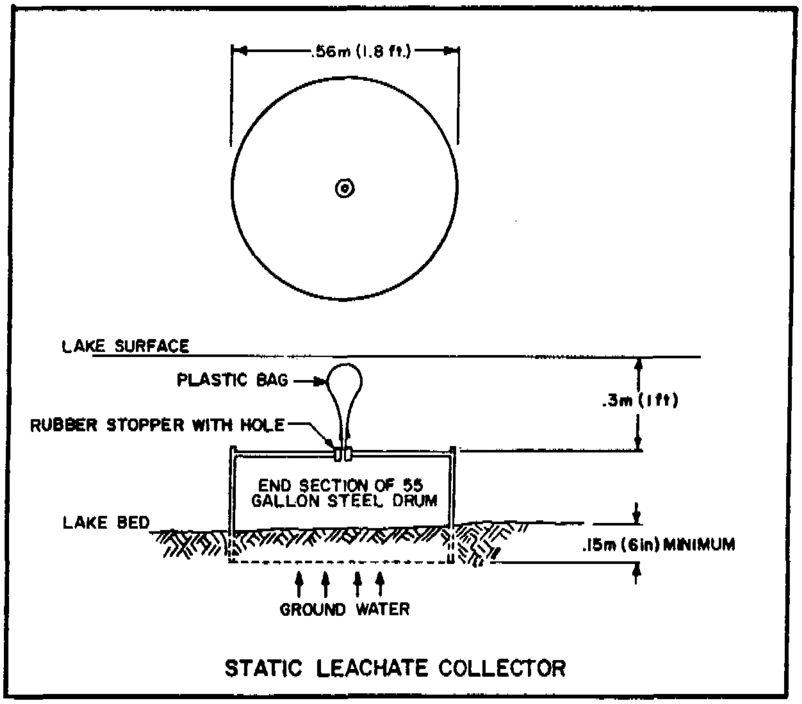


Figure 5. Septic Leachate Plumes and their Correlation with Amonia-Nitrogen Contents of Water Samples Taken at Peak Concentration.



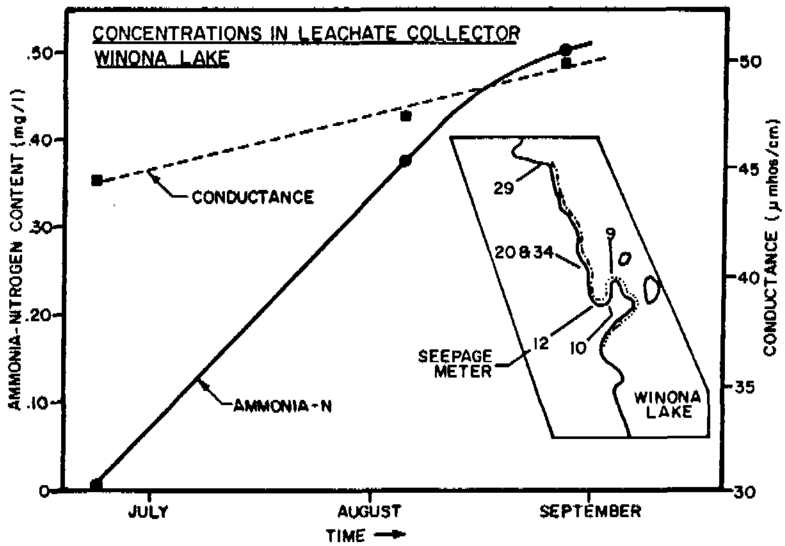


Figure 6. The Static Septic Leachate Collector and the Observed Increase in Ammonia-Nitrogen and Conductivity in the Isolated Discharge of a Small Plume at Winona Lake.

restricts the normal mixing of lake water with recharged groundwater, eventually filling completely with the recharged water. Groundwater flowing in through the bottom slowly replaces the original water in the drum until the internal concentration of constituents mirrors that of the recharging groundwater (Table 1).

Table 1. Increase in Constituents in Groundwater Static Seepage Collector Installed at Winona Lake on Small Shoreline Plume

Constituent (mg/L-ppm)	June 22nd	August 4th	August 25th
Ortho-P	.0003	.0016	.0050
Total-P	.0155	.0189	.0124
Nitrate-N	.0020	.0246	.0170
Ammonia-N	.0020	.3822	.5128
Conductivity*	44.1	47.2	49.7

*Conductivity in μmhos/cm^2

The nutrient concentration was monitored at three-week intervals to show the rate of replacement and composition. Typical effluent exhibits a rough conductivity:total phosphorus: total nitrogen ratio of 200:8:20. Considering this basic ratio, one would expect to find a total nitrogen (N) content of .560 ppm in seepage containing a conductance increase of 5.6 μmhos above background. Based upon the observed increase in conductance, an effluent concentration of 2.8 percent should have been present. To compare with this, a portion of the seepage was removed from the meter then diluted and circulated through the SEPTIC SNOOPER. The sample showed an effluent concentration from 1.0-2.4 percent equivalence to a standard Otis effluent which was close to the anticipated value.

CONCLUSION

The ENDECO Type 2100 Septic Leachate Detector, the SEPTIC SNOOPER System, combined with static leachate collectors form effective tools with which regulatory agencies can monitor the condition of shoreline septic units. By moving at a near constant speed along shorelines, the device can map the position and approximate magnitude of underground plumes emerging into the lake waters from hydraulically overloaded or "short circuited" systems. Discrete water samples can then be taken efficiently and economically for nutrient and bacterial analyses.

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COST ESTIMATION

COMPUTER PROCEDURE FOR COMPARISON OF LAND TREATMENT AND CONVENTIONAL TREATMENT: PRELIMINARY DESIGNS, COST ANALYSIS AND EFFLUENT QUALITY PREDICTIONS

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ABSTRACT

During 1972 a manual for the design of wastewater treatment facilities was developed by the U.S. Army Engineer Waterways Experiment Station. To complement the design manual and assist the field design engineer, the computer model CAPDET (Computer Assisted Procedure for the Design and Evaluation of Wastewater Treatment Systems) was developed. In response to field users' requests for design information on land treatment, a land treatment module was developed and implemented into CAPDET. The CAPDET program provides planning level design and cost evaluations for any wastewater treatment system.

There are five basic land treatment system alternatives available for use in the land treatment module of CAPDET: rapid infiltration, overland flow, spray irrigation, ditch irrigation and flood irrigation. Unit process design data must be included with each land treatment system. These data include: application rate, width of buffer zone, field application period, wastewater generation period, cost of land, period of storage, site preparation, distribution pumping and recovery system. Other design features (i.e., lining for storage reservoirs) can be included, if needed. Also, wastewater characteristics (BOD_5, BOD_5 soluble, COD, COD soluble, maximum flow, average flow, minimum flow, TKN, NO_3, NH_3, PO_4, cations, anions, settleable solids, suspended solids, volatile solids, oil and grease, pH and temperature) and the cost analysis data (interest rate, plant life, construction cost index, supply cost index, wage rate and inflation index) are necessary input parameters.

The land treatment module of CAPDET is designed to generate three major types of output: total land requirements, the percolate water quality summary and costs (capital, average annual, and operation and maintenance). The Stage II cost estimating routines from the 1975 Environmental Protection Agency reference entitled "Costs of Wastewater Treatment by Land Application" served as the principal data source for the model [1].

A land treatment system design would be upgraded to the level that construction details are well enough defined so that an adequate estimate of quantity of construction materials, man-hours of labor, etc., which are necessary to build and operate a land treatment system, can be made. These variables are called cost elements. Realizing the enormous number of cost elements utilized in plant construction, a "modified" cost element approach is being developed. Other major emphases in updating the land treatment module of CAPDET are on the computation of the required land treatment application area based on nitrogen requirements, crop uptake and revenue option, and the percolate water quality predictions.

CAPDET can be used as a planning tool to estimate the costs and designs of land treatment systems and as a screening tool to compare a wide range of alternative land treatment designs quickly with a common economic design base, each capable of meeting specified percolate water quality criteria. The model can simultaneously rank alternatives on the basis of cost-effectiveness. In addition, schemes for alternative conventional treatments can be input and ranked against land treatment to compare costs and treatment efficiency.

BACKGROUND: CORPS WASTEWATER MANAGEMENT PROGRAM

The passage of the Federal Water Pollution Control Act in 1972 (Public Law 92-500) gave the U.S. Army Corps of Engineers new direction in their water resource planning missions. The Corps became involved in the effort to define and solve water pollution problems early in 1972. At this time the Corps undertook the study of five large metropolitan areas in an effort to develop a set of long-range alternatives for treating wastewater in these cities. These studies eventually led to the "Urban Studies Program," a program to develop long-range plans for administering and managing the total water resources of a major area. Wastewater management continues to be a major study item within the Urban Studies Program. At the same time, the Corps realized its own recreation sanitary facilities and military installations required wastewater treatment upgrades. The Office, Chief of Engineers (OCE) was then required to provide the sanitary engineers in field offices guidance for planning and design of treatment systems within the domains of military installations, recreation areas and Urban Studies.

CORPS WASTEWATER MANAGEMENT GUIDANCE: THE CAPDET MODEL

During 1972, in response to Public Law 92-500, a series of manuals to provide guidance to the field offices for wastewater treatment facility design and cost estimating were developed [2]. To complement the design manuals and to assist the field engineer, the computer model CAPDET (Computer Assisted Procedure for the Design and Evaluation of Wastewater Treatment Systems) was developed and placed in use in 1974. The objective of the CAPDET program is to provide a screening tool capable of designing and evaluating a large number of alternative wastewater treatment systems, each capable of meeting the specified effluent criteria, and simultaneously ranking these alternatives on the basis of their cost effectiveness. The program has since had wide application, not only within the Corps, but also in other Federal and state agencies, universities, and private consulting firms.

THE CAPDET SYSTEM

The computer format of the CAPDET model follows a plan formulation similar to that used by sanitary engineers when considering alternative treatment schemes. CAPDET is coded in the English language to the greatest extent possible, to increase the usability of the program. The program can be run by planners and engineers with a minimum of instruction. The wastewater is described by the user specifying the concentration or magnitude of any or all 20 wastewater characteristics: BOD_5, soluble BOD_5, COD, soluble COD, settleable solids, suspended solids, volatile solids, TKN, NO_2, NO_3, NH_3, PO_4, maximum flow, minimum flow, average flow, cations, anions, pH, temperature, and oil and grease. The alternative treatment methods are input by the planner using engineering judgement to select among the 50 unit processes available in CAPDET. The wastewater treatment design schemes may be physical, chemical, biological, land treatment or a combination of these, and can be arranged to meet primary, secondary or tertiary treatment levels. The user may also set effluent quality limits on any of the 20 waste descriptors, thereby screening out plants that do not satisfy treatment criteria. The 50 unit processes have been programmed using standard sanitary engineering design formulations and by using average values for the design parameters in a "default" data deck. The "default" data are easily accessible to the user, thus placing the user in control of the final designs. Cost data must also be supplied by the user to update the cost curves contained in the computer program. The cost analysis parameters include the interest rate, life of the treatment plant, construction cost index, supply cost index, wage rate and the inflation index.

The CAPDET computer program forms all the possible treatment schemes from the alternatives the user supplies. The computer program processes the alternatives and performs a cost analysis on each unit process. The output results in a ranking of the 100 most cost-effective schemes which meet the stated water quality criteria. Cost information (capital, operation and maintenance, and average annual) for each plant and each unit process within each plant, the preliminary designs and the water quality predictions for each process, are given. Figure 1 shows a schematic of inputs, CAPDET program elements and output. The program is designed to be run numerous times with a logical sequence of review and fine-tuning of promising designs.

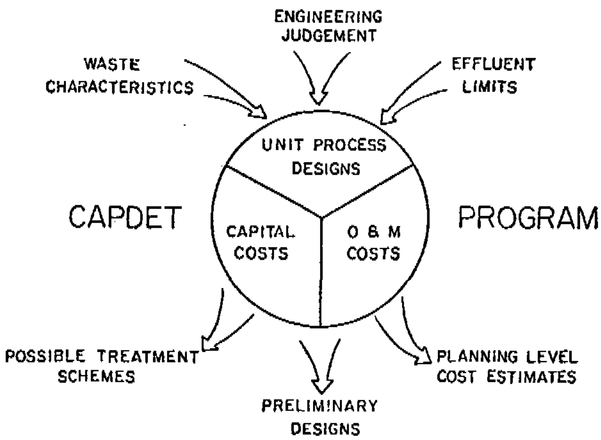

Figure 1. CAPDET program schematic.

CORPS LAND TREATMENT RESPONSIBILITIES

The Federal Water Pollution Control Act Amendment of 1972 (Public Law 92-500) has required consideration of wastewater reclamation and recycling by land treatment as an alternative to conventional unit processes for the treatment of municipal wastewater. This applied to all Environmental Protection Agency (EPA) 208 Area-wide Studies and 201 Facility Planning Programs, the Corps of Engineers Urban Studies Program, Corps recreational facilities, and the planning and design of Corps-constructed facilities. By Federal regulation, all military bases must consider land treatment as an alternative to conventional waste treatment. As the construction agent of the Army, the Corps of Engineers has the task of meeting the requirements of Public Law 92-500 and conducting research where needed.

THE LAND TREATMENT MODULE OF CAPDET

During 1975, in response to field users' requests, a cooperative effort between the U.S. Army Cold Regions Research and Engineering Laboratory (CRREL) and the Environmental Laboratory (EL) of the U.S. Army Engineer Waterways Experiment Station (WES) was initiated to include a land treatment module in the CAPDET system. The land treatment module was developed and implemented into the CAPDET system by the time of the first Corps land treatment training course in December 1975. There are five basic land treatment system alternatives in the land treatment module of CAPDET: rapid infiltration, overland flow, spray irrigation, ditch irrigation and flood irrigation. These various modes of land application of wastewater can be utilized with any conventional treatment scheme. Often land treatment is used to upgrade the treatment level at an existing system. An example of a land treatment scheme is shown in Figure 2 as the conventional sanitary engineering block diagram, and is translated in Figure 3 to CAPDET format.

This example input, if processed through the CAPDET system, would provide designs, costs and effluent quality predictions for 15 different treatment plants. The scheme describes preliminary treatment (bar screening, grit chamber, comminution), followed by one of three types of lagoons (aerated, facultative aerated, stabilization pond), followed by chlorination, transmission lines to the field, and then one of the five modes of land application. These 15 plants would be ranked according to average annual cost. Schemes with conventional biological, chemical and physical unit processes can also be input and ranked against land treatment schemes to compare costs and treatment efficiency.

The land treatment module is processed through CAPDET in the same manner as the conventional treatment processes. However, there are several basic differences in the requirements for inputs to

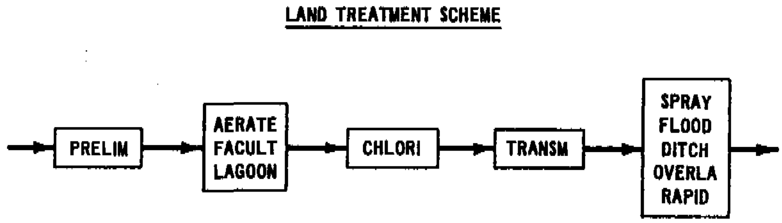

Figure 2. Block diagram.

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TITLE LAND TREATMENT SCHEME
LIQUID LINE
BLOCK     PRELIM
BLOCK     AERATE FACULT LAGOON
BLOCK     CHLORI
BLOCK     TRANSM
BLOCK     SPRAY FLOOD DITCH OVERLA RAPID
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Figure 3. CAPDET format.

the land treatment module. Land treatment systems are extremely site-specific. There are many factors in the planning stage which must be taken into account, but are not quantifiable or easily adapted to computer procedures. The designer must make a number of decisions prior to utilizing the land treatment module of CAPDET for cost estimates and treatment performance comparisons. A major input to the land treatment module is the application rate. Therefore, the designer must have the site selected and be familiar with the soils, geology, climate and land use characteristics so that an application rate can be determined. Other data inputs required by the land treatment module of CAPDET which need to be determined from the site-specific information are field application period, width of buffer zones, wastewater generation period, period of storage, site preparation needs, cost of land, distribution pumping and the recovery system for renovated water. Other design features (i.e., lining for storage reservoirs) may be specified, if needed, as design options for cost adjustments.

The land treatment module is designed to generate three major types of output: total land requirements, the percolate water quality summary and costs (capital, average annual, and operation and maintenance). The input data described above are processed through CAPDET as shown in Figure 4. Cost components of land treatment systems include: application systems,

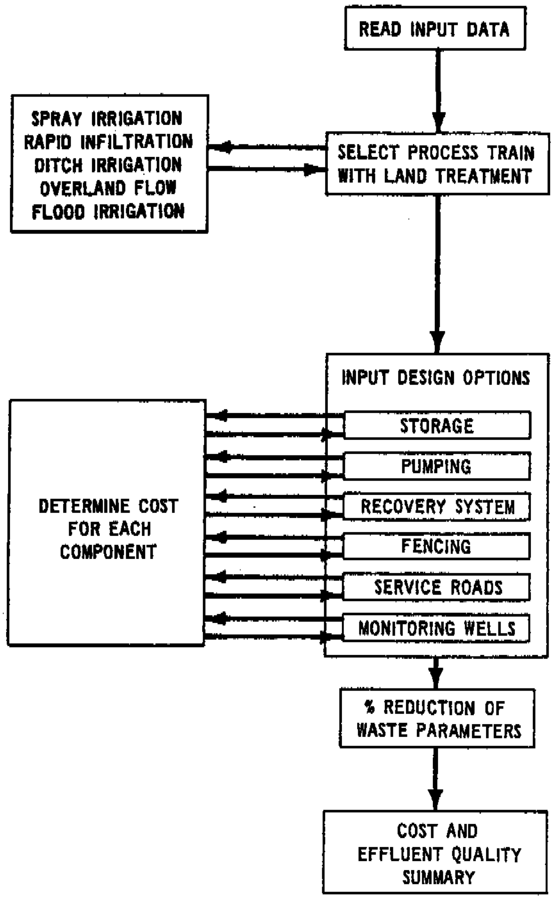

Figure 4. Land treatment process module.

renovated water recovery systems, distribution systems, storage areas, field preparation, monitoring systems, administration and laboratory facilities, service roads and fencing. Each component can represent a significant capital expenditure or yearly operating expense in a land treatment system. The CAPDET program generates costs for each component. The principal data source for the cost data base in the land treatment module is taken from the Stage

II cost estimating routines in the 1975 EPA reference entitled "Costs of Wastewater Treatment by Land Application" [1]. Treated water quality predictions are taken from the 1975 EPA reference entitled "Evaluation of Land Application Systems" [3].

FUTURE CAPDET DEVELOPMENTS

OCE and EPA are sponsoring, through an interagency agreement, the development of a new cost estimating procedure for the CAPDET program. The criteria for development of the new cost system were:

- must be well defined
- must be consistent
- must be updatable
- must be sensitive to local conditions

A decision was made to follow a "cost element" approach to estimate wastewater treatment costs. In this approach, the sanitary engineering designs would be upgraded to the level that construction details would be well enough defined to adequately estimate quantities of materials, man-hours of labor, etc. necessary to build and operate the plant. These quantities are the cost elements. Realizing the enormous number of cost elements utilized in plant construction, a "modified" cost element approach was chosen in which only elements contributing a major input to the total cost would be used. It was felt that a breakdown of unit processes into cost elements on which actual construction is based, with application of their attendant unit prices, would provide sensitivity to field conditions and a consistency between unit processes, and eliminate the need for cost indices.

The cost elements are determined by the construction details, i.e., design features, that describe the unit process. As an example, a capital cost basin design element is reinforced concrete:

<u>Cost Element</u>   <u>Design Feature</u>

$$\text{amount of concrete} = f\left\{\begin{array}{l}\text{diameter}\\ \text{depth}\\ \text{wall thickness}\\ \cdot\\ \cdot\\ \cdot\end{array}\right\}$$

Cost elements are identified for each unit process such that the majority of plant costs, both capital and operating, are defined by this method. The magnitude of each cost element is summed for the total plant and unit prices input to determine the system costs.

The development of this new cost estimating system also includes bringing all treatment process designs up to the state-of-the-art of research. Five of the processes which are encompassed by the new work unit are the land treatment modes. The land treatment module is given high priority in light of recent EPA and Corps needs. A cooperative effort between CRREL and the EL in FY78 includes refinements to the land treatment subroutines in CAPDET.

A subroutine to address the nutrient uptake by crops will be added to the land treatment module of CAPDET. Nutrient uptake values for various crops will be taken from the Process Design Manual for Land Treatment of Municipal Wastewater [4]. The cost analysis for a land treatment system will include the economic return from a cash crop.

Since nitrogen is one of the limiting factors in land treatment, a nitrogen balance subroutine will be added for prediction of the nitrogen in the percolating water from a land treatment system. The equations used in the nitrogen balance are similar to those described in the Manual [4]. The nitrogen balance equation will also be used as an option to determine the required land application area based on nitrogen requirements. The water balance equation stated in the Manual will be used to determine the wastewater hydraulic loading rate [4]. The water balance equations would be developed separately for the slow infiltration, rapid infiltration and overland flow processes.

The equations for prediction of the percolate water quality from the three types of application systems (slow infiltration, rapid infiltration and overland flow) are to be improved using data from current research findings. The main parameters which will be emphasized are NO_3 and PO_4 for infiltration systems and BOD, suspended solids, TKN, NH_3, NO_3 and PO_4 in overland flow systems.

The development of the "modified cost estimating" system is approximately one-half complete. When finished, the CAPDET system will contain 60 unit processes including conventional chemical, physical and biological processes, land treatment processes and processes suitable for flows under 0.4 m^3/s (0.5 million gallons per day). It will be the most comprehensive cost data base available to engineers nationwide. A verification of the cost estimating system is being performed by EPA utilizing data collected in a recent nationwide survey of wastewater treatment plant costs.

ACKNOWLEDGMENTS

The design revisions to the land treatment module of CAPDET are sponsored by the Directorate of Civil Works, Office, Chief of Engineers. The cost estimating system development is sponsored jointly by EPA Water Program Division and the Directorate of Civil Works, Office, Chief of Engineers.

CRREL is the lead laboratory of the Corps of Engineers responsible for basic and applied research in land treatment of wastewater. The EL of WES is the lead laboratory for research in wastewater treatment for water-based disposal and the development of the CAPDET system.

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[2] Department of the Army, Engineering Manual Series. Part I: Design of Major Systems - Wastewater Treatment Facilities Manual; Part II: Design of Small Systems - Wastewater Treatment Facilities Manual; Part III: CAPDET User's Guide, EM 1110-2-501.

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COST ESTIMATION

COMPUTER TECHNIQUES FOR THE ANALYSIS OF LAND TREATMENT SYSTEMS

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ABSTRACT

The paper will describe a computer software package that has been prepared to analyze potential land treatment sites, evaluate their suitability for various types of application, calculate land areas and storage requirements for each suitable type, and summarize the analysis. Data analysis and procedures have been adopted from the U.S. Environmental Protection Agency's Evaluation of Land Application Systems Technical Bulletin.

After reviewing the analysis information, the engineer can then evaluate site characteristics and the surrounding area in regard to the final land treatment system configuration, including storage sites, transmission alignments, as well as application and recovery methods. The information can then be input to a separate cost analysis program. The cost analysis procedures were adopted from the U.S. Environmental Protection Agency's Costs of Wastewater Treatment by Land Application Technical Bulletin and the U.S. Army Corps of Engineers Computer Assisted Procedure for the Design and Evaluation of Wastewater Treatment Systems (CAPDET) Program.

The cost program utilizes cost curves, based on various design parameters calculated in the analysis program. Adjustment factors are available to index labor, material, and construction costs to different areas, as well as having the capability of analyzing the effect of revenues on the project. These revenues may result from crop sales, reclaimed wastewater sales, or land leasebacks.

The program package allows the engineer the opportunity of a detailed preliminary investigation including sensitivity to variables with the economy of computer assisted design.

INTRODUCTION

The study of land treatment systems as viable alternatives to conventional systems was given new life by the passage of Public Law 92-500 in 1972. Section 208 of the law provides for large scale areawide planning and Section 201 provides for specific community or facility plans. The law mandates the evaluation of land treatment systems in all "208" and "201" programs; and the federal government in fact encourages the implementation of land treatment. This encouragement is evidenced in the federal funding policy passed in 1977 (Clean Water Act of 1977). Under this policy, "innovative" treatment systems, which include land treatment, are given 85 percent funding for design and construction as opposed to 75 percent funding for conventional systems. In addition, the law provides that, if the land treatment system fails to perform satisfactorily, the federal government will provide 100 percent funding for the design and construction of a replacement system. A further advantage is given land treatment systems in that they are allowed a 15% advantage in the cost effective analysis.

Our recent involvement with land treatment systems began in 1976 when Boyle Engineering Corporation was contracted by the U.S. Army Corps of

Engineers to develop Land Treatment Alternatives for the Phoenix Urban Study. The Phoenix Urban Study was undertaken to satisfy Section 208 of Public Law 92-500. The study area is centered around the City of Phoenix and includes an area of approximately 5960 square kilometers, (2300 square miles) with an estimated population of around 1.2 million persons. See Plate 1.

In December 1977, a "201" study was initiated which included a sizeable portion of the original Phoenix Urban Study area. This "201" study was initiated largely for administrative reasons; however, it will not effect the final results of the overall study, which will be a coordinated areawide wastewater management plan.

METHODOLOGY

The process of evaluating the land treatment alternatives was a multistep process which began by identifying broad-general areas suitable for land treatment; and ended with a small, refined group of site specific alternatives having staged construction schedules and detailed cost estimates.

The first step in any analysis is to define alternatives. In the case of land treatment, the study area must first be assessed for the location of suitable areas for potential sites. A preliminary evaluation must then be performed to establish the feasibility of using the various land treatment methods at each of the identified sites. Unfeasible sites and alternatives can then be eliminated and a more detailed investigation made of the remaining sites to determine cost effectiveness. The final step is to perform a detailed design with a staged implementation schedule to form a basis of comparison for all the feasible alternatives. The following procedure has been found to be most effective in accomplishing the above described goal.

Preliminary Assessment

The first step in generating the land treatment alternatives was to locate, within the study area, those general regions suitable for the location of land treatment sites.

Each of the land treatment methods-irrigation, infiltration, and overland flow-have specific requirements regarding physical site characteristics. All sites must also reflect considerations of environmental and population settings.

The first general consideration for site location was existing and projected land use patterns and general location in respect to existing or potential wastewater collection points. In the Phoenix Urban Area it was attempted to locate land treatment facilities as far as practical from urban or commercial centers. This was done primarily for aesthetic and public health reasons, such as odor, aerosol drifts, and insect breeding. Projected land use patterns were obtained from the Land Use Plan for Maricopa County, Arizona (prepared by the Maricopa Association of Governments). Fortunately, in the Phoenix area, there are many thousands of acres of agricultural or other open land which lends itself ideally for the location of land treatment sites.

With respect to existing or potential wastewater collection points it was attempted to locate facilities at the downstream end of natural drainage basins, assuming that wastewater collection facilities would generally follow natural drainage patterns. Every attempt was made to locate the land treatment facilities where they could serve as large an area as possible by gravity.

Topography and soils were of great importance in the selection of areas suitable for land treatment sites. The three basic land treatment methods have differing topographic requirements for their selected sites. Irrigation, the most common land treatment method, works most efficiently on level or gradually sloping ground, which allows a maximum amount of water to enter the soil to be utilized by the plants. However, this method can also tolerate steeper slopes of up to 15% provided controls are exercised for runoff and erosion.

The overland flow method by its nature requires a sloping terrain. Slope distance is a function of the spray diameter, loading rate, and degree of renovation desired. For overland flow, the primary requirement is that the existing topography be such that terrace slopes of 2 to 8% can be formed economically. The cost and impact of the earthwork required are the major constraints.

The infiltration-percolation method requires a relatively level site. Most infiltration sites require considerable earthwork to create holding basins and ponds, and often several of these are required to allow periodic

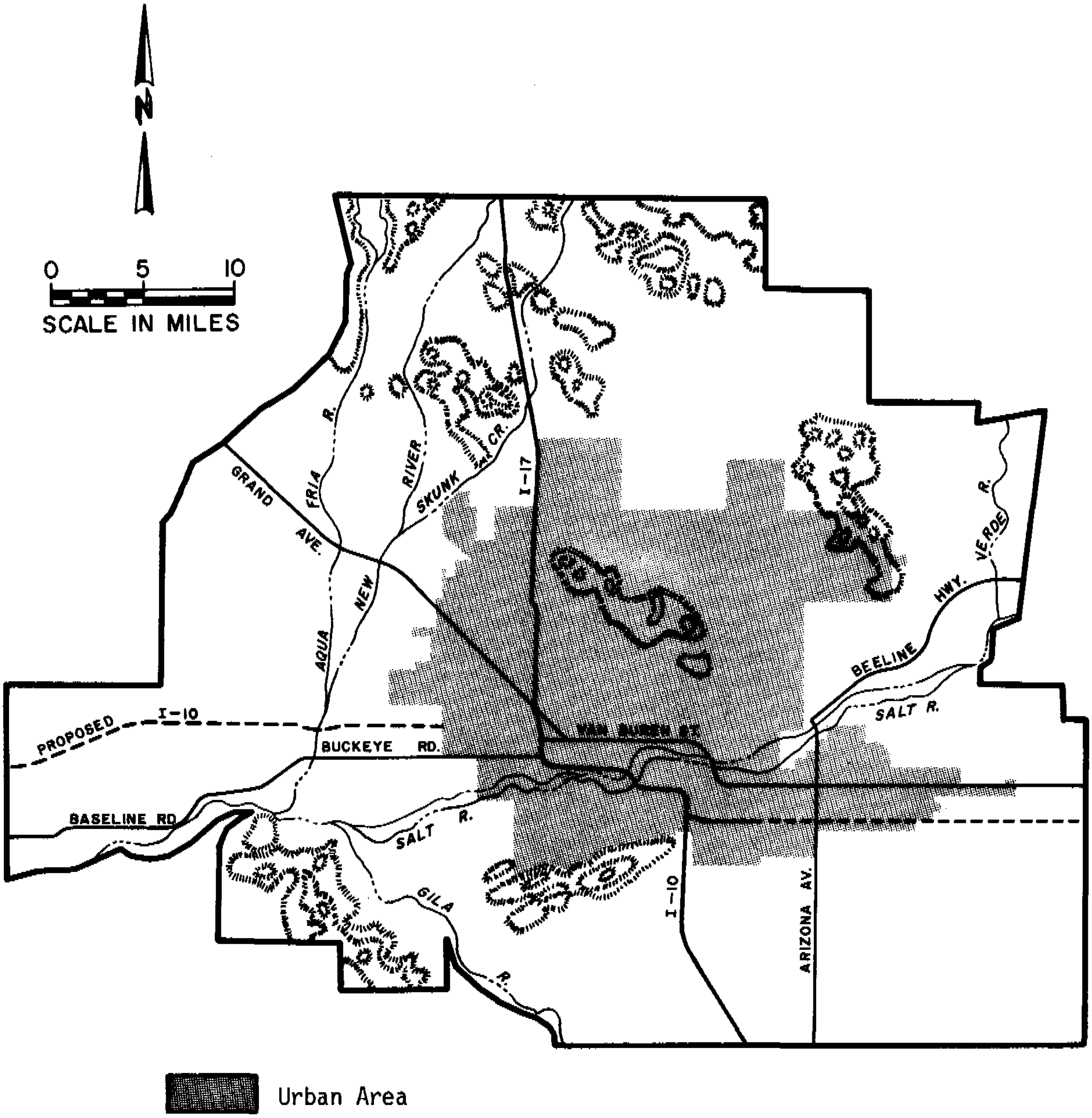


Plate 1 - Phoenix Urban Study Area

draining and cleaning to renovate the site to improve infiltration-percolation rates.

Maricopa County and the Phoenix Urban Study Area contain 14 different soil associations which range from very hot, very dry soils from recent alluvium such as the Gilman-Estrella-Avondale association, to warm, sub-humid soils of the mountain and low hills typified by the Barkerville-Cabezan-Rock Outcrop association. Each of the 14 associations has different technical characteristics and properties such as pH, permeability and corrosivity. The location of these associations, and a tabulation of their various properties was obtained from the General Soil Map of Maricopa County, published by the U.S. Department of Agriculture Soil Conservation Service.

The most important soil property considered was the permeability, which is given by the Soil Conservation Services in ranges for a particular soil.

(<0.15 cm/hr) (<.06 in/hr) very slow
(0.15 - 0.5 cm/hr) (.06 - 0.2 in/hr) slow
(0.5 - 1.5 cm/hr) (0.2 - 0.6 in/hr) moderately slow
(1.5 - 5.0 cm/hr) (0.6 - 2.0 in/hr) moderate
(5.0 - 15.2 cm/hr) (2.0 - 6.0 in/hr) moderately rapid
(15.2 - 50.8 cm/hr) (6.0 - 20 in/hr) rapid
(50.8 + cm/hr) (20 + in/hr) very rapid

The land treatment of municipal wastes can have a significant effect on both groundwater quality and quantity. It can either be a pollutant or a means of improving groundwater quality, depending on the quality of the natural groundwater. In areas of extremely high total dissolved solids content, wastewater can improve this factor enough to make the groundwater suitable for irrigation and other purposes. Conversely, the suitability of high quality groundwater presently pumped in some areas and used for domestic purposes could be lessened by wastewater application, particularly in areas of shallow groundwater.

After reviewing the various physical, environmental, and other criteria, thirteen potential site areas were identified. These areas were used in the next phase of the program to locate specific sites.

Large Array

After general areas suitable for land treatment had been located, specific sites had to be identified. Field investigations and aerial photographs were used extensively in this process. Once specific sites had been found, and wastewater flows established, each site had to be analyzed for type of treatment, size of site, recovery methods and reuse options, as well as cost. Also, at this time, numerous design criteria and assumptions had to be established. Since very little research had been done on land treatment systems in arid environments, and also owing to the site-specific nature of such systems, much of the criteria on loading rates, treatment efficiencies and area requirements had to be extrapolated from several sources.

Cost data was found to be more readily available. The June, 1975 EPA document, "Costs of Wastewater Treatment by Land Application," was used as the basic cost tool in the analysis.

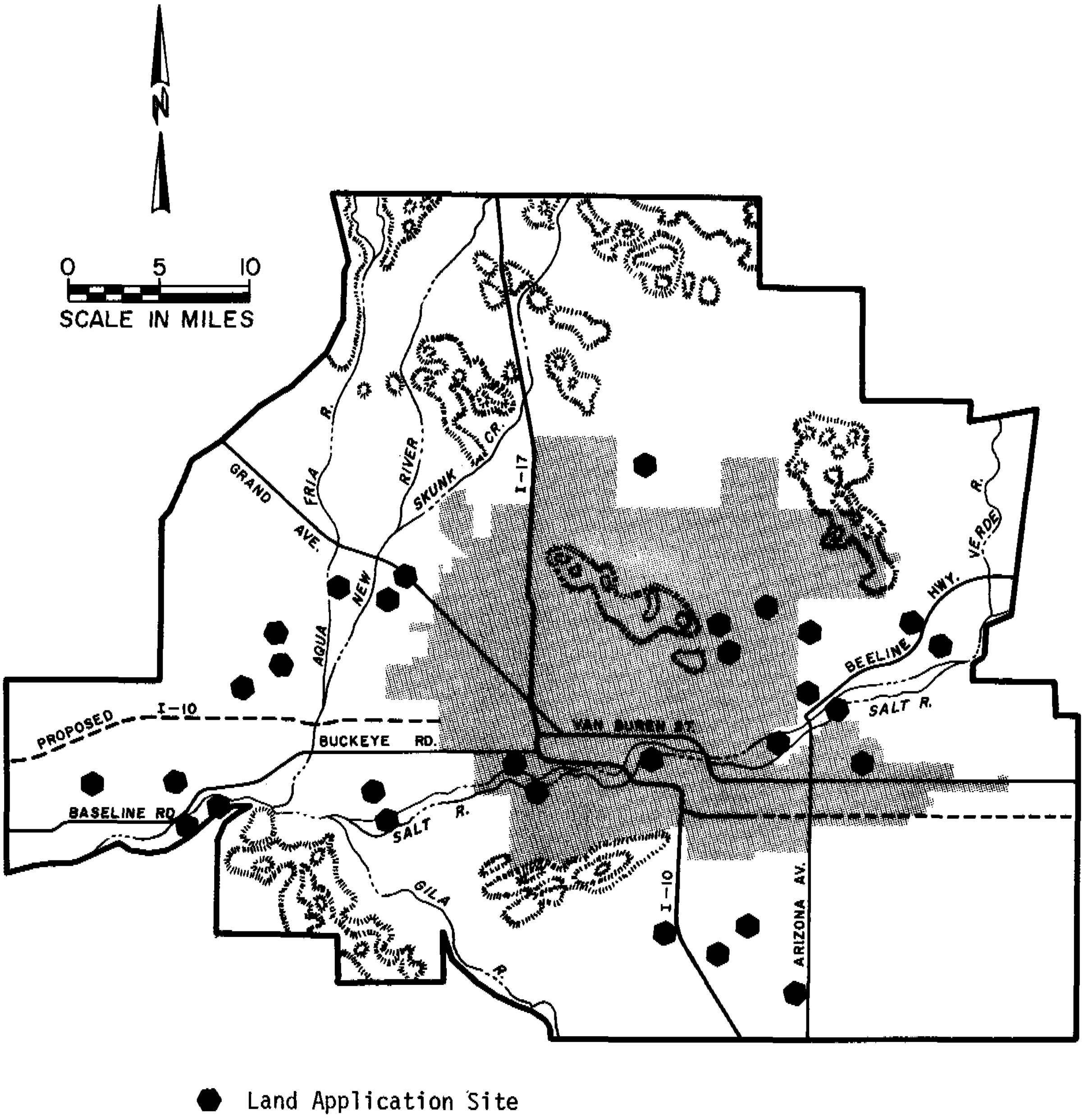
At this point in the study, it was found that approximately 30 to 40 individual sites were technically feasible and would require the above-mentioned engineering and cost analyses. (See Plate 2).

It was at this point decided that the most accurate and efficient methods of analyzing this many sites was by means of Boyle's in-house computer facilities. The system is comprised of a Computer Hardware Incorporated (CHI) 2130 central processing unit with 32 k bytes of core storage, a cathode ray tube (CRT) monitor and data input keyboard. Peripheral units consist of a CHI 1114 Disk Controller with two Memorex 660 Disk Drives providing a total of 40.96K bytes of on-line storage; a CHI 1103 Drum Printer rated at 600 lines per minute, 132 print positions and 64 character set; an IBM 1442 Model 6 Card Reader/Punch with an 80 card column, rated at 300 cards per minute; and a UCC 2000 30-inch drum plotter capable of multicolor plots.

The computer programs used will be discussed in detail in a subsequent section of this paper.

Small Array

A complex process of evaulation followed the analysis of the large array resulting in the elimination of about 20 sites from further consideration. It was decided at this point that recovery of treated effluent would be a prerequisite for land treatment systems to distinguish it from reuse.



Land Application Site

Plate 2 - Preliminary Land Treatment Sites

This effectively eliminated all irrigation systems from further consideration since if recovery were required, these systems could not compete economically with infiltration-percolation systems. Overland flow systems were also eliminated due to a lack of suitable terrain in the study area.

Two distinct types of infiltration-percolation systems were identified at this point. The difference between the systems was in the method of recovery of the renovated water. One system used in areas of shallow groundwater, relied on the use of recovery wells to pump the renovated water from the system. In areas where the groundwater was deeper than 38 meters (125 feet), it was found that the use of this method of recovery was not practical. In these areas, it was decided to study a system in which the entire treatment area would be excavated to a depth of 15 feet at which point an impermeable lining would be installed. Renovated water would be collected by means of an underdrain system installed above the lining.

At this point in the study, it was also decided to investigate the application of either primary or secondary effluent at the land treatment sites. In addition, the concept of series operation was introduced at this time. Series operation would include the application of wastewater to an infiltration-percolation basin, recovering the treated effluent and reapplying the effluent to a second basin to enhance treatment efficiency and pollutant removal.

Series operation was investigated for those sites where the primary reuse option was recreation. The State of Arizona requires that all discharges to a water course used for recreation (bodily contact) contain less than 0.2 mg/l of phosphorus. It was not felt that these levels could be guaranteed with a single pass land treatment system. As phosphorus removal is primarily a function of travel distance through the soil, an investigation of using infiltration of basins in series was considered.

It can be seen that although the number of specific sites had been reduced from the large array, we were now studying between 50 and 75 alternatives. In addition to options for the treatment of primary or secondary effluent, as well as series operation, there were numerous flow variations to be studied for several regional and sub-regional concepts.

<u>Staged Construction</u>

After the small array of alternatives had been analyzed, another complex process of review and evaluation took place. The result of this evaluation was the selection of a small number of sites for the final detailed analysis. Involved in this analysis was the development of a staged construction schedule at five year increments for each site and flow alternative. Once again, the number of individual analyses was increased despite a decrease in the number of actual sites studied.

Added to the cost analysis in this phase of the project were estimates of flood protection costs, as well as capital and operation and maintenance costs for pilot land treatment studies. Due to the site specific nature of land treatment systems and variations in soil conditions, it was felt prudent that pilot systems be established at each proposed land treatment site prior to actual construction.

MODELING TECHNIQUES

At the outset of this project it was recognized that, due to the size of the study area, many different sites each with several alternatives would have to be evaluated. As the Preliminary Assessment was being completed, it was apparent that the ultimate total of alternatives, including variations in flow, would result in literally hundreds of alternatives to be analyzed. At this point, a systems approach was confirmed and existing computer software packages were reviewed. It was quickly made evident that the only computer program available was one developed by the U.S. Army Corps of Engineers entitled, "Computer Assisted Procedure for the Design and Evaluation of Wastewater Treatment Systems" (CAPDET). This program primarily evaluated cost for conventional as well as land treatment systems. The approach was modelled after procedures as outlined in the U.S. Environmental Protection Agency's <u>Cost of Wastewater Treatment by Land Application Bulletin</u>, originally published in 1975. Although the program was extremely useful, the land application portion was still in its infancy of development. In addition, the problem of engineering analysis still remained.

Boyle Engineering Corporation proceeded to develop a complete computer software package that would meet the following criteria:

1. Analyze each site as to its suitability for the various types of land treatment.

2. Input had to be site specific and factual; soil permeability, topography, evaporation, etc.

3. Analysis had to be quick and economical.

4. Output should be hardcopy to provide engineering documentation for inclusion in project reports and design appendices.

5. Engineering analysis output should provide convenient input to cost program.

6. Cost program should be flexible to allow for variation in numerous financial indexes.

7. Should provide capability of various implementation options, lease-purchase, effluent revenue, staged construction, etc.

8. Cost output should be concise and yet easy to read.

9. Summary tables should present numerous sites for easy comparison.

Utilizing the latest land treatment technology, a comprehensive computer software package was developed. The resulting package consists of three distinct main programs with ten primary subroutines and numerous other smaller programs. The first main program (LAPPL) provides the engineering analysis, the second program (LAP 3) is a modification of the CAPDET cost program, and the last main program (LAP 4) provides summaries of the various numerical factors of each site. A flow diagram is illustrated on Plate 3 that depicts the processing scheme.

The cost analysis program (LAP 3) was initiated by computing preliminary values of pumping head and capital costs for pumping stations. This is accomplished by a subroutine - PLIM. Costs of facilities for transmission are then calculated and output by means of subroutines - TRANEX and -TRANOT, respectively. By means of previously calculated information the program then proceeds to calculate capital and annual costs for each of the various feasible methods at that site. These calculations and output subroutines are listed below:

SPRAEX & SPRAOT	-	spray irrigation
OVEREX & OVEROT	-	overland flow
RAPIEX & RAPIOT	-	rapid infiltration
DITCEX & DITCOT	-	ditch irrigation

Output from both the engineering analysis program (LAPPL) and the cost program (LAP 3) is then summarized in matrix form by a third program,(LAP 4.)

Engineering Analysis (LAPPL)

Once favorable site areas were identified by the Preliminary Site Assessment, specific sites could be located. Computer input forms were then completed for each site that enumerated such information as site number, title, available land area, slope, soil permeability, depth to groundwater, groundwater quality, evaporation, contributing wastewater planning areas or flows, and effluent characteristics. (See Plate 4.) With this information the computer program analyzed each site. Plate 5 illustrates an example of the analyses output.

The analysis was initiated by determining if the site was suitable for each of the land treatment methods. This was accomplished by comparing site slope, depth to groundwater and soil permeability to limiting parameters of each specific method as presented in the latest EPA land treatment guidelines. For those methods surviving the initial comparison hydraulic, organic, nitrogen and phosphorous loading rates were calculated to determine the governing rate for each method. Land area requirements were then calculated, including allowances for buffer zone, storage and access roads. Once this sequence had been completed, the program then summarized the input and calculations and output to a high speed printer that provided hardcopy for engineering review. A summary of pertinent data was then stored on magnetic disc in matrix form for later use in the cost analysis program (LAP 3).

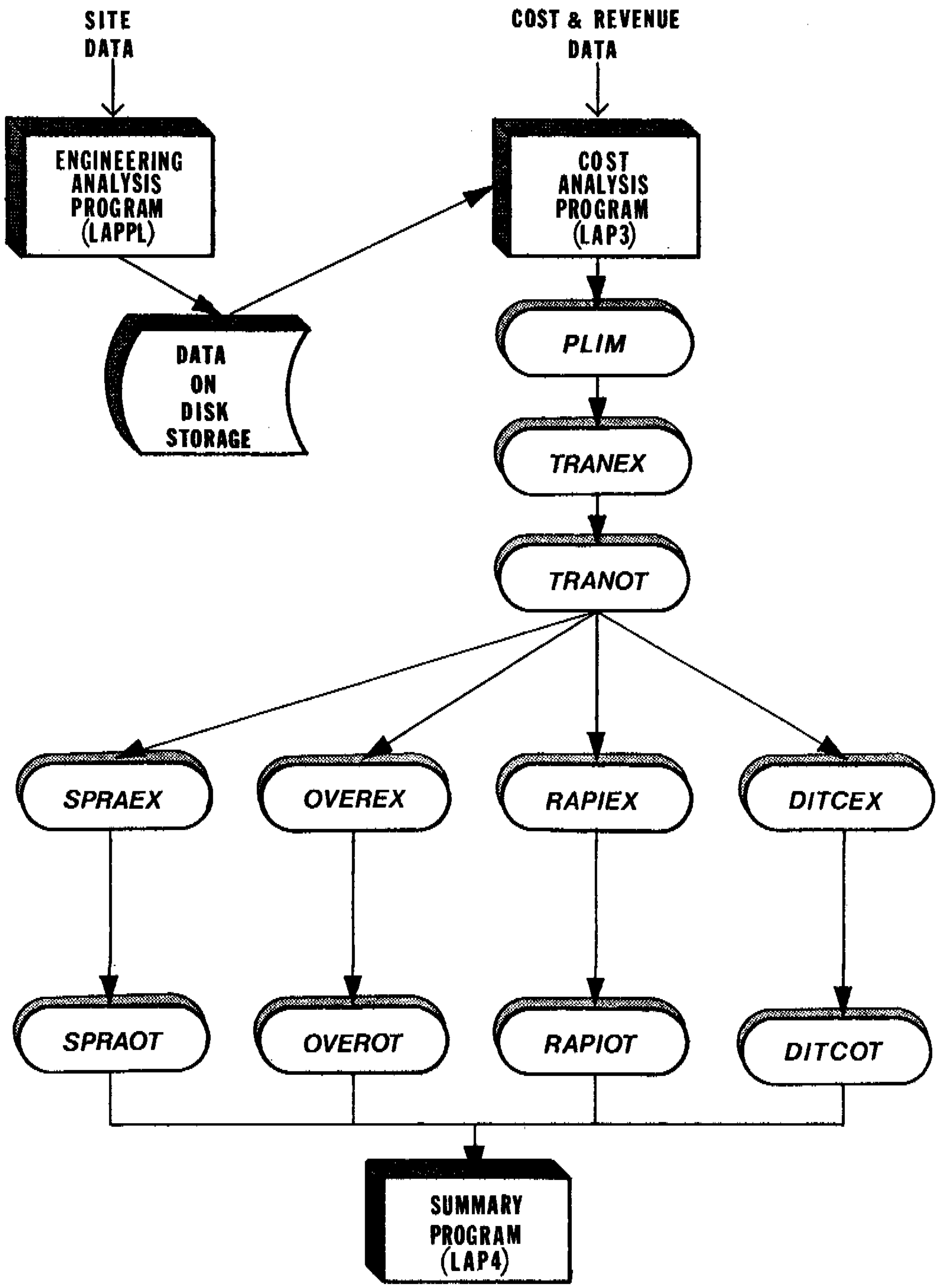


PLATE 3

LAND APPLICATION

COMPUTER SOFTWARE PACKAGE

PHOENIX URBAN STUDY - SMALL ARRAY DATA

BY____________ Sheet 1 of 3

DATE____________ J-C01-100-54

SITE IDENTIFICATION

1 ____________________ 80

1 4 SITE NUMBER (1 to 40)

APPLICATION ALTERNATIVES

SPRAY IRRIGATION - SOLID SET = 1

SPRAY IRRIGATION - CENTER PIVOT = 2

5 OVERLAND FLOW = 3

RAPID INFILTRATION = 4

DITCH IRRIGATION = 5

TRANSMISSION

6 ENTER METHOD OF CONVEYANCE

1 = OPEN CHAN, 2 = FORCE MAIN, 3 = GRAVITY PIPE

7 PAVEMENT REPLACEMENT 1 = REPLACE, 2 =NO REPLACE

8 15 LENGTH OF CHANNEL OR PIPE - FEET

16 20 PIPE DIAMETER - INCHES OR CHANNEL PERIMETER - FEET

21 25 DEPTH OF COVER OVER PIPE - FEET

26 30 PUMPING HEAD - FEET

STORAGE

31 INCLUDE EMBANKMENT COST (ENTER 1)

32 INCLUDE LINING COST (ENTER 1)

PHOENIX URBAN STUDY - SMALL ARRAY DATA

SITE NUMBER__________ Sheet 2 of 3

SITE PREPARATION

33 SITE CLEARING (ENTER 1)

34 GROUND COVER 1 = HEAVILY WOODED, 2 = BRUSH + GRASS, 3 = GRASS

35 40 TERRACE CONSTRUCTION - CY OF CUT PER ACRE (OVERLAND FLOW SYSTEMS ONLY)

41 46 LAND LEVELING - ENTER CY OF CUT PER ACRE (ALL BUT OVERLAND FLOW)

47 LINING (ENTER 1) (RAPID - INFILTRATION ONLY)

48 54 LINING COST $/SF

DISTRIBUTION PUMPING

55 57 DISTRIBUTION PUMPING - ENTER TOTAL PUMPING HEAD - FEET

RECOVERY SYSTEM

58 ENTER METHOD

1 = UNDERDRAINS (IRRIGATION SYSTEMS ONLY)
2 = TAILWATER RETURN (ALL EXCEPT RAPID INFILTRATION)
3 = RUNOFF COLLECTION (ALL EXCEPT RAPID INFILTRATION)
4 = WELLS (ALL EXCEPT OVERLAND FLOW)
5 = UNDERDRAINS WITH RECOVERY WELLS
6 = NO RECOVERY

59 60 NO. OF RECOVERY WELLS

61 63 RECOVERY WELLS - DEPTH IN FEET (ALL EXCEPT OVERLAND FLOW)

64 66 UNDERDRAIN SPACING - 100 or 400 FEET

67 RUNOFF COLL. (ALL EXCEPT RAPID INFL.) 1 = GRAVITY PIPE, 2 = OPEN DITCH

PHOENIX URBAN STUDY - SMALL ARRAY DATA

SITE NUMBER__________ Sheet 3 of 3

MONITORING WELLS

68 69 NUMBER OF WELLS

70 72 WELL DEPTH - FEET

MISCELLANEOUS

73 SERVICE ROADS - ENTER 1

74 FENCING - ENTER 1

75 AQUISITION METHOD 0 = PURCHASE, 1 = LEASE, 2 = NONE

76 EXISTING DISTRIBTUION SYSTEM - ENTER 1

77 LEASE BACK - ENTER 1

78 CROP REVENUE - ENTER 1

79 EFFLUENT REVENUE - ENTER 1

80 NO STORAGE - ENTER 1

BEC
RR/slw
2/13/78

PROGRAM LAPPL

ENGINEERING ANALYSIS

// JOB OE01 OE11
// LON
LAPPL. SUBMITTER ______ JOB NO. ______ CLIENT/PROJECT ______
// XEQ LAPPL

SITE NUMBER AND TITLE

MAXIMUM ALLOWABLE SITE SLOPES AND EFFLUENT SAR

INFILTRATION/ PERCOLATION	IRRIGATION	OVERLAND FLOW	EFFLUENT SAR
10	20	30	40

MAXIMUM HYDRAULIC LOADING RATES, FT/YR

LOW-RATE INFILTRATION	HIGH-RATE INFILTRATION	IRRIGATION	OVERLAND FLOW
10	20	30	40

SOIL DESCRIPTION

CONSTITUENT LOADING RATES--LB/ACRE/YEAR

LOW-RATE INFILTRATION	HIGH-RATE INFILTRATION	IRRIGATION	OVERLAND FLOW	
				BOD5
				NITROGEN
10	20	30	40	PHOSPHORUS

MINIMUM ALLOWABLE DEPTH TO GROUNDWATER--FT.

LOW-RATE INFILTRATION	HIGH-RATE INFILTRATION	IRRIGATION	OVERLAND FLOW
10	20	30	40

EVAPORATION RATES

LOW-RATE INFILTRATION	HIGH-RATE INFILTRATION	IRRIGATION	OVERLAND FLOW
10	20	30	40

CONSTITUENT REMOVAL EFFICIENCES

BOD5		NITROGEN		PHOSPHORUS		
MAX.	MIN.	MAX.	MIN.	MAX.	MIN.	
						LOW-RATE INFILTRATION
						HIGH-RATE INFILTRATION
						IRRIGATION
10	20	30	40	50	60	OVERLAND FLOW

CONTRIBUTING WASTEWATER PLANNING AREAS--OPTIONAL INPUT OF FLOWS

WPA	Q	WPA	Q	WPA	Q	WPA	Q	WPA	Q	WPA	Q	WPA	Q	WPA	Q
5	10	15	20	25	30	35	40	45	50	55	60	65	70	75	80

EFFLUENT CONSTITUENTS AND SITE CHARACTERISTICS

BOD5—MG/L	N—MG/L	P—MG/L	SAR	TDS—MG/L	SITE SLOPE	SITE PERC. RATE FT/YR	SITE DEPTH TO GROUNDWATER
10	20	30	40	50	60	70	80

SITE GROUNDWATER QUALITY--MG/L

TDS	NITRATE	BORON	FLUORINE	SODIUM	MAGNESIUM	CALCIUM	[illegible]
10	20	30	40	50	60	70	80

COMMENTS

A-1

PLATE 4

```
SITE  1  NORTH SCOTTSDALE-PRIMARY EFFLUENT-FIRST PASS-5.5 MGD
    FEASIBLE TREATMENT METHODS AS SPECIFIED IN INPUT--
         INFILTRATION/PERCOLATION

    MAXIMUM ALLOWABLE SLOPES---
              S=0.040 FOR INFILTRATION/PERCOLATION SYSTEMS
              S=0.150 FOR IRRIGATION SYSTEMS
              S=0.060 FOR OVERLAND FLOW SYSTEMS
    MAXIMUM ALLOWABLE HYDRAULIC LOADING RATES---
              =   4. FEET/ACRE/YEAR FOR LOW-RATE IRRIGATION SYSTEMS
              =  18. FEET/ACRE/YEAR FOR HIGH-RATE IRRIGATION SYSTEMS
              = 240. FEET/ACRE/YEAR FOR INFILTRATION/PERCOLATION SYSTEMS
              =  40. FEET/ACRE/YEAR FOR OVERLAND FLOW SYSTEMS
    MAXIMUM ALLOWABLE SAR FOR THIS SITE =  9.
    SOIL DESCRIPTION---
         LAVEEN LOAM-PALE BROWN-MODERATELY ALKALINE K= .6 TO 2.0
    MAXIMUM SITE PERCOLATION RATE =     950. FT/YR
    BASED UPON SLOPE, THE FEASIBLE TREATMENT METHODS FOR THIS SITE ARE---
              INFILTRATION/PERCOLATION
              LOW-RATE IRRIGATION
              HIGH-RATE IRRIGATION
    BASED UPON PERCOLATION RATE, THE FEASIBLE TREATMENT METHODS ARE---
              LOW-RATE IRRIGATION
              HIGH-RATE IRRIGATION
              INFILTRATION/PERCOLATION
              OVERLAND FLOW
    COMBINING FEASIBLE TREATMENT METHODS SPECIFIED IN INPUT, SITE SLOPE, AND SOIL PERCOLATION RATE,
    THE FOLLOWING TREATMENT METHODS WILL BE CONSIDERED---
              INFILTRATION/PERCOLATION

    EVAPORATION RATES---
         LOW-RATE IRRIGATION                       0.00 IN/YR
         HIGH-RATE IRRIGATION                      0.00 IN/YR
         INFILTRATION/PERCOLATION                 80.00 IN/YR
         OVERLAND FLOW                             0.00 IN/YR

    MINIMUM ALLOWABLE DEPTHS TO GROUNDWATER---
         LOW-RATE IRRIGATION                         5. FT.
         HIGH-RATE IRRIGATION                        5. FT.
         INFILTRATION/PERCOLATION                   15. FT.
         OVERLAND FLOW                              15. FT.
         ALL METHODS--QUALITY CONSIDERATION        100. FT.

    COMMENTS
    --------
         SITE GROUNDWATER QUALITY DATA SUPPLIED BY KEN SCHMIDT, FEB 4, 1978
```

PLATE 5, SHT 1

SITE 1 NORTH SCOTTSDALE-PRIMARY EFFLUENT-FIRST PASS-5.5 MGD
AVAILABLE AREA AT SITE = 5000. ACRES
INFILTRATION/PERCOLATION

THE WASTEWATER PLANNING AREAS CONTRIBUTING TO THIS SITE ARE--

IDENTIFICATION	POPULATION EQUIVALENT	AVG. FLOW--MGD
101 SET FLOW	55000.	5.50
TOTAL	55000.	TOTAL 5.50

	HYDRAULIC LOADING	ORGANIC LOADING	NITROGEN LOADING	PHOSPHORUS LOADING
FLOW= 6165.4 AF/YR = 5.5 MGD = 8.5 CFS	HYDR. APPLICATION RATE = 6165. AF/YEAR	BOD5 APPLICATION RATE = 6421. LB/DAY	N APPLICATION RATE = 1834. LB/DAY	P APPLICATION RATE = 458. LB/DAY
	MAXIMUM LOADING RATE = 240. FT/YR/AC	MAXIMUM LOADING RATE =25000. LB/YR/AC	MAXIMUM LOADING RATE =22000. LB/YR/AC	MAXIMUM LOADING RATE =10000. LB/YR/AC
WASTEWATER CHARACTERISTICS	ACTUAL LOADING RATE = 65. FT/YR/AC	ACTUAL LOADING RATE =24982. LB/YR/AC	ACTUAL LOADING RATE = 7137. LB/YR/AC	ACTUAL LOADING RATE = 1784. LB/YR/AC
BOD5= 140.0 MG/L N = 40.0 MG/L P = 10.0 MG/L	LAND REQUIREMENTS = 25.6 ACRES	LAND REQUIREMENTS = 93.8 ACRES	LAND REQUIREMENTS = 30.4 ACRES	LAND REQUIREMENTS = 16.7 ACRES
	PERCOLATION RATE = 950. FT/YR	REMOVAL EFFICIENCY = 99 PERCENT MAX. = 90 PERCENT MIN.	REMOVAL EFFICIENCY = 40 PERCENT MAX. = 20 PERCENT MIN.	REMOVAL EFFICIENCY = 99 PERCENT MAX. = 70 PERCENT MIN.
		REMOVAL RATES = 6357. LB/DAY MAX. = 5779. LB/DAY MIN.	REMOVAL RATES = 733. LB/DAY MAX. = 366. LB/DAY MIN.	REMOVAL RATES = 454. LB/DAY MAX. = 321. LB/DAY MIN.

THE GOVERNING CONSTITUENT FOR DETERMINING MAXIMUM LAND AREA REQUIREMENTS IS ORGANIC LOADING
THE REQUIRED LAND AREA INCLUDING A BUFFER ZONE OF 200 FEET ON THREE SIDES AND 500 FEET ON ONE SIDE IS 151. ACRES
TOTAL AREA REQUIRED, INCLUDING ROADS AND FENCING, IS 170. ACRES

WARNING--EFFLUENT SAR OF 26. EXCEEDS RECOMMENDED LEVEL OF 9.

EFFLUENT CHARACTERISTICS---

BOD5 = 1.4 TO 14.0 MG/L
N = 24.0 TO 32.0 MG/L
P = 0.1 TO 3.0 MG/L

SITE GROUNDWATER CHARACTERISTICS

DEPTH =	250.FT	TDS =	500.MG/L	N =	24.MG/L
B =	0.MG/L	FL =	0.MG/L	NA =	174.MG/L
MG =	2.MG/L	CA =	5.MG/L	MN =	0.MG/L

Cost Analysis (LAP 3)

After having reviewed the output from the engineering analysis (LAPPL) program decisions were made regarding specific configurations that would affect the costs of the systems. Each site was evaluated as to whether to lease or purchase the land. Transmission was then considered from wastewater source to the land treatment site. An alignment was selected, pump station requirements and transmission line criteria such as size, trenching conditions, pavement replacement, pumping head, and other pertinent requirements were established. Site conditions and desirable configurations were then finalized. These included site cleaning, distribution methods, earthwork requirements, recovery methods and post treatment criteria. Once these basic decision points had been made, effluent disposition then had to be decided. If the reclaimed water was to be reused in some form then effluent revenue values had to be established. If the land treatment system utilized crop irrigation, crop revenue values also had to be determined. After wastewater conveyance, treatment, and revenue values had been established the computer input was complete and the analysis run.

Computer Output

The resultant output from the cost analysis (LAP 3) is further summarized in comparative form by a third program (LAP 4). The hardcopy provides a number of breakdowns by parameters. Each cost parameter is broken down into capital cost, amortized cost, labor, material, power, total annual operation and maintenance as well as total annual cost. Each of these are also individually totalled. The summary program simply takes these totals and in matrix form provides a ready comparison of totals by site. Unit cost of treatment is also output. (See Plate 6)

SUMMARY

As the Phoenix Urban Study is nearing its completion, it appears that land treatment of wastewater will be a viable alternative to conventional treatment for at least one or two of the selected subregional systems. The depth and accuracy of analysis necessary for land treatment to have reached this point would have been extremely difficult under current funding levels without the use of the computer software package previously described. For a study area as large as the Phoenix area, literally hundreds of alternatives had to be evaluated. The cost to accomplish this by hand methods would have been prohibitive. The computer program proved to be an invaluable tool allowing the engineering team to maximize the level of analysis without exceeding reasonable costs. This level of detail was essential to the various advisory groups and other parties involved in the decision making process.

ACKNOWLEDGEMENT

This paper is a summary of several years of work by numerous individuals on the Phoenix Urban Study which was totally funded by the U.S. Army Corp of Engineers for the purposes of assisting local goverments in complying with Section 208 of Public Law 92-500.

SITE 1 NORTH SCOTTSDALE-PRIMARY EFFLUENT-FIRST PASS-5.5 MGD
APPLICATION METHOD = INFILTRATION/PERCOLATION

STUDY PERIOD = 20 YEARS
INTEREST RATE = 6.625 PCT
LAND COST = $ 4500./ACRE
POWER COST = 3.1 CTS/KWHR
LAND LEASE = $ 85./ACRE/YEAR
CROP REVENUE = $ 150./ACRE/YEAR
EFFLUENT REVENUE = $ 5./MG

CONST. COST INDEX (01/78) = 141.
SUPPLY COST INDEX (01/78) = 200.
AVERAGE WAGE RATE (01/78) = 8.00
CAPDET CONST. COST RATIO = 1.580
CAPDET SUPPLY COST RATIO = 1.590

COST COMPONENT	TOTAL CAPITAL COST, $	AMORTIZED CAPITAL COST, $/YEAR	LABOR, $/YEAR	POWER, $/YEAR	MATERIAL, $/YEAR	TOTAL O&M, $/YEAR	TOTAL COST $/YEAR
LAND							
(1) TREATMENT AREA 93.82(ACRES)	0.	0.	0.	0.	0.	0.	7974.
(2) BUFFER ZONE 57.47(ACRES)	0.	0.	0.	0.	0.	0.	4885.
(3) ROAD,STORAGE,ETC. 19.25(ACRES)	0.	0.	0.	0.	0.	0.	1696.
SUB TOTAL 170.55(ACRES)	0.	0.	0.	0.	0.	0.	14496.
STORAGE	0.	0.	0.	0.	0.	0.	0.
APPLICATION SYSTEM							
DISTRIBUTION	288987.	26488.	22887.	0.	3773.	26660.	53149.
SITE CLEANING	61930.	5676.	0.	0.	0.	0.	5676.
EXCAVATION WITH LINING	6987657.	640485.	0.	0.	0.	0.	640485.
SUB TOTAL	7338575.	672650.	22887.	0.	3773.	26660.	699311.
RECOVERY OF RENOVATED WATER							
RECOVERY	214739.	27234.	8904.	7331.	1991.	18227.	45462.
SUB TOTAL	214739.	27234.	8904.	7331.	1991.	18227.	45462.
ADDITIONAL COST							
MONITORING WELL	21272.	1949.	935.	1193.	576.	2705.	4655.
SERVICE ROAD	37552.	3442.	0.	0.	1192.	1192.	4634.
FENCING	33902.	3107.	0.	0.	272.	272.	3379.
SUB TOTAL	92726.	8499.	935.	1193.	2040.	4169.	12669.
TOTAL	7646041.	708384.	32727.	8525.	7805.	49058.	771939.

PLATE 6

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COST ESTIMATION

THE CLAW MODEL FOR ESTIMATING THE COSTS OF LAND APPLICATION OF WASTEWATER

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The cost of land application of wastewater (CLAW) model and its uses are described. The CLAW model consists of five basic steps: pre-application treatment, transmission, effluent storage, application system, and recovery of renovated water. Cost estimates can be generated for alternative land application techniques using a variety of design assumptions and unit costs.

Cost estimates are generated for six land application of wastewater techniques: solid-set irrigation, center pivot irrigation, border strip irrigation, ridge and furrow irrigation, overland flow, and infiltration basins. The cost estimates are for facility designs ranging from 10 to 1000 liters per second (l/sec) for each type of application system. The least cost alternative is an infiltration basin; while overland flow is the second least cost alternative. Center pivot irrigation is the least cost irrigation alternative.

Cost estimate sensitivity to variations in design costs is evaluated for center pivot irrigation systems with two different design capacities 50 and 500 l/sec. Design parameters analyzed include: Federal construction subsidies, input costs, the application rate, the length of the irrigation season, and crop selection. Receipt of a 75 percent construction subsidy significantly reduces treatment costs (by 55 percent) to the local community. The most important design parameter for irrigation systems is crop selection with its associated limits on the application rate and irrigation season. For example, irrigation of a forest 30 weeks per year is twice as costly as year round irrigation of reed canarygrass.

INTRODUCTION

Evaluation of land application as an alternative for advanced wastewater treatment requires determination of its relative cost effectiveness. Ideally, such a cost analysis should compare and evaluate alternative land application technologies with other advanced wastewater treatment options.

This analysis focuses on the relative cost effectiveness of six alternative land application techniques. A computerized cost model for evaluating land application of wastewater is presented. This model is used to make cost estimates for the six land application techniques. The analysis concludes with a discussion of the effects of relaxing the assumptions used to generate the cost estimates.

COST OF LAND APPLICATION OF WASTEWATER MODEL

The cost of land application of wastewater (CLAW) model was developed for economic analyses of land application of wastewater. The model, as described by Young (1976), is based on Pound, Crites, and Griffes' (1975) stage II detailed planning cost methodology for land application. This model can be used to evaluate land application under a variety of scenarios. Effluent quality parameters are not included in the model. Thus, the user must provide design information capable of meeting desired effluent limitations.

The CLAW model consists of five basic steps: preapplication treatment, transmission, effluent storage, application system and recovery of renovated water. 1) Preapplication treatment is assumed to be aerated lagoon treatment although other methods of pretreatment can be included.[1] 2) Effluent can be transmitted to the application site using forced main or gravity main transmission systems. The model is capable of varying the distance that the wastewater is transmitted and can account for some variations in topography. 3) The storage function is separated into three parts. First, there is the storage of wastewater which is normally assumed not to be applied to the ground for some portion of the year. This value can be varied at the user's discretion. Second, supplemental storage capacity can be built into the cost estimates as a safety factor. Third, the model permits stream discharge of wastewater for some portion of the year when it is not being applied to the land. 4) Application systems in the CLAW model include: solid set crop irrigation, center pivot crop irrigation, surface irrigation of crops, solid set irrigation of woodland, overland flow, and infiltration basins. 5) Finally the model includes recovery of the renovated wastewater if needed. This option includes recovery wells, underdrains, and the option of not recovering the effluent.

Total and average cost estimates can be generated using the CLAW model by specifying a set of 37 price and treatment option parameters (Young, 1976). For example, the user of the model can specify physical relationships such as facility size and the type of land application technology to be used. The use of options such as chlorination of the effluent prior to applying it to land can be controlled by the user to reflect alternative regulatory constraints. The crop to be grown on the irrigation site can also be varied.

In addition to the physical variables which are necessary for the use of CLAW, different input prices can be used. For example, different wage rates and electricity rates can be used. Different values for the EPA sewer construction cost index can be specified to examine capital costs. Since a land treatment site can be used for several years, two important variables are the local bond rate or discount rate and the discount period.

COST ESTIMATES FOR LAND APPLICATION OF WASTEWATER

The CLAW model as described, can be used to estimate the costs of land application of wastewater under a wide variety of assumptions. The remainder of this analysis addresses cost estimates for land application of wastewater. First, a 50 liter per sec (l/sec) center pivot irrigation system is examined to provide an example of the capabilities of the CLAW model. Second, costs are estimated for the six alternative methods of land application in the CLAW model. Third, the sensitivity of the cost estimates to changes in the design assumptions is discussed.

Example - Center Pivot Irrigation

One use that can be made of the CLAW model is for preliminary design of land application systems. A design example for a 50 liter per second (l/sec) center pivot land application system is illustrated in Table 1. The type of information provided by the CLAW model includes the assumptions underlying the cost estimates and a breakdown of the cost estimates.

The assumptions underlying the cost estimates are important since variations in these values can cause changes in the estimated costs. In this example, it is assumed that reed canarygrass is irrigated at a rate of 5 cm per week for 40 weeks a year. The value of reed canarygrass is assumed to be $16.50 per

[1]Aerated lagoon pretreatment is assumed since it is generally the least cost method to obtain an adequate level of treatment prior to land application. Land application does not require high levels of pretreatment of the effluent. Alternative secondary treatment techniques are likely to be used prior to land application when the treatment authority has an existing facility in place or when additional treatment is deemed necessary for public health reasons.

metric ton.[2] The number of hectares irrigated is dependent upon the average flow of wastewater, the irrigation season, and the application rate. In this example, there are slightly more than 77 hectares irrigated, at a cost of $3,700 per hectare. Prior to land application, the wastewater receives aerated lagoon treatment followed by chlorination. The sensitivity of the cost estimates to these underlying assumptions is addressed in the final section of this paper.

Examination of the cost components (Table 1) indicates that land application is an expensive undertaking. Construction of the 50 liter per second center pivot system initially costs over $2 million, with an additional $400,000 needed for land purchase. Using the assumptions of a 20 year design life and a 6 percent interest rate, these values translate into $189,000 per year for construction costs and $26,000 per year for land costs. Annual operation costs are $83,000. These costs are partially offset by $16,400 of net crop revenue.[3] The annual total cost of operating this center pivot land application system is $281,000. Construction and land costs combine to be the largest component of total cost — $215,000 per year.

Alternative Land Application Systems

The CLAW model can also be used to estimate land application costs using solid set, border strip, and ridge and furrow irrigation, overland flow, and infiltration basins in addition to center pivot irrigation. Cost estimates for the six land application techniques for facility sizes ranging from 10 to 1000 l/sec are illustrated in Table 2 using assumptions similar to those specified in Table 1. The application rates for overland flow and infiltration basins are assumed to be 15 and 45 cm per week, respectively.

Infiltration basins are the least expensive method for land application presented in Table 2.[4] Overland flow is the second least costly method for land application, followed by center pivot irrigation. The two surface irrigation techniques are generally slightly less expensive than solid set irrigation, except at very small facility sizes.

The average total cost estimates are broken into three major components: capital costs, operating costs, and net crop revenue. Of the components, capital costs are the largest in all instances. Capital costs are roughly three times the magnitude of operating costs. Net crop revenue is considerably smaller than either capital or operating costs.

Net crop revenue which varies between application techniques and facility sizes, can significantly reduce the costs of the land application systems. Crop revenues range from 3 to 10 percent of annual total costs for the irrigation alternatives, depending upon facility size. Crop revenue reduces costs more for larger facilities than for smaller ones. This is due to the economies of size in land application of wastewater, and the CLAW model's assumption that there are no economies of size in cropping. Net crop revenue per acre does not change as facility size increases, but capital and operating costs per acre decreases, making net crop revenue more important for larger facilities. Net crop revenues are

[2]This value is based on research by Bradley (1978).

[3]The net crop revenue estimate includes all costs for cultivation, harvesting, and crop management less revenue from crop sales to yield the net revenue from the farming operation.

[4]One needs to be careful in comparing the alternative methods for land application. The various methods such as infiltration basins, overland flow, and irrigation may not provide equivalent levels of wastewater treatment. Available sites for land application may limit the type of land application method which is available for a particular community. Infiltration basins or overland flow may not be practical due to soil permeability conditions at the particular site selected. Infiltration basins require a highly permeable soil and overland flow requires an impervious soil. Alternatively, the various irrigation techniques may also be limited by site characteristics. Center pivot, border strip and ridge and furrow irrigation techniques require relatively flat terrains, whereas solid set irrigation can be used in areas with greater slopes and in woodland areas.

Table 1. Estimated costs for a 50 liter per sec center pivot irrigation facility

Assumptions	
Average flow = 50 l/sec	Irrigation season = 40 weeks
Crop = Reed canarygrass silage	Price of crop = $16.50/MT
Effluent transmission = 3.2 km	Pumping head effluent distribution = 45 m
Storage capacity = 14 weeks	Application rate = 5 cm/wk
Price of land = $3,700/ha	Hectares irrigated = 77.3
Sewer construction cost index = 248.7	Sewage treatment plant cost index = 232.5
Interest rate = 6%	Facility life = 20 yrs
Wage rate = $6.00/hr	Wholesale price index = 140
Price of electricity = $0.03/kwh	Price of chlorine = 2.7¢ kg
Federal share of construction cost = 0.00	
Lagoon pretreatment	
Pre-application chlorination	
Site clearing - brush and trees	
3 monitoring wells 9 meters deep	

Cost Components	
Initial construction costs = $2,163,900	Annual construction costs = $188,700
Initial land costs = $391,200	Annual land costs = $26,300
Annual labor costs = $43,600	Annual materials costs = $12,200
Annual electric costs = $26,000	Annual chlorine costs = $1,000
Annual farming costs = $23,000	Annual crop revenue = $39,400
Annual net crop revenue = $16,400	Annual operating costs = $82,800
Annual capital costs = $215,000	Annual total costs = $281,400

considerably lower for overland flow since fewer acres are irrigated at a higher application rate. It should be noted that there was no increase or decrease in assumed reed canarygrass yield associated with the higher application rate.[5] With infiltration basins, it is assumed that there is no net crop revenue since it is assumed that no crop will be grown in conjunction with the infiltration system.[6]

Sensitivity Analysis

The cost estimates in Tables 1 and 2 are based on a set of design assumptions. Variations in these underlying assumptions will result in changes in

[5]The cropping submodel of CLAW is based on limited data. While it is recognized that crop yield may vary with changes in the application rate, adjustments in reed canarygrass yields would be conjecture. Therefore, yields are assumed to be constant. Additionally, the cost estimates are not sensitive to marginal changes in net crop revenue.

[6]Revenue estimates from sale of renovated groundwater from infiltration systems could possibly be used to adjust the cost estimates presented in Table 2.

Table 2. Annual total cost estimates by type of land application system.

Facility size (l/sec)	Solid-set irrigation	Center pivot irrigation	Border strip irrigation	Ridge and furrow irrigation	Overland flow	Infiltration basins
			Capital Costs			
10	95,800	89,500	91,700	91,600	88,600	79,800
25	154,200	138,000	140,900	141,000	135,200	117,400
50	246,100	214,900	217,800	218,100	205,400	174,700
100	421,300	356,500	358,700	359,300	334,900	277,300
500	1,759,800	1,426,600	1,399,600	1,402,700	1,302,300	1,033,900
1,000	3,391,000	2,716,500	2,637,300	2,643,600	2,464,300	1,922,700
			Operating Costs			
10	28,600	30,700	36,000	37,000	27,500	24,200
25	49,200	52,000	59,200	63,600	45,900	40,000
50	78,400	82,800	95,100	100,200	71,000	61,300
100	131,200	139,400	158,200	169,800	115,700	98,100
500	513,000	535,800	625,800	654,600	429,200	348,700
1,000	959,200	992,600	1,193,000	1,207,900	792,200	632,600
			Net Crop Revenue			
10	3,300	3,300	3,300	3,300	1,100	0
25	8,200	8,200	8,200	8,200	2,700	0
50	16,400	16,400	16,400	16,400	5,500	0
100	32,800	32,800	32,800	32,800	10,900	0
500	163,900	163,900	163,900	163,900	54,600	0
1,000	328,000	328,000	328,000	328,000	109,300	0
			Total Costs			
10	121,000	117,000	124,500	125,300	115,000	104,000
25	195,200	181,900	191,900	196,500	178,400	157,400
50	308,100	281,400	296,500	301,900	271,000	236,000
100	519,700	463,000	484,100	496,300	439,700	375,400
500	2,108,900	1,798,200	1,861,500	1,893,400	1,676,800	1,382,500
1,000	4,022,100	3,381,100	3,502,300	3,523,600	3,147,100	2,555,300

the estimated costs. To indicate the magnitude of these variations, a sensitivity analysis was performed for two sizes of center pivot irrigation systems: 50 and 500 l/sec. The sensitivity of the cost estimates to the availability of federal construction subsidies, changes in input costs, variations in the application rate, changes in the length of the irrigation season or amount of storage required, and variations in the crop irrigated are illustrated in Table 3.

Under U.S. public laws 92-500 and 95-217, communities constructing wastewater treatment facilities can receive grants from the federal government to pay for a portion of the construction costs. Normally these grants will be for 75 percent of the construction costs, although for innovative systems such as land application, the construction subsidy can be as high as 85 percent of construction costs. Eligible costs for a subsidy include construction costs and land costs for the irrigation site and lagoon storage facilities.

Receipt of a 75 percent construction subsidy for a center pivot irrigation system will reduce costs by about 55 percent to the local community (Table

Table 3. Cost sensitivity of center pivot irrigation to changes in the design assumption

	Facility Size			
	50 l/sec		500 l/sec	
	Average Total Costs	Percent Change in Average Total Costs	Average Total Costs	Percent Change in Average Total Costs
Average total costs from Table 2	$281,400	0	$1,798,200	0
75% construction subsidy	126,900	-54.9	783,100	-56.5
10% increase in initial construction costs (SCCI = 273.6, STPCS = 255.7)	300,300	6.7	1,916,400	6.6
10% increase in materials costs (WPI = 154)	282,600	0.4	1,804,700	0.4
10% increase in the wage rate (wage = $6.60/hr)	285,700	1.5	1,818,400	1.1
10% increase in the electric rate (electric rate = $0.033/kwh)	284,000	0.9	1,824,100	1.4
10% increase in the interest rate (6.6%)	292,800	4.1	1,877,200	4.4
10% increase in facility design life (22 years)	272,000	-3.3	1,738,700	-3.3
10% increase in crop price ($18.15/MT)	277,400	-1.4	1,758,800	-2.2
$1,000/ha increase in land costs ($4,700)	288,500	2.5	1,864,500	3.7
6.25 cm/wk application rate	271,500	-3.5	1,705,400	-5.2
3.75 cm/wk application rate	297,900	5.9	1,951,500	8.5
41 week irrigation season	275,200	-2.2	1,739,300	-3.3
1 week of supplemental storage	286,100	1.7	1,844,600	2.6
1 km increase transmission distance	290,100	3.1	1,816,000	1.0
Irrigation of corn silage for 40 weeks	277,300	-1.5	1,757,000	-2.3
Irrigation of corn silage for 30 weeks	340,100	20.9	2,375,200	32.1
Irrigation of reed canarygrass for 52 weeks	207,300	-26.3	1,117,200	-37.9
Irrigation of reed canarygrass for 30 weeks	345,600	22.8	2,430,110	35.1
Solid-set irrigation of forest for 52 weeks	259,800	-7.7	1,667,700	-7.3
Solid-set irrigation of forest for 40 weeks	348,900	24.0	2,518,800	40.1
Solid-set irrigation of forest for 30 weeks	436,200	55.0	3,397,700	88.9

3). For a 50 l/sec system, costs fall from $281,000 per year to $127,000 per year; while for a 500 l/sec facility, costs fall from $1,798,000 to $783,000 per year. Thus, receipt of a construction subsidy can significantly reduce waste treatment costs for land application systems. Young (1978) has shown that receipt of a subsidy for land application results in a larger percentage decrease in total costs than a similar percentage subsidy for conventional wastewater treatment systems.

Small changes (10 percent increase) in the cost parameters used to generate the cost estimates are analyzed. Parameters analyzed include construction, materials, labor, and electricity costs. Also analyzed are the interest rate, facility life, and crop price. A 10 percent increase in any of these variables does not result in large cost increases. The largest cost increase is for a 10 percent increase in construction costs which results in an average total cost increase of about 6.6 percent. A 10 percent increase in the interest rate results in slightly more than a 4 percent increase in average total costs. A 10 percent increase in the crop price reduces costs by about 1.4 to 2.2 percent. A similar variation in reed canarygrass yields would also produce small percentages changes in total costs (1.4 to 2.2 percent).

Land cost might be anticipated to play a significant role in the cost of land application systems. From Table 3 it can be noted that a $1,000 per hectare increase in land cost, i.e., more than a 25 percent increase in land costs, results in a 2.5 percent increase in average total cost for the 50 l/sec facility, and a 3.7 percent increase for the 500 l/sec center pivot system. Thus it can be concluded that land costs do not exert significantly larger impacts on treatment costs than many of the other design parameters, especially when compared to increases in construction costs or changes in the interest rate.

Variations in design parameters, such as the application rate and storage requirements, have larger impacts on average treatment costs than the changes in the variables already discussed. A 1.25 cm per week reduction in the application rate results in a 6 percent increase in costs for the 50 l/sec facility, and an 8.5 percent increase for the 500 l/sec facility. A similar increase in the application rate results in proportionately smaller decreases in average total costs. In other words, decreases in the application rate from the assumed 5 cm per week will affect costs more than a similar increase in the application rate. Lengthening the irrigation season by one week will reduce costs from 2 to 3 percent, depending upon facility size. It should be noted that a one week increase in the irrigation season implies one week's less storage and an associated decrease in the size of the irrigation site since that one week's wastewater will not have to be applied during the remainder of the year.

Probably the most important decision to be made regarding design of a land application system is selection of the particular crop to be irrigated with the municipal wastewater. Selection of a particular crop may carry with it limits on the length of the irrigation season and on the application rate.

The impact of changing crops and the associated changes in the length of the irrigation season are also illustrated in Table 3. Substitution of corn silage while retaining a 40 week per year irrigation season results in a 1.5 to 2 percent decrease in average total costs since the corn is a more valuable crop than the reed canarygrass. But corn may not be able to be irrigated for 40 weeks a year. If the irrigation season for corn is limited to 30 weeks per year, average total costs will increase by 20 to 30 percent depending upon facility size. A similar reduction in the irrigation season for reed canarygrass will result in a 23 to 35 percent increase in annual total costs. If reed canarygrass can be irrigated year round, substantial cost reductions result. Costs fall to $207,000 per year for the 50 l/sec facility (a 26 percent decrease), and $1,117,000 for the 500 l/sec facility (a 38 percent decrease in average total cost).

If it is assumed that forest land will be irrigated, rather than crop land, costs will be considerably higher. Irrigation of forest land requires the use of solid set irrigation with a more costly spacing of irrigation risers. Substitution of forest for reed canarygrass with a 40 week per year irrigation season raises costs by 24 percent for the 50 l/sec facility and by 40 percent for the larger 500 l/sec facility (Table 3). If forest land can be irrigated year round while reed canarygrass cannot, costs will be about

7 percent lower for forest irrigation. If, on the other hand, forest land can only be irrigated for 30 weeks per year and still retain its renovative capacity for wastewater, costs will increase by 55 percent for the 50 l/sec facility and by 89 percent for the 500 l/sec facility. If it is assumed that application rates change with selection of particular crops, the cost changes will be magnified.

SUMMARY

The cost of land application of wastewater (CLAW) model is capable of estimating the costs of alternative land application techniques using a wide variety of assumptions. Cost estimates for six alternative techniques were presented in this analysis. Comparison of these techniques indicates that infiltration basins followed by overland flow are the least expensive methods for land application. Of the irrigation alternatives examined, center pivot irrigation is generally the least expensive while solid set irrigation is the most expensive alternative.

The sensitivity of the cost estimates to changes in the design parameters was analyzed. Results indicate that availability of a subsidy for construction of the wastewater treatment facility can result in large cost decreases to the local treatment authority. The most important cost parameter for designing a land application system is crop selection. In addition to providing a revenue which can be used to offset the cost of the land application system, crop selection carries with it limits on the length of the irrigation season and on the application rate. The least expensive irrigation alternative examined is year round irrigation of reed canarygrass, while the most expensive alternative identified is solid set irrigation of forest land for 30 weeks per year. Irrigation of forests for 30 weeks per year is over twice as expensive as year round irrigation of reed canarygrass.

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COST ESTIMATION

CASE STUDY COMPARISONS OF OPERATION AND CAPITAL EXPENDITURES OF LAND AND CONVENTIONAL WASTEWATER TREATMENT SYSTEMS OF SMALL MUNICIPALITIES

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Land treatment as an alternative form of wastewater treatment has become increasingly important in recent years. With the inception of Public Law 92-500, the Federal Water Pollution Control Act amendments of 1972 established guidelines for greater improvement in the quality of wastewater treatment. The amendments state that land treatment must be considered in the initial planning stages for wastewater treatment projects, before a community can qualify for federal grant money. Land treatment is being considered as a viable means of treating wastewater, because traditional treatment processes cannot handle the increasingly complex waste load. Other forms of advanced wastewater treatment systems can be expensive to build and operate, particularly for a small community which historically treated its wastes with privately owned septic tanks.

This study addressed several questions concerning the land treatment method for small municipalities. The institutional, physical, financial and agricultural characteristics of various types of land treatment systems were documented. Operation and construction costs were determined for six land treatment systems in Michigan. This information was compared with operation and construction costs of several conventional treatment systems for small municipalities. Data on the economic and institutional characteristics of each facility were collected from state and local officials. Operation and construction cost information were collected chiefly from local community audit reports.

A comparative cost analysis of the operation expenses of the two general categories of treatment, land and conventional, showed that the land treatment systems were generally less expensive to operate than conventional systems. Comparisons of individual cost categories were made along with statistical analysis. Salary and wage expense for conventional-type treatment systems was higher than the expense incurred in the same category for the land treatment systems. The conventional systems were also more expensive in terms of operation and maintenance, and overall total costs. Land treatment systems were generally more expensive in terms of their administrative expenses. The comparison of construction costs has shown that for the systems included in this study, land treatment systems required a smaller capital expenditure than conventional treatment systems.

INTRODUCTION

Many small communities in Michigan are located around lakes or along streams that have a limited capacity to assimilate biochemical oxygen demand (BOD) and nutritient loads from wastewater. These municipalities are faced with a choice between conventional tertiary treatment (basically phosphorus removal) and a form of land treatment for final discharge of their domestic wastewater [Malhotra, 1975].

Small rural communities are somewhat of a special problem, because they seldom have enough funds to finance a sewage treatment system to meet new water quality requirements. Grant money is available from state and federal sources, but in many instances, communities still bear a heavy burden for raising the local share of funding for a system. Because of this and other unique difficulties facing small communities, land treatment for advanced wastewater treatment is a viable alternative.

There are currently 60 wastewater treatment systems operating and 6 others under construction which incorporate some form of land treatment into their design (systems which use the soil in the treatment process). This is 17.8 percent of the 369 municipal wastewater treatment systems in Michigan (Table 1). Four systems contain two methods of application. Of the 66 municipalities operating land treatment systems, approximately 50 have received grant money from state and federal sources.

The total number of persons served by land treatment systems in Michigan is approximately 171,871. This is 2.74 percent of the population of approximately 6,272,594 being served by municipal sewage treatment systems. Facilities such as mobile home parks, apartment houses, industries, schools, campgrounds, commercial establishments, hospitals, restaurants, and federal, state and local institutions may have their own sewage treatment units.

Objectives of Study

This report presents the results of a study of the institutional, physical, financial, and agricultural characteristics of various types of land treatment systems of wastewater by small Michigan communities. Operating and construction costs are presented for six land treatment systems. The various land treatment methods include spray irrigation, flood irrigation, ridge and furrow irrigation, and seepage lagoons. This information was compared with the costs of conventional treatment methods for four communities.

Specifically, the study objectives were to:

1. Identify the institutional, physical, agricultural, and financial characteristics of selected case study systems.
2. Present data on the construction, operating and maintenance costs for selected land and conventional treatment systems currently operating in Michigan.
3. Compare and analyze categories of cost information of land treatment systems with conventional systems.

Table 1. Distribution of Land Treatment Systems in Michigan

1. Distribution by Land Application Type:

Type	Number	% of Total
Flood Irrigation	16	23.5
Ridge and Furrow	4	5.8
Seepage Lagoons	26	38.2
Spray Irrigation	17	25.0
Subsurface Application	5	7.3
Total[1]	68	100.0
Percent of Total Wastewater Treatment Systems in Mich.		17.8

2. Distribution by Population Served:

Population Served	Number of Systems
0 - 500	12
501 - 1,000	17
1,001 - 1,500	10
1,501 - 2,000	12
2,001 - 3,000	4
3,001 - 4,000	3
4,001 - 5,000	1
5,001 - 6,000	1
6,001 - 9,000	3
9,001 or above	1
Total[2]	64
Percent of Persons Served by Land Treatment Systems in Michigan	2.7%

[1]There are 66 systems, but two use two application types. Six systems are still under construction. These figures only include municipal systems.

[2]Population served in two of the systems is unknown.

Source: "The Superlist," 1976 Wastewater Engineering Section, Department of Natural Resources, State of Michigan.

Comparative Cost Analysis

Information collected on the capital investments and operation costs of the case study treatment systems was combined into several accounts. Calculations based on these expense accounts were used to determine the operation costs of these systems on the basis of dollars per thousand liters treated. This calculation was based on a 365-day year and the average daily flow rates for each respective system. Categories of operation costs for the land treatment and conventional treatment systems were plotted for comparative cost analysis purposes. Linear regression equations were also plotted for these operation cost categories. Statistical tests were conducted to determine if the slopes and intercepts of the linear regression equations were significantly different from one another. Capital or construction costs analysis was also used to evaluate alternative methods of wastewater treatment.

SAMPLE COMMUNITIES

A summary of the characteristics of the land treatment systems for the sample communities is shown in Table 2. Several considerations were involved in the selection of these communities. These are systems treating mainly domestic and a small percentage of light industrial wastewater. Operation and maintenance of the system had to be acceptable in the sense that the effluent quality was meeting approval of regulatory agencies. Detailed financial information must have been available.

Comparative Cost Analysis Accounts

Information on capital investments and operation costs of the system were collected from local community officials and local government unit's audit reports. The information on operation costs was combined into four major expense accounts: (1) salaries and wages; (2) utilities; (3) operations and maintenance which included supplies, maintenance, equipment rental, professional services and miscellaneous; and, (4) general administrative which included office, insurance and transportation expenses.

Comparative Cost Analysis of Operation Expenses

Total operating costs of all categories are shown in Figure 1. The Hart land treatment system consistently has the lowest total operation cost of all the systems in the study. This is because it has, in general, the lowest cost per 1,000 liters treated in all operating cost categories, except utilities.

The land treatment systems operated by Middleville and Harbor Springs and the

Table 2. Summary of Physical, Financial, Agricultural and Institutional Characteristics of the Land Treatment Systems

Municipality	Harbor Springs I	Harbor Springs II	Wayland	Middleville	Farwell	Hart	Dimondale
Application Type	Spray	Flood	Spray	Spray	Flood	Ridge & Furrow	Seepage
Population Served	W 3,580 S 5,800	S 7,500	1,860	1,800	900	1,815	972
Design Flow (MLD)	1.703	Total I&II 3.785	1.892	.832	.510	3.406	.757
Lagoons (hectares)	8.91	7.29	12.55	8.91	.52	12.55	Conv. Sec. Treatment
Irrigation Area (hectares)	11.34	20.25	21.46	12.15	2.75	32.4	1.62
Total Area (hectares)	173.34	Additional 63.99	53.33	50.625	16.2	89.1	13.36
Total Construction Cost	(1970) $1,274,287	(1974) $5,930,731	(1971) $2,219,763	(1970) $382,430[1]	(1970) $1,222,354	(1972) $1,438,915[2]	(1969-1971) $938,468
Total Construction Cost (1976 $)	$2,378,417	$7,144,796	$3,694,555	$712,732[1]	$2,272,650	$2,168,252[2]	$1,564,080

[1]Costs do not include previously existing collection system.
[2]Costs do not include previously existing system which was incorporated into the 1972 system.

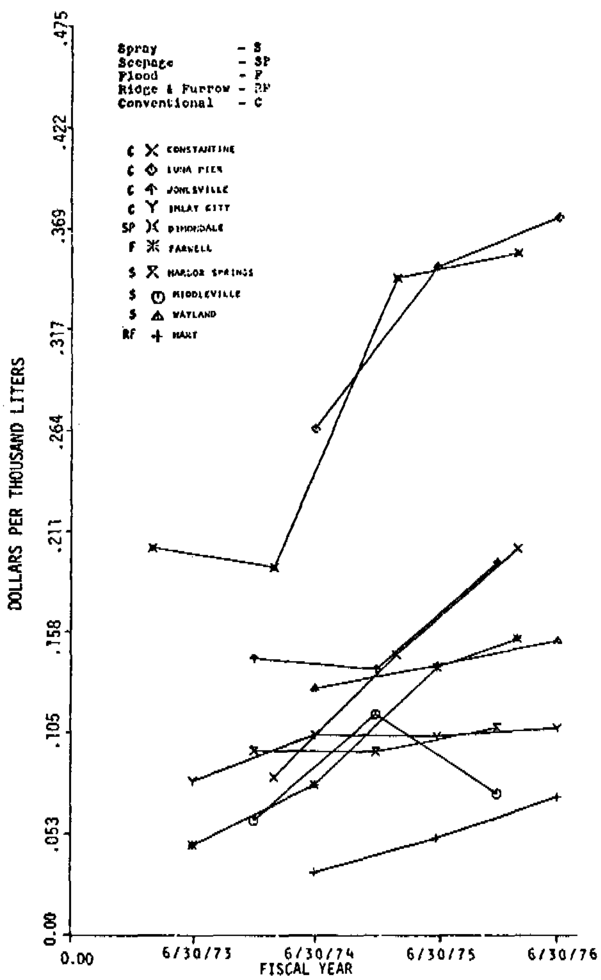

Fig. 1. Total Operating Costs

conventional treatment system at Imlay City are the next most consistently inexpensive systems to operate. Several points need to be clarified in terms of these systems to explain their operation costs. The Imlay City treatment plant does not use any process for phosphorus removal. If chemical coagulation or any other process were instituted, the operation costs would be higher. The peak in the Middleville system's cost curve is due to an expensive engineering study conducted during the 1974 fiscal year. The Middleville system has the second least expense of the systems in terms of salaries and wages. This is because much of the system is automated.

Harbor Springs has the next to least expensive cost for the general operation and maintenance category. In the general administrative expense category, it is generally the most expensive. This is understandable in that the system is an authority which serves several units of government. Also, "double billing" and bookkeeping expenses take place. The authority must bill each individual local unit of government for its use of the system, and in turn, each unit of government bills its local user for the use of the system. Additionally, the authority keeps books for its separate part of the system.

The Imlay City system has the lowest utility expense per 1,000 liters treated. In addition, this system has had the lowest expense for salaries and wages for all of the conventional treatment systems.

For the Hart, Harbor Springs and Imlay City systems, it may be possible that the economies of scale are occurring for the total operation expense on a dollars per 1,000 liters treated basis. These three systems had the highest average daily flows of the systems in the study.

The curve displaying the Farwell system's costs shows that they are increasing fairly rapidly. This is because the amount spent on salaries and wages, general operation and maintenance and general administrative expenses increased rapidly. However, the utility expense category maintained a fairly constant level. There is no immediate explanation for the increases in the other categories.

Total operation costs in dollars per 1,000 liters for the Constantine system are increasing dramatically. These substantial increases in costs can be partly explained in that flows in this system were less in the last two fiscal years because of a strike and consequent stoppage of production and output of wastewater by a large industrial user in the community. The actual dollars spent for total operation of the system increased even as flows decreased. However, if flows to the system had remained fairly stable, the increase in costs per 1,000 liters may not have been as dramatic.

The Wayland system appears to be consistently the most expensive land treatment system (excluding Dimondale which uses conventional primary and secondary treatment processes). It's utility expense, general operation and maintenance expense and general administrative expenses are consistently higher than the other land treatment systems. The amount spent for professional services is generally higher than any other land treatment system in the study.

The next most expensive system appears to be the Jonesville system. This conventional system has the highest costs in the general operation and maintenance category of the conventional systems, excluding Dimondale. However, this system is fairly inexpensive in terms of utility expense and general administrative expenses.

Luna Pier and Dimondale's treatment systems are the most expensive treatment systems included in the study. Dimondale's system, which incorporates conventional primary and secondary treatment units in its process along with seepage lagoons for tertiary treatment, has either the highest or second highest expenses on the basis of dollars per 1,000 liters treated of all the cost categories except general administrative expenses.

The Luna Pier system has the highest costs per 1,000 liters treated in the salary and wage category of all the systems. It also has the highest expenses on the same basis for the utility category (excluding Dimondale). In the general administrative expense category, it ranks highest among the conventional treatment systems.

Linear Regression Analysis of Operation Expense

A two variable linear regression technique was used on three general groups of data. The cost of operating the system is the dependent variable and time is the independent variable. The general groups were all land treatment systems, excluding Dimondale; all conventional treatment systems, excluding Dimondale; and all conventional treatment systems, including Dimondale. Figure 2 shows the regression lines for the total operating costs.

Statistical tests were conducted to determine if the slopes and intercepts of the regression lines were significantly different from one another. The regression lines were pooled and the t statistic and significance level calculated for the beta coefficients in the pooled multiple regression equation. The tests revealed that only the slopes for salaries and wage expense regressions between land treatment and conventional systems (including and excluding Dimondale) were singificantly different at approximately the 95 percent level of significance. A broad range of significance levels were found in the other comparisons. Further details of the statistical procedure and results can be found in Williams, Connor and Libby [1977].

It was found that the salary and wage expense for conventional type treatment systems is substantially higher than for land treatment systems. For the

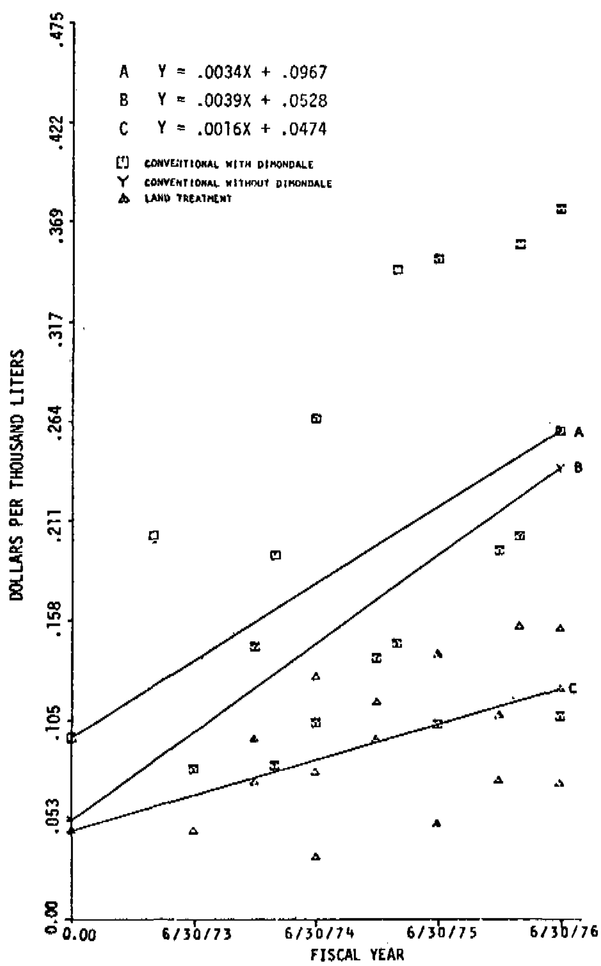

Fig. 2. Regression of total operation costs

fiscal year ending June 30, 1976, the salary and wage expense for the conventional systems was approximately $.108 higher per 1,000 liters treated than the land treatment systems. This is understandable because many of the conventional treatment systems require more manhours of labor to permit normal operation than land treatment systems. Conventional systems may require at least one or more full time employees at the treatment plant at all times. Alternatively, a land treatment system can be run in some instances by a person who works for several departments at the local unit of government. In other words, operation labor is much less than that required at similarly sized, secondary treatment plants using the activated sludge process or trickling filter. Lagoon and irrigation systems do not require two or three-shift operator attention. Operator attention in Michigan is primarily devoted to dike maintenance around ponds in spring and summer and to effluent distribution and crop management during the irrigation season [Malhotra].

These costs may also be higher because it is possible that employees on a conventional system are relatively more skilled or specialized and receive a higher wage rate.

Expenses in this category are increasing at a faster rate for conventional systems than land treatment systems. A partial explanation of this increase is the elasticity of labor supply. Assuming that the supply for skilled labor is more inelastic relative to the unskilled labor supply for these wastewater treatment systems, it is possible that the price for skilled labor for conventional treatment systems will increase faster than the price for unskilled labor for land treatment systems.

The regression analysis results for the utility expense category were not as clear. By including, or not including, the Dimondale system in the conventional treatment group, the regression line changed position substantially. It is quite possible that the difference in the conventional utility expenses and land treatment utility expenses is due mainly to the variance of the cost of electricity in each area.

General operation and maintenance expenses were higher for conventional treatment than land treatment. The expense of general operation and maintenance of the conventional systems for the fiscal year ending June 30, 1976 is approximately $.092 per 1,000 liters treated greater than the land treatment systems.

If Dimondale is included as a conventional treatment system, the regression results showed that the expense of operation and maintenance has increased approximately the same rate in both types of systems.

The total of the first three expense categories revealed that the conventional systems are substantially more expensive to operate, and operating expenses are increasing faster for conventional systems than land treatment systems. For the fiscal year ending June 30, 1976, the conventional systems are approximately $.121 per 1,000 liters treated more expensive than the land treatment systems.

The regression results for the general administrative expenses revealed that land treatment systems in the study were generally more expensive on the basis of dollars per 1,000 liters treated than the conventional treatment systems. For the fiscal year ending June 30, 1976 the land treatment systems were approximately $.042 per 1,000 liters treated more expensive than the conventional treatment systems in this expense category.

The regression lines for all costs incurred in the operation of the systems show that the conventional systems included in the study were more expensive to operate than the land treatment systems (Fig. 2). For the fiscal year ending June 30, 1976, the conventional treatment systems were approximately $.118 (line B less line C) per 1,000 liters treated more expensive than the land treatment systems in this total operation costs category. It also shows that costs for these conventional systems are increasing approximately twice as fast as the costs of operation of the land treatment systems as a whole.

A final note on the comparative cost analysis of operation costs is needed. The revenue collected by the Middleville and Hart treatment systems for the use of land and sale of crops from the treatment site would help offset the operation costs of the system. If the revenue obtained from the farming enterprise were subtracted from operation expenses, the result would be a reduction in the cost of operation at the Middleville system by $1,200 in the 1975 fiscal year. This is a reduction in operation costs from $.073 to $.070. Approximately $1,140 was received from the farm operation at the Hart system for the fiscal year ending June 30, 1975. This is a reduction in operation costs from $.050 to $.049 per 1,000 liters treated. The community of Wayland also hoped to offset part of its operation costs with an expected $6,000 profit from its farm operation in 1977. This would be a reduction of $.023 per 1,000 liters treated.

Capital Cost Analysis

Capital, or construction costs, are also an important factor to be considered when evaluating alternative methods of wastewater treatment. In general, the community will annually incur, along with operation expenses, the expense of bond issues used to construct the treatment system. Construction costs vary depending upon type of system, its components, and the overall size. In addition, land treatment system's capital costs also vary because of the availability of land, type of land and type of application system used. Various types of application systems require different amounts of land.

Table 3 presents the capital expenditures in 1976 dollars. The average construction costs of land treatment, including and excluding the Dimondale system, are presented along with the conventional system average capital costs. A weighted average construction costs was calculated by dividing the total flow of all the land treatment systems into the total capital cost of all land treatment systems. The same procedure was done for the conventional systems.

This comparison shows that for the systems included in this study, land treatment systems required a smaller capital expenditure than conventional treatment systems. However, this result may be affected somewhat because the cost of Middleville's original collection system is not included in costs of land treatment systems. On an average daily flow basis, the capital costs for land treatment were approximately $.328 per liter less than the conventional systems (excluding Dimondale). On the design flow basis, the result was a difference of $.324 per liter with the land treatment systems being less expensive than the conventional systems (excluding Dimondale). In only one case studied, the Dimondale system, were the circumstances better for the construction of conventional primary and secondary treatment units. However, due to the relatively high operation cost of the Dimondale system, it may have been advisable to construct a land treatment system at a larger capital investment and saved on operation costs over the long run.

Table 3. Construction Cost Comparison of Treatment Types - 1976 Dollars

System	Average Cost	Weighted Average Cost Per	
		Average Flow	Design Flow
		---Per Liter---	
Land Treatment	$2,476,436.13	$2.338	$1.483
	2,324,376.79*	2.508	1.532
Conventional	$2,668,067.12	$2.666	$1.807
	2,392,070.37*	2.929	1.845

*Calculations include the Dimondale system.

CONCLUDING POINTS FOR COMPARATIVE COST EVALUATION

Some concluding points should be considered when evaluating alternative treatment methods. Total expenditure required to purchase land for land treatment systems will generally be higher than for land purchased for a conventional system. This is basically due to the quantity needed. However, this land is an asset that will more than likely increase in value rather than depreciate like the equipment incorporated into the system.

It is important to remember that there is a dual cost structure. In other words, one should not only be concerned with the prices of the inputs, but also the input's salvage value when the system has exhausted its useful life. In the case of land treatment systems, the salvage value of the system would be positive because of the value of the usable land. The land could be sold by the local unit of government to pay debt retirement requirements, or to help finance a new type of system. Land could possibly be used for agricultural and forestry production or developed for recreation purposes, wildlife habitats or a variety of commercial uses. Even the ponds could be filled with soil and regraded to the natural topography. For conventional systems, however, the salvage value of the system may be small or even negative because the majority of the assets involved in the conventional system would likely have no alternative use.

There is also the advantage of producing forestry and agricultural products for sale on certain types of land treatment systems which can help defray operation and maintenance expenses. However, operation costs may also increase because of the complexity of the farm operation. The farm operation should be run as a separate enterprise to determine its profitability. The expected production and profitability should be closely studied in an evaluation of alternatives. These types of opportunities do not exist with conventional systems. However, they may be able to sell conditioned sludge from the treatment process as a fertilizer and soil conditioner to farmers and home gardeners. Sludge is not removed and conditioned from lagoons in the land treatment systems.

CONCLUSIONS

The comparative cost analysis of the operation expenses of the two general categories of types of treatment, land and conventional, has in general shown that the land treatment systems on the whole seem to be less expensive to operate than conventional systems. This finding is even more significant in that the land treatment systems may be achieving a higher level of wastewater treatment than the conventional facilities. As indicated previously, the quality of treatment generally achieved by the land treatment process is of a higher degree than conventional treatment processes which incorporate chemical coagulation for phosphorus removal.

More specifically, the land treatment systems were less expensive to operate than the conventional systems in two of the four individual expense categories documented: Salary and Wage Expense, and General Operation and Maintenance Expense. In the Utility Expense category, the operation costs were approximately the same. The conventional systems appeared to be less expensive to operate than the land treatment systems in the General Administrative Expense category. The total of all expense categories shows that the land treatment systems were experiencing lower operation costs per 1,000 liters treated. Additionally, the yearly operation costs of the conventional treatment systems were increasing approximately twice as fast as the land treatment facilities.

The comparison of the capital investment required for construction of the systems showed that land treatment was somewhat less costly than the conventional systems. Although the documentation of the capital costs indicates the general expense of construction, it is difficult to make cost comparisons across types of systems because of the inherent physical characteristics of each system. Each type of alternative system for a single community must be evaluated in terms of its expected operation costs and capital expenditure. Determination of these costs require detailed planning and study taking into account all cost factors relative to the respective community situation. Alternatively, evaluation of the average occurrences of operating characteristics and costs such as this study has done with case histories and time series data, regression analysis indicates what can be expected from the use of ultimate methods of treatment.

With the results of this study in hand and the advantages of land treatment, the question arises as to why land treatment is not more widely used. There are several reasons which are particularly apparent. Soils in many areas are not suitable for the land treatment process. There may be local opposition to the land treatment concept for various reasons including aesthetic reasons, health concerns, and lack of understanding of the land treatment concept. The amount of land required may be unavailable or the cost of purchasing or leasing such land may be prohibitive. In addition to these, it may be the case that local consulting firms have a lack of knowledge or experience with land treatment. It is clear that although there are distinct advantages for certain communities which use the land treatment method for wastewater treatment, there are also disadvantages in some situations. Therefore, land treatment should not be considered a panacea for every community's sewage problems.

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MATHEMATICAL MODELING

SIMULATION OF THE MOVEMENT OF CONSERVATIVE CHEMICALS IN SOIL SOLUTION

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A numerical method is introduced to simulate the movement of conservative chemicals in soil by water. The method is essentially based upon a finite element approximation to the equation of continuity, and each element constitutes a complete mixing cell. The number of cells represents a degree of mixing. The theoretical justification of the method is presented and the accuracy of the method is examined using experimental data obtained from a large lysimeter. It is found that the method can simulate the general trend of the movement of chemicals reasonably well, but fails to simulate the high frequency of variations that occur near the soil surface.

INTRODUCTION

In land treatment various substances in the wastewater take part in the chemical and biological reactions that determine the extent of total water renovation. Since the rates of such reactions generally depend upon the concentrations of the reacting substances, it is important to find these concentrations in terms of time and space for accurate determination of the conversions.

For the case of a solute which does not react with soils, solute molecules move in the same way as water molecules. Determination of the concentrations of the non-reactive solute in terms of time and space requires knowledge of net water movement as described by Darcy's law, and the degree of mixing or bulk diffusivity of the solute, accounting for the effect of molecular diffusion. In addition to such knowledge, rate equations are required for the general case of a solute reacting with soils. Since the reaction affects the solute, solute molecules move differently from water molecules.

Boast (1973) reviewed mathematical models simulating the movement of chemicals through soils, which may be considered a special case of general porous media. The problem of mixing in porous media was extensively studied in chemical engineering, particularly in the design of packed bed reactors (Uhl and Gray, 1966). Flow through soils is mostly in the laminar flow region defined as $Re < 1$. Re is the Reynolds number, given as

$$Re = du\,\rho/\mu \qquad (1)$$

where

- d = particle diameter
- u = mean velocity over the cross section
- ρ = density of liquid
- μ = viscosity of liquid

And in the laminar flow region mixing is caused primarily by molecular diffusion and the relative motions of liquid particles, and to a lesser extent by turbulent diffusion. The degree of mixing is often characterized by the Peclet number, Pe, given as

$$Pe = du/(D\lambda) \quad (2)$$

where

D = bulk diffusivity
λ = void fraction

For uniformly packed porous media with very little dead space, the Peclet number remains between 0.3 and 1.0 for a wide range of Reynolds numbers (10^{-1} to 10^2) (Uhl and Gray, 1966).

Kunze and Kirkham (1961) tagged water with deuterium hydroxide (DOH) and determined the bulk diffusivity in flow through Colo clay loam soils and glass beads. They found that the bulk diffusivity of both media was of the same order as the molecular diffusivity of water. Corey and Horton (1968) determined bulk diffusivity through saturated Vaucluse soil, and the value was about 100 times greater than the molecular diffusivity of water ($2x10^{-5}$ cm^2/s).

A chloride ion is usually considered the most nearly ideal tracer for determining bulk diffusivity through soils. Smith (1972) determined bulk diffusivity in 15 widely varying soils using chloride and found that the values ranged from $1.39x10^{-5}$ cm^2/s for the Amerillo and Quincy soils to $1.94x10^{-4}$ cm^2/s for Houston soil. He also found that chloride moved through the soils 1.04 to 1.67 times faster than it would have if it had been associated uniformly with all the soil water. There are two explanations for the deviation in the residence time of chloride. One is due to electrical anion exclusion (Smith, 1972) and the other is due to the dead pores, which are not accessible to even the main water flow pathways (Passioura, 1971, Coats and Smith, 1964). The soil solution in the dead pores comes to equilibrium with the rest of the soil solution primarily by means of molecular diffusion (Philip, 1968).

In this work we evaluate a numerical method simulating the movement of chemicals in soil solution. The method is essentially based upon a finite element approximation to the equation of continuity. The theoretical justification of the method is presented and the accuracy of the method is examined using experimental data obtained from a lysimeter by using chloride as a tracer.

THEORY

The equation of continuity in a one-dimensional Cartesian coordinate system is given as

$$\frac{\partial}{\partial z}\left(D\frac{\partial C}{\partial z}\right) - \frac{\partial}{\partial z}(uC) - f(C) = \frac{\partial C}{\partial t} \quad (3)$$

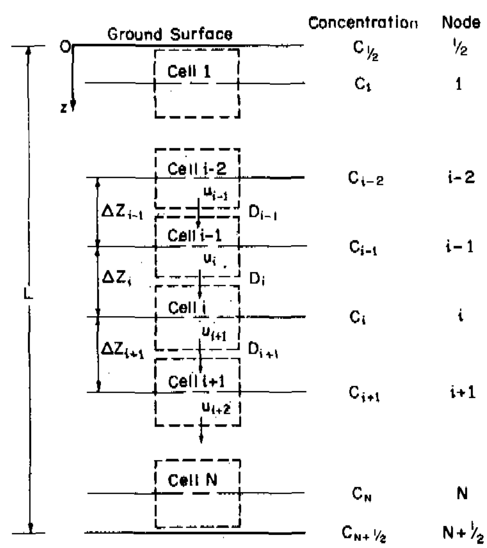
where

C = concentration of a solute
t = time
z = space variable; the z-axis coincides with the direction of gravity
$f(C)$ = sink term simulating reactions and plant uptake, if present.

Equation (3) with proper initial and boundary conditions can be solved by the finite difference method. Although the finite difference method is accurate, it is not always efficient in solving field problems. Particularly for a system in which there are many different solutes involved in various reactions, simultaneous solution of many partial differential equations such as (3) is needed and numerical analysis becomes quite elaborate. Therefore we used another method.

In this method a soil system with total thickness L is divided into N complete mixing cells with unit area of cross section and variable thickness (Fig. 1). A finite element method using a series of complete mixing cells was frequently used in simulating a degree of mixing (Uhl and Gray, 1966). Dutt et al. (1972) developed a computer model

Figure 1. A series of complete mixing cells.



simulating a bio-physicochemical process in soils. In their model complete mixing was assumed in each equally divided segment. However, the number of segments was not determined according to the degree of mixing.

It is well established that a number of division has a definite relation with a degree of mixing (Deans and Lapidus, 1960; Uhl and Gray, 1966). The justification for the use of a series of complete mixing cells as an approximation to (3) and the relation between a number of division and a degree of mixing is as follows.

The material-balance equation of the solute in the cell is given as

$$u_i C_{i-1} - u_{i+1} C_i = [(\Delta z_i + \Delta z_{i+1})/2](f(C_i) + \frac{dC_i}{dt}) \quad (4)$$

On the other hand a finite difference approximation to (3) may be written as

$$2[D^n_{i+1}(C^n_{i+1} - C^n_i)/\Delta z_{i+1} - D^n_i(C^n_i - C^n_{i-1})/\Delta z_i]/(\Delta z_i + \Delta z_{i+1}) - 2[(C^n_{i+1}+C^n_i)u^n_{i+1}/2 - (C^n_i + C^n_{i-1})u^n_i/2]/(\Delta z_i + \Delta z_{i+1}) - f(C^n_i) = \frac{dC^n_i}{dt} \quad (5)$$

Equation (5) is also written as

$$(D^n_{i+1}/\Delta z_{i+1} - u^n_{i+1}/2)\, C^n_{i+1} - (D^n_{i+1}/\Delta z_{i+1} + D^n_i/\Delta z_i + u^n_{i+1}/2 - u^n_i/2)C^n_i + (D^n_i/\Delta z_i + u^n_i/2)C^n_{i-1} = [(\Delta z_i + \Delta z_{i+1})/2][f(C^n_i) + \frac{dC^n_i}{dt}] \quad (6)$$

If $\quad D^n_i/\Delta z_i = u^n_i/2 \quad$ for any i $\quad (7)$

(6) reduces to:

$$u^n_i\, C^n_{i-1} - u^n_{i+1}\, C^n_i = [\Delta z_i + \Delta z_{i+1})/2][f(C^n_i) + \frac{dC^n_i}{dt}] \quad (8)$$

Therefore, a finite difference approximation to (3) coincides with a finite element approximation using a series of complete mixing cells under condition (7).

For the case where all Δz_i (i=1, ...N) are equal, (7) is written as

$$N = Pe^*/2 \quad (9)$$

where

$$Pe^* = Pe^*\ (z,t) = u^n_i\ L/D^n_i \quad (10)$$

Equations (9) and (10) imply that N is generally a function of time and space variables. Suppose we take an average value of Pe* over time for given soils, then N is determined by (9) and we can approximately simulate a degree of mixing. When the effect of dead pores is significant, it is possible to take such an effect into account in the present method as follows.

Each cell is further divided into two parts. One part is defined to have the same function described before. Another part is merely a completely closed dead space, as if the total accessible space were reduced by this dead space. Now (4) reduces to:

$$u_i C_{i-1} - u_{i+1} C_i = \varepsilon \Delta z [f(C_i) + \frac{dC_i}{dt}] \quad (11)$$

where

ε = correction factor to account for unaccessible space and $1 \geq \varepsilon > 0$

For practical applications, a soil system is divided into several soil horizons. The term soil horizon merely refers to a zone of soil which is considered to be homogeneous in terms of mixing characteristics. In this case each soil horizon is characterized by two parameters, Pe* and ε. These parameters could be determined by the use of a tracer. For instance, if the residence time frequency function $Q_e(t)$ (Uhl and Gray, 1966) is determined experimentally, the mean residence time τ can be calculated as

$$\tau_e = \int_0^\infty tQ_e(t)\ dt \quad (12)$$

The theoretical mean residence time τ_t can be calculated from the volume of pore space and the flow velocity, and ε is given as

$$\varepsilon = \tau_e/\tau_t \tag{13}$$

Also Pe* can be determined by comparing theoretical residence time frequency function $Q_t(t,Pe^*)$ with $Q_e(t)$. Once Pe* is known, one can calculate N from (9).

EXPERIMENT

Experiments were conducted using a lysimeter 91 cm in diameter and 183 cm in height (Iskandar and Nakano, 1978). The lysimeter was made of corrugated aluminum culvert to minimize possible water channeling. To avoid contamination of the soil solution from possible release of metals from the wall, the inside of the lysimeter was coated with an inert material. The lysimeter was filled with Windsor sandy loam obtained from local fields in Lebanon, N.H. The soils were packed carefully to a bulk density similar to field density. The soils consisted of three horizon -- 1st, 2nd and 3rd as shown in Table 1. The lysimeter was placed in a greenhouse.

Soil solution was obtained by the use of suction plates. Several 10.2 cm plates were placed at various depths and one 25.4 cm plate was placed near the bottom of the lysimeter. The vacuum system was designed to assure that the vacuum used to collect soil solution samples did not exceed a certain level and the vacuum was applied uniformly to all the suction plates.

Table 1. Physical Data for the Windsor Soils.

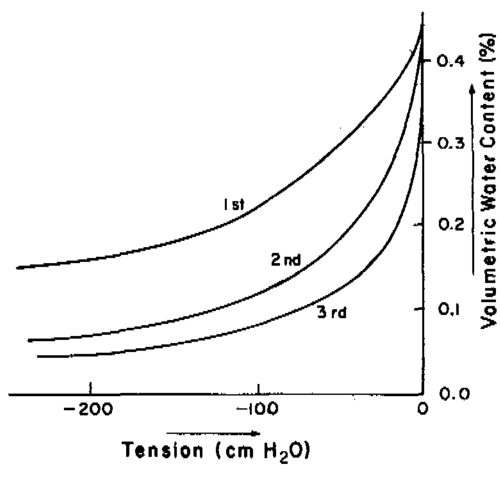
Horizon	Depth (cm)	Average bulk density (gm/cm^3)	Particle size analysis of soils (percent fraction)			
			Clay (<.001-.01) mm	Silt (.01-.075) mm	Fine sand (.075-.05) mm	Medium sand (.05-2.0) mm
1st	0-25	1.41	2	11	57	30
2nd	25-45	1.59	2	36	46	36
3rd	45-150	1.55	2	31	50	17

Soil temperature at depth was monitored by thermocouples connected to a data logger. The lysimeter wall was insulated with Styrofoam to prevent heat flux from the wall and the measured temperature indicated that the variation of temperature in a horizontal plane was negligibly small.

Moisture flow was monitored four ways: (a) by measuring the volumetric moisture content in the soil at depth using a neutron probe, (b) by measuring the soil moisture tension in the soil at depth using tensiometers, (c) by measuring the rate of water flowing out of the lysimeter by a tip bucket with a recording device, and (d) by measuring pan evaporation from a standard Class A pan situated near the lysimeter.

Since the hydraulic characteristics of a soil depend on its texture and structure, the desirability of measuring the characteristics in situ or in undisturbed samples has been recognized. The simultaneous measurement of volumetric moisture content and tension enabled us to determine the retention curve for each horizon as shown in Figure 2. Preliminary determination of

Figure 2. Retention curves for the three horizons of Winsor soils.



hydraulic conductivity was made from successive measurements of volumetric water content profiles (Rose et al., 1965). It was found that the hydraulic conductivity of all three horizons of soil may be conveniently expressed in terms of an exponential function of a polynomial of the volumetric water content as

$$K^i(x^i) = \exp [A_1^i P_1 (x^i) + A_2^i P_2 (x^i) + \dots] \quad (14)$$

where

K^i = hydraulic conductivity of ith horizon [cm/day]

x^i = nondimensional volumetric water content

$= \theta^i/\theta_s^i$

θ^i = volumetric water content of ith horizon

θ_s^i = saturated volumetric water content of ith horizon

A_n^i = constant

$P_n(s)$ = shifted and orthonormal Legendre polynomials

$$= \sqrt{2n+1} \sum_{r=0}^{n} (-1)^r \binom{n}{r} \binom{n+r}{r} x^r$$

The final adjustment of conductivity was made by the trial and error method, such that the computed water content profiles best fitted the experimental water content profiles. It was found that increasing more than the first two terms in (14) did not significantly improve the accuracy of computation. The final values of A_n^i are given in Table 2.

Table 2. Values of the Constant A_n.

Horizon	A_1	A_2
1	-6.498	-6.700
2	-3.4923	-5.328
3	-2.941	-3.579

In the computation a standard method for an implicit finite difference approximation to Darcy's flow equation was used. Although saturated flow appears for a relatively short period, the approximate method of an artificial transitional zone (Nakano, 1978) was used to treat the boundary value problem between saturated and unsaturated flow.

At the bottom of the lysimeter a constant water table was maintained. Water was applied to the soil surface by flooding. When surface water was present, the head pressure was accounted for in the computation. Since almost no vegetation existed at the time of the experiment, loss of water was solely due to evaporation from the surface, and an amount of water adjusted based on pan evaporation was taken away from the first segment of soil. Prior to each experiment the initial water content profile was measured with the neutron probe.

The predicted volumetric water content profiles generally agree reasonably well with the measured profiles and the error was found to be within about 10%. In Figure 3 the measured profiles after the application of 5.08 cm (2 in.) of water by flooding is shown. In Figure 4 the predicted accumulative leachate was plotted in comparison with the measured values. Agreement between the predicted and experimental values is quite good.

Having determined the hydraulic characteristics of the lysimeter, we proceeded to evaluate a numerical method simulating the movement of chemicals in soil solution. A certain amount of ammonium chloride was resolved in applied water and after the application the concentration of chloride and nitrate in soil solution was measured with time and at depth in soil.

RESULTS AND DISCUSSION

The main objective of the first experiment was to determine the number of division for each horizon that best simulates the actual mixing characteristics. 5.08 cm (2 in.) of water was applied regularly once a week. After the system became stationary, we applied 5.08 cm (2 in.) of NH_4Cl solution instead of water in one week (14 Jan. 1976). After that the soil solution was monitored during every application day for the following 11 weeks while 5.08 cm

Figure 3. Volumetric water content profiles after an application of 5.08 cm (2 in.) water.

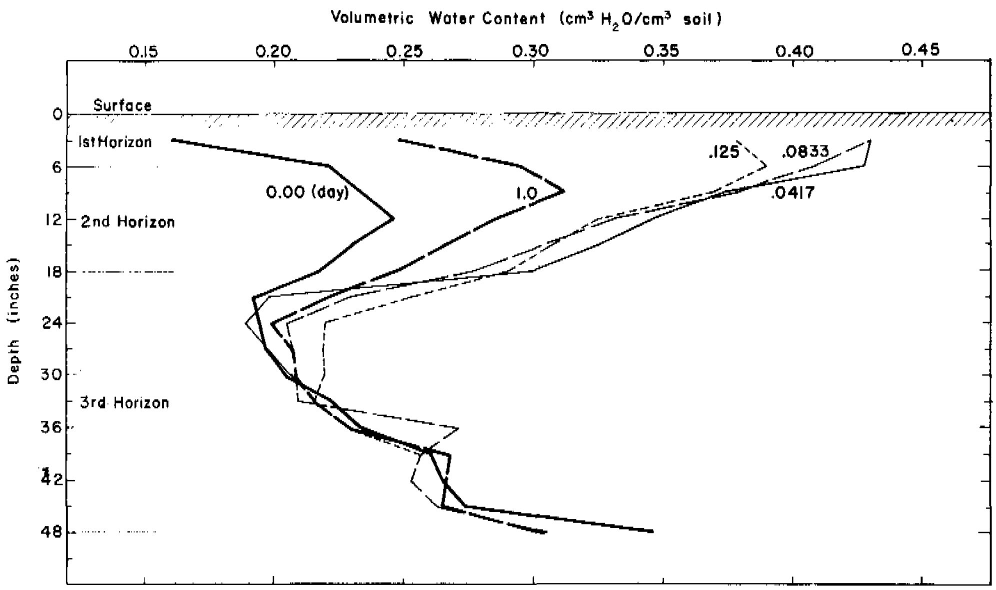
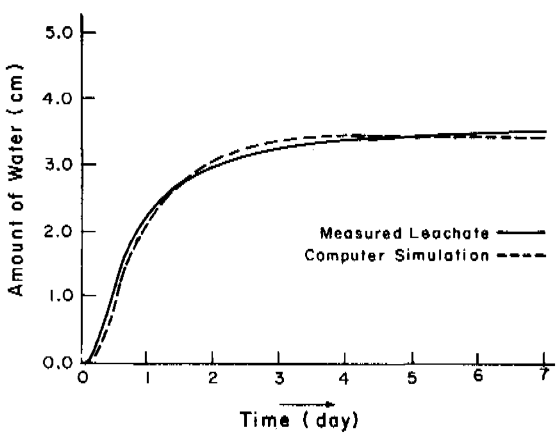


Figure 4. Experimental and predicted accumulative leachate after an application of 5.08 cm (2 in.) water.

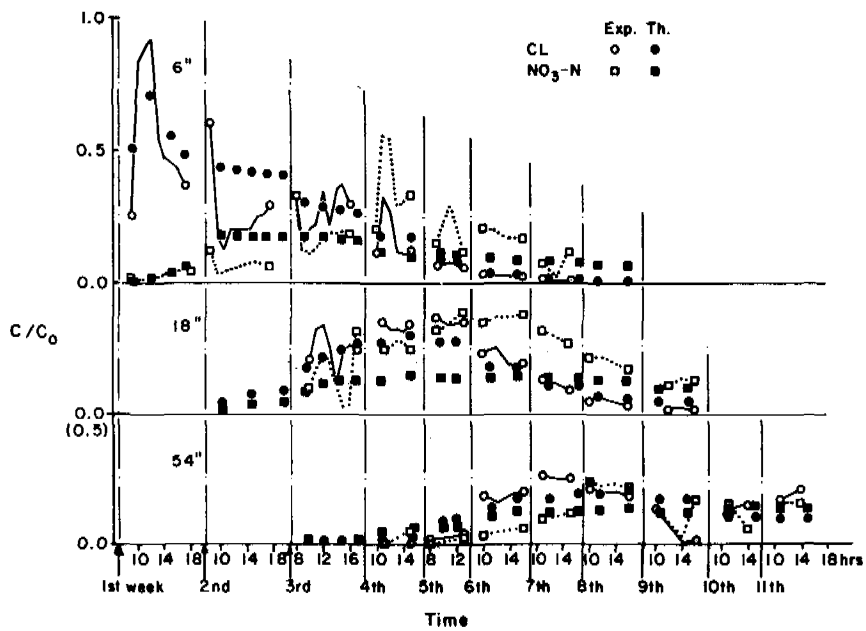


(2 in.) of water continued to be applied regularly once a week.

The results of the experiment are shown in Figure 5, where the concentration C/C_0 (C_0 is the initial concentration of applied NH_4Cl solution) of CL and NO_3-N is plotted against time at three depths in the soil. The arrows in the time axis indicate the time of application, and after each arrow about 8 hours of measured data were plotted. It is noted that the time axis is not continuous.

From the figure it is easy to see that dispersion of chloride in the 1st horizon was rather remarkable. Although the horizon is only 15.24 cm (6 in.) deep, it took almost 7 weeks for all chloride to flow out. After passing through the 1st horizon, chloride did

Figure 5. Measured and predicted concentrations of chloride and nitrate nitrogen in soil solution after the application of 5.08 cm (2 in.) ammonium chloride solution. An arrow in the time axis indicates time of application.

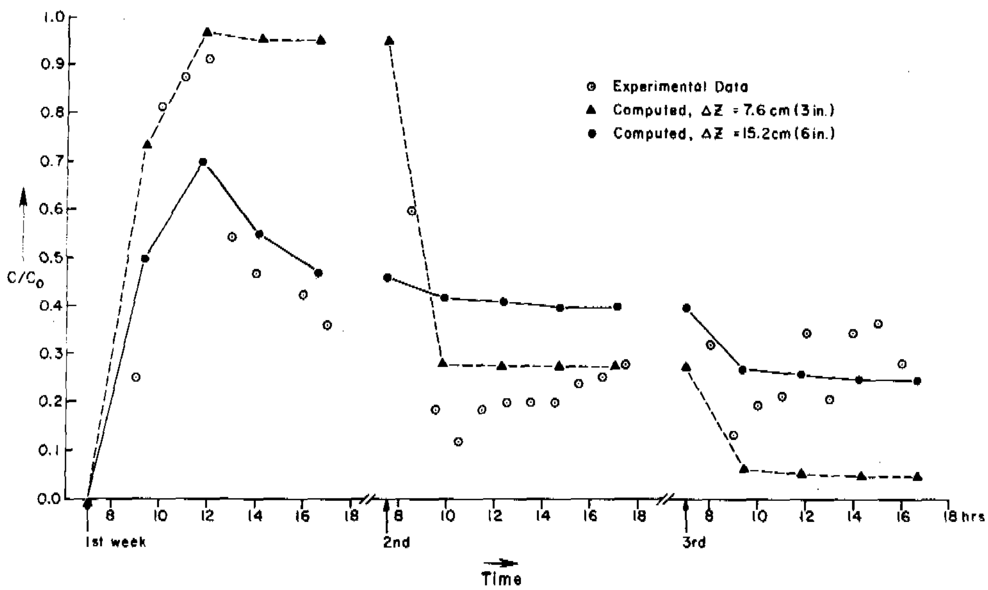


not seem to disperse noticeably as in the 1st horizon. Another interesting feature is that the variation of chloride concentration with time was the largest at 6-in depth, but it gradually became smaller while chloride passed through the 2nd and 3rd horizons. When 5.08 cm (2 in.) of chloride solution was applied by flooding, all the solution infiltrated into the soil within about 1-1/2 hours. Despite such a short time span of application, after passing through 54 in. of sand soils, chloride was so well dispersed that even the location of the peak chloride concentration is not easy to accurately determine. This feature is considered to be the combined effect of an unsteady flow field generated by successive but intermittent applications of 5.08 cm (2 in.) water and the bulk diffusivity of the soil.

Although nitrate can be produced from sources other than applied ammonium and is not as good a tracer as chloride, in view of the relatively large amounts of ammonium application we thought we might be able to draw some useful but approximate conclusions from the nitrate data. Therefore we combined the nitrogen model by Dutt et al. (1972) that includes nitrogen transformation due to mineralization-immobilization of ammonia and organic nitrogen, nitrification of ammonia nitrogen, and immobilization of nitrate nitrogen and we computed the nitrate concentration.

As a trial we used the uniform space increment Δz = 7.62 cm (3 in.) for all the horizons and computed the concentration of chloride. The results of computation at 6-in. depth are shown in Figure 6. The computed values underestimate a degree of mixing in the 1st horizon and chloride passes through almost twice as fast as in the experiment. For the next trial we used the uniform space increment Δz= 15.24 cm (6 in.) for all the horizons. With this increment the 1st horizon becomes one complete mixing cell. This increment was found the best fit for the 1st horizon, but introduced too much mixing for the 2nd and 3rd horizons. After another few trials we found that the selection of increments of 6, 3 and 3 in. for the 1st, 2nd and 3rd horizons respectively seemed to fit the experimental data best and the result of these computations was also plotted in Figure 5.

Figure 6. Comparison of the predicted concentration of chloride with the experimental data at 6-in. depth.



It is apparent that the present method predicts a rather smooth change of chloride concentration with time and fails to simulate the seemingly irregular and high frequency of measured variation of chloride. This difficulty is most pronounced in the 1st horizon. If the measurement was accurate, then the cause of such a difficulty must be related to the actual mechanism of mixing in the soils, particularly in the 1st horizon.

Although we assumed that complete mixing occurs in the 1st horizon, it is obvious that the actual mechanism of mixing is not like mixing with an impeller. One possible mechanism for producing a high frequency of measured variation of chloride would be mixing by channeling through macropores. Unfortunately we do not have enough experimental data to substantiate such a hypothesis.

Despite the difficulty in simulating the high frequency of variation, the method appears to simulate the general trend reasonably well. Particularly at the 54-in. depth where almost all high frequency variations die out, the predicted chloride concentration agrees well with the measured value.

The nitrogen model (Dutt et al., 1972) generally overestimates the nitrate concentration at the beginning and underestimates later. This difficulty appears to be inherent in the fixed order kinetics that does not account for such biological processes as dynamic population growth of nitrifiers. The experimental data indicate that production of nitrate is very slow in the first week after application of the NH_4Cl solution. This one week lag appears to correspond to the delayed appearance of the peak nitrate concentration a week after the chloride peak at both 18 and 54-in. depth.

The bulk diffusivity of the soil can be approximately determined. Assuming that L=140 [cm], N=15 and $u=2\times10^{-4}$ [cm/s], we estimate that the bulk diffusivity is on the order of 10^{-3} [cm^2/s] from (9) and (10). This is about 100 times greater than the molecular diffusivity of water and agrees with the value reported by Corey and Horton (1968).

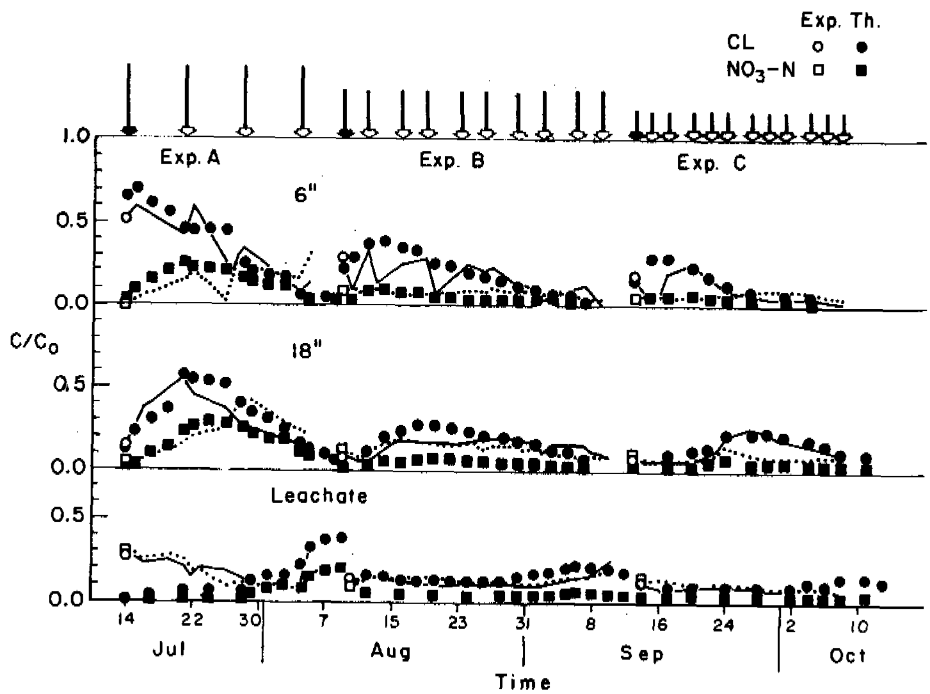
Having determined the number of division for each horizon, we conducted another series of experiments to examine the present method. In these experiments the application rate was set at 10.16 cm

(4 in.) a week, but the water was applied in three different ways within a week. In Exp. A, 4 in. of water was applied every Monday. In Exp. B, 2 in. of water was applied every Monday and Thursday, while in Exp. C 1-1/3 in. of water was applied every Monday, Wednesday and Friday.

In all three experiments the same amount of NH_4Cl was used and NH_4Cl solution was applied only once for each experiment on the specified Monday. The initial concentrations of NH_4Cl solution C_o were 33.7, 67.4 and 101.1 [meq./liter] for Exp. A, B and C respectively. After application of the solution, the soil solution was monitored every day while 4 in. of water continued to be applied in the specified way every week. During this series of experiments the soil pH decreased and remained at an average of 4.5 due to a large amount of NH_4Cl application. In order to account for the effect of pH on nitrification of ammonia we modified the rate constant according to Alexander (1961).

The results of the experiments and computations are shown in Figure 7. From the figure it is clear that the way of application within a week affects the movement of chloride in a rather complicated manner, because unsteady flow fields generated by three different ways of application differ significantly from each other. Although a large discrepancy occurs between the experiment and the simulation of the leachate concentration in Exp. A due to the effect of the previous experiment, the present model can simulate the general trend of chloride movement reasonably well.

Figure 7. Comparison of the predicted concentrations of chloride and nitrate nitrogen with the experimental data at 6-in. depth, 18-in. depth, and in leachate. An arrow indicates time of application and length of the arrow is proportional to the amount of water applied.



CONCLUSION

The present model can simulate the general trend of chloride movement reasonably well, but fails to simulate high frequency of variations, particularly at 6-in. depth. The nitrogen model (Dutt et al., 1972) tends to overestimate the nitrate concentration at the beginning and underestimate later. For the case where a solution containing nitrogen is applied intermittently, such as a few inches a day once a week, modification based upon dynamic microbial growth may improve the accuracy of the prediction.

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MATHEMATICAL MODELING

AN ADSORPTION MODEL FOR PREDICTION OF VIRUS BREAKTHROUGH FROM FIXED BEDS

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Groundwater comprises 95% of the U.S. fresh water supply. The potential for biological and chemical contamination of this supply by percolation from such sources as surface spreading of untreated and treated wastewater, sludge land-spreading, septic tanks and landfill leachates is very high. Previous laboratory and field studies give equivocal results regarding the magnitude of the threat of virus contamination of groundwater supplies which is posed by these waste disposal practices. Initial experimental and mathematical modeling efforts are described for the prediction of breakthrough of low levels of virus from percolating columns under conditions of both adsorption (application of wastewater to uncontaminated beds) and elution (application of clean water to contaminated beds). This breakthrough is described by the ion exchange/adsorption equations with the effects of external mass transfer and nonlinear adsorption isotherms included. Predictions are in qualitative agreement with reported observations from experiments which measured virus uptake by columns packed with activated carbon or a silty soil.

INTRODUCTION

In the United States, municipal raw sewage typically contains 7000 enteric virus particles per liter (1). The virus concentration of treated wastewater which is applied to land depends primarily on the degree of treatment of the municipal raw sewage. The first significant reduction occurs in the biotreating unit, about 90% for activated sludge treatment. Viruses are sequestered by the activated sludge biomass and most of those removed from the wastewater appear in the sludge phase. It is believed that little virus deactivation occurs in the activated sludge process; probably about 90% of the viruses sequestered by the biomass leave the unit as viable viruses. The primary defense against the release of virus in wastewater to the environment is disinfection, usually by chlorine addition. However, even the final effluent from this process which is the final product of most treatment facilities today, contains on the order of 0.1 to 1.0 virus per liter.

Therefore, the transmission and survival of enteric viruses in percolating soil beds are essential public health concerns in land waste disposal-groundwater supply cycles. Groundwater quality with respect to this type of contamination depends on virus concentration and state of suspension at the point of application, the chemical composition of the water, the physical and chemical characteristics and depth of soil bed and aquifer volume. The focus of this study is the development of a model to describe the movement of single virus macromolecules through soil beds where the initial conditions may correspond either to a process of adsorption, that is the application of virus-laden solution to an uncontaminated bed, or to a process of elution, that is the application of virus-free solution to a contaminated bed.

Table 1. Virus Transport Through Percolating Laboratory Columns

Adsorbent (Depth)	Virus (Influent Loading)	Solution	Flow Rate cm/min	Virus Removal Capacity	Ref.
Activated Carbon (4-25 cm)	T_4 phage (10^6 - 10^8 PFU/ml)	Distilled water and dissolved salts	0.5-130	Low removal, rapid breakthrough at lowest flow, instant breakthrough at highest flow	(2)
Activated Carbon (45 cm)	Type I polio (8.8×10^3 PFU/ml)	Filtered secondary effluent	4-8	Low removal, instant breakthrough, pH effect studied	(3)
Sterile Sand (20 cm)	f_2 phage and Type I polio (10^7 - 10^9 PFU/ml)	Distilled water and dissolved salts	0.08-0.16	Low removal in NaCl solns., high removal in $CaCl_2$, $MgCl_2$, 30-60 min. breakthrough	(4)
Sandy Forest Soil (20 cm)	T_7 phage & Type I polio (3-6.5×10^6 PFU, single application)	Deionized water	Continuous and intermittent Avg = 0.005	High removal, T_7 and polio differ under intermittent flow, 60 min. breakthrough	(5)
Sterilized Oahu Soils low humic (4-15 cm) volcanic cinder (15-38 cm)	Type II polio (1.5×10^5 PFU/ml)	Clear water	Intermittent Avg = 0.01	Humic soil, high removal (99+%) and fast breakthrough. Cinder low removal (<40%) and instant breakthrough	(6)
9 Soils (45-50)	T_1, T_2 and f_2 phages (2×10^7-10^8 PFU/ml)	Distilled water and dissolved salts	0.008-0.03	99+% removal, no breakthrough data	(7)

Table 2. Virus Transport through Percolating Field Soil Beds

Adsorbent (Depth)	Virus (Influent Loading)	Solution	Flow Rate cm/min	Virus Removal Capacity	Ref.
Sewage Infiltration site soils (5-28 meters)	f_2 phage, Type I polio, encephalomyocarditis (10^5 PFU/ml)	Settled sewage	0.06-0.6	48 hours to f_2 breakthrough in 28 meter well, enteric virus breakthrough in 72 hrs.	(8)
Sandy Soil (2-7 meters)	Enteric viruses (0.04 PFU/ml)	Secondary effluent spray irrigation, rainfall	0.005, higher w/rain	1% influent concentration found in 7 meter well after rainfall	(9)
Sandy Gravel (60 meters)	Type III polio virus	Secondary effluent	-	No virus breakthrough	(11)
Sandy Soil (3-9 meters)	Enteric viruses (.002-.075 PFU/ml)	Secondary effluent	0.02	No virus breakthrough	(12)

PREVIOUS STUDIES

Numerous laboratory and field studies have been conducted to determine removal capacities and rate of transport of viruses in percolating beds of various solids. Six recent laboratory investigations listed in Table 1 examined removal in shallow beds of activated carbon, sand and various soils. The viruses studied included a variety of bacteriophages and attenuated poliovirus strains. The poliovirus is usually regarded as a good model for studying enteric virus transport in wastewater treatment systems. Influent concentrations are elevated several orders of magnitude above typical levels for treated or untreated sewage. The first three studies listed were also conducted at relatively high flow rates compared to usual field percolation rates of about 0.01 cm/min, and only one study was done with a representative treatment plant effluent.

Activated carbon, Hawaiian volcanic cinder soil and sterile sand (when only NaCl is present in the virus solution) displayed low virus removal capacities. Columns packed with soils generally showed higher capacities for virus removal. Drewry and Eliassen (7) concluded from their results with three bacteriophages that percolation through only a few meters of the types of soils which they studied (five from San Mateo County, California, and four from Washington County, Arkansas) would be sufficient for protection from virus contamination.

Table 2 summarizes the results of four field studies in which the water from wells beneath sewage spreading grounds was sampled for virus contamination. The first study listed indicates virus penetration at 28 meters when primary treated effluent was seeded with the viruses listed and applied to land at a sewage infiltration site in Massachusetts (8). In a study of virus transport at the St. Petersburg, Florida, wastewater renovation project, Wellings, Lewis and Mountain (9) found that endogeneous enteric virus from a chlorinated secondary effluent had penetrated a 7 meter well. The largest virus plumes consistently appeared in the test wells following a rainfall at the test site. This observation supports the hypothesis that virus adsorption to soil is reversible and dependent upon the ionic character of the percolating water. In addition to these examples of virus isolation in the field, Mack, Lu and Coohon (10) have reported isolation of poliovirus type 2 from a 30-meter well located more than 100 meters from the edge of a wastewater drainfield in Michigan.

No viruses were found by Merrel and Ward (11) after 60 meters of percolation in the well-known Santee Project (San Diego) although the PFU assay procedure was not used in this study. Recently, complete virus removal by percolation through 3-9 meters of soil has been described at the Flushing Meadows (Phoenix, Arizona) wastewater renovation project (12). This study also involved the use of unseeded influents to the percolation bed.

This review shows that virus transport through packed beds is a complex function of several variables: permeate composition and rate of application, virus concentration as well as soil or packing type and depth. From these results, few additional quantitative, or even qualitative conclusions can be reached regarding effluent quality or groundwater protection.

VIRUS ASSOCIATION WITH WASTEWATER SOLIDS

Wastewater microbiological contaminates may be removed in a packed bed by filtration and/or adsorption. The percolation/adsorption model described below for virus removal is based on the premise that the movement of single virus particles is the principal determinant of bed effluent quality, and therefore the principal threat to groundwater quality. The initial experimental studies of this research involved the determination of the distribution of viruses in solutions containing suspended solids normally present in wastewater effluent from secondary treatment.

The fraction of viruses which associate with small particles was studied by performing batch adsorption experiments of Type 1 poliovirus to samples of suspended solids from the activated sludge unit of a local Los Angeles wastewater treatment facility. Solids concentration was varied from 0.02 mg/ml to 4.8 mg/ml. Virus concentration was varied from 10^2 to 10^5 infectious particles per milliliter as determined by the plaque-forming unit (PFU) assay (13).

Results of previous studies (14,15) of virus adsorption capacity of biological solids have been described by the empirical Freundlich isotherm $q = mC^n$ where q is the number of viruses adsorbed per unit mass of solids in equilibrium with a solution virus concentration C and m and n are empirical constants. These results show that for experiments in which C is varied at high solids concentration, $W > 1$ mg/ml, n is close to unity and m is approximately constant. However, there has been no correlation of these results with the measurements of adsorption in which solids concentration is varied at constant C and different values of m and n are obtained. Our results, which indicated a significant effect of solids concentration on adsorption capacity, were correlated by a modified isotherm in which n was again close to unity but m was a decreasing function of solids concentration,

$$m = (0.63 \pm .08) W^{-(0.81 \pm .03)}$$

This increasing capacity with decreasing solids concentration is probably caused by an increase in solids surface area due to solid aggregate dispersal.

The isotherm equation can be rearranged to form an expression for the ratio of free, unassociated virus to total virus in the wastewater as a function of solids concentration

$$\frac{C}{C_t} = 1/(1 + mW)$$

Figure 1 compares this ratio computed from our experimental determination of the coefficients m and n with the ratio computed using the isotherms determined by Sobsey, et al (14) and Clarke, et al (15).

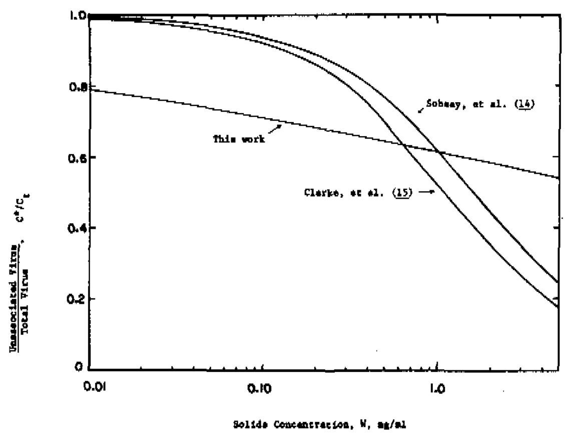

Fig. 1. Fraction of Unassociated Viruses in Equilibrium with Wastewater Solids

Our results indicate that at the low solids concentration level of a wastewater treatment effluent (about 10 mg/ℓ), 80% of the total virus concentration would be unassociated compared to about 98% calculated from the previous studies. At the high solids concentration typical for an activated sludge mixed liquor our results give an estimate of 57% unassociated compared with 30% from previous studies. These data are in qualitative agreement with the observations of Wellings, et al (16) which show that while the absolute number of viruses decreases as the degree of wastewater treatment increases, the ratio of free virus to solids-associated virus is increased.

PERCOLATION/ADSORPTION MODEL OF VIRUS TRANSPORT

The percolation/adsorption model consists of a virus mass balance, an adsorption rate, and an equilibrium isotherm relationship. These are shown in Figure 2 along with the simplifying assumptions and boundary conditions.

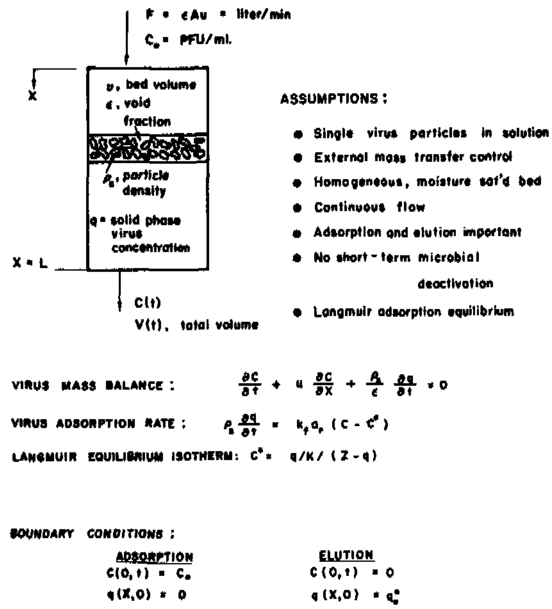

Fig. 2 Percolation/Adsorption Model

The unsteady-state virus mass balance for an infinitesimal volume element of the bed is the sum of a liquid-phase depletion term where C is virus concentration in the liquid, a convection term where u is liquid velocity, or percolation rate, through the bed, and a solid-phase accumulation term where q is virus concentration in the solid. The rate of virus adsorption is controlled by external mass transfer where k_f is the mass transfer coefficient and $(C-C^*)$ the concentration driving force. Although information on the equilibrium distribution of viruses between solution and solid phases is sparse, the equilibrium concentration C^* has often been fit to a Langmuir isotherm where K and Z are the Langmuir constants.

The solution of these three equations constitute a model which permits calculation of the concentration history, or breakthrough curve, of virus in the bed effluent, C(t). Two simple types of boundary conditions are considered. The simplest *adsorption* boundary condition corresponds to a solid phase which initially has no adsorbed virus and a bed influent at constant virus concentration. The simplest *elution* boundary condition corresponds to a solid phase which initially contains an amount of adsorbed virus q_o^* which would be the solid phase equilibrium concentration with a liquid phase at concentration C_o, and a bed influent containing no virus.

Analytical solutions of the adsorption model for several specialized cases are available in the literature (17,18). The specialized result which applies depends on the controlling mechanism, in this case external mass transfer, and on the nature of the equilibrium. This equilibrium behavior is characterized by the separation factor, r, defined as

$$r = 1/(1 + K\,C_o)$$

where K is the Langmuir isotherm constant and C_o is the influent virus concentration. Two limiting cases of adsorption column behavior called constant pattern and proportional pattern are shown in Figure 3.

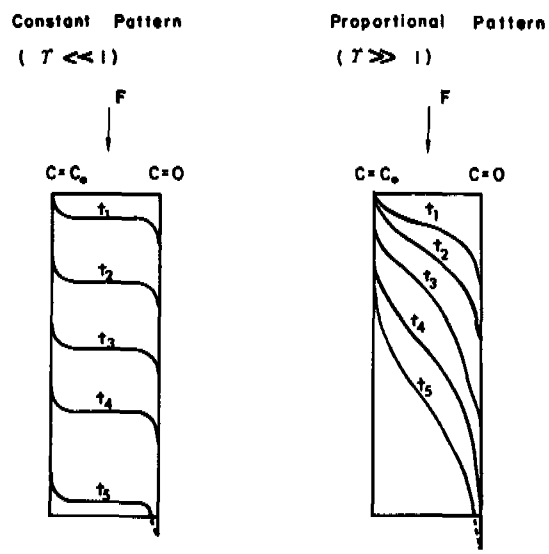

Fig. 3 Adsorption Column Behavior

For constant pattern, which occurs when r << 1, the liquid-phase virus concentration is shown, at early time t_1, to decrease as a sigmoidal function of bed depth from C_o near the inlet where the solid phase is saturated. Once formed, this total concentration profile moves down the bed without distortion as more of the solid-phase becomes saturated. Finally, at some later time t_5, the first virus begins to appear in the bed effluent. The objective here is to predict the time at which this low-level breakthrough occurs. When the separation factor is much larger than unity, the concentration profile changes shape as the run time of the adsorption operation increases. This proportional pattern behavior derives its name from the fact that bed effluent concentration is proportional to bed depth.

For elution from a saturated bed the separation factor, indicated by r^+, is the reciprocal of the factor for adsorption, $r^+ = 1/r$. That is, if constant pattern behavior is found to apply during adsorption, then proportional pattern will apply during elution.

Analytical solutions for these limiting cases are not too complicated and are available in the above cited literature. When equilibrium data was available, it was found that for the virus separations listed in the earlier tables, separation factors were usually in the range $0.5 < r < 2.0$. To calculate breakthrough curves for these cases requires a more general solution.

This general analytical solution to the percolation/adsorption model shown in Figure 2 is derived by redefining the virus adsorption rate as a reaction kinetic expression as suggested by Thomas (c.f., Vermeulen (18));

$$\rho_B \frac{\partial q}{\partial t} = \kappa\left[C\left(1 - \frac{q}{q_o^*}\right) - \frac{1}{K}\,(C_o - C)\,\frac{q}{q_o^*}\right]$$

The concentration driving force term in brackets includes the Langmuir isotherm and κ, called the reaction kinetic coefficient, is related to the external mass transfer coefficient by

$$\kappa = k_f a_p/[1 + (r-1)(q/q_o^*)]$$

where a_p is the effective mass-transfer area between the fluid and solid phases. As a consequence of assigning a constant value to the concentration dependent coefficient κ an error is introduced into the solution. Vermeulen (18) has shown that good results can be obtained when the solid-phase concentration ratio, q/q_o^*, is assumed to be one-half over the entire bed depth if the value of the adsorption separation factor r, is less than unity.

The solution for the separate cases of adsorption and elution are:

solution for adsorption

$$\frac{C}{C_o} = \frac{J(rS,T)}{J(rS,T)+(1-J(S,rT))\exp[(r-1)(T-S)]}$$

solution for elution

$$\frac{C}{C_o} = \frac{1-J(rS,T)}{1-J(rS,T)+J(S,rT)\exp[(r-1)(T-S)]}$$

where

$$J(\alpha,\beta) = 1-e^{-\beta} \int_0^{\alpha} e^{-\zeta} I_o(2\sqrt{\beta\zeta})d\zeta$$

$$S = \kappa\epsilon\nu/F$$

$$T = (\kappa\epsilon/\rho_B Z)(r/K)(V-\nu\epsilon)/F$$

The dimensionless parameters S and T are known as the column-capacity parameter and throughput parameter respectively. Vermeulen (18) gives tabulations and limiting values of the function J of two variables α and β where I_o is a modified Bessel function of the first kind.

VIRUS BREAKTHROUGH CALCULATIONS

The general solution of the percolation/adsorption model is now compared with Cookson's (2) measurements of T_4-phage virus adsorption from electrolyte solutions to activated carbon. The Langmuir isotherm parameters are obtained from Cookson's batch equilibrium measurements shown in Figure 4.

These data were obtained by measuring the equilibrium uptake of T_4-phage from electrolyte solutions of known initial virus concentrations by known masses of activated carbon. At equilibrium, the number of viruses remaining in solution is C^* and the number of viruses adsorbed per milligram carbon is q. The data agrees well with the Langmuir isotherm model and readily permits evaluation of the adsorption constants with good precision. From a least squares evaluation, $Z = 1.6\text{x}10^9$ virus adsorption sites per mg carbon and $K = 4\text{x}10^{-7}$ ml per virus. These equilibrium constants permit evaluation of the adsorption separation factor and the breakthrough curves for the fixed bed experiments performed by Cookson.

The computed breakthrough curves for the experimental conditions at which the highest and lowest efficiency in virus removal were observed in the fixed bed adsorption columns are shown in Figure 5. In both fixed-bed experiments the columns were packed with the same activated carbon on which the equilibrium measurements were made. The influent T_4-phage concentration was about one million viruses per milliliter. The separation factor at this concentration is 0.7.

The high efficiency experiment was performed in a short bed, L = 7.5 cm, at a low influent flow rate, u = 0.0633 cm/s with small-grain activated carbon packing. A single measurement of virus concentration in bed effluent was reported, but the time or amount of solution passed through the bed when the measurement was made were not given. This observed concentration was less than 2000 viruses per ml ($C(t)/C_o$ less than 0.002), which was the lower limit of the assay technique used. The calculated breakthrough curve indicates that after enough virus solution has passed through the bed to displace one bed volume, the effluent virus concentration should be about 10,000 PFU/ml, ($C(t)/C_o \sim 0.01$ on the figure). The remainder of the breakthrough curve with increasing percolation time is as shown to an effluent concentration of 10^5 PFU/ml after about 75 hours.

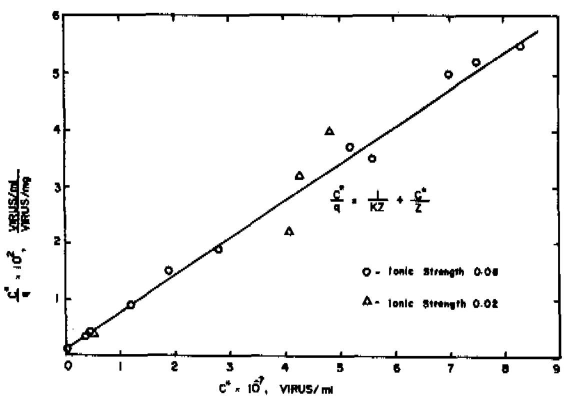

Fig. 4 Langmuir Adsorption of T_4-phage to Activated Carbon

The low efficiency experiment was performed in a longer bed at a higher influent flow rate with large-grain carbon packing. Both the calculated curve and the single experimentally-determined effluent concentration indicate rapid breakthrough of virus from the bed, although the calculated curve is again higher than the observed value.

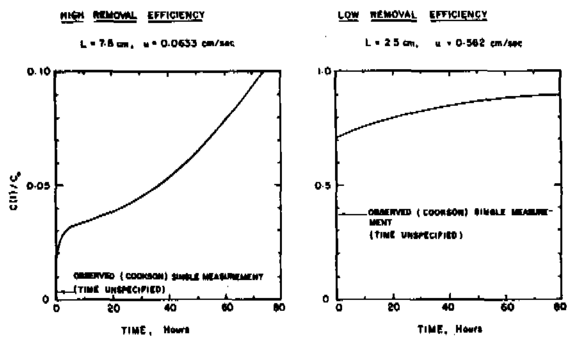

Fig. 5 T_4-Phage Breakthrough Prediction from Activated Carbon Beds. $C_o = 10^6$ PFU/ml, r = 0.7

Equilibrium adsorption data of virus to soils is scarce. Drewry and Eliasson (7) studied the adsorption of f_2-phage to an Arkansas soil and as indicated in the above summary table, they concluded that this silty soil offered high resistance to virus penetration. The equilibrium data for this study were not very good, and statistically the choice of a mathematical function to describe the isotherm was quite arbitrary. The fit of the data to a Langmuir isotherm is shown on Figure 6. The two-sigma confidence limits on the isotherm coefficients, K and Z, are very large. This equilibrium data was used to calculate the breakthrough curves from a fixed-bed of this same soil as shown on Figure 7.

The result of the adsorption breakthrough calculation for a shallow depth of this Arkansas soil at a high percolation rate shows that the bed can be run for almost one year before large quantities of virus begin to appear in the bed effluent. This high adsorptive capacity is consistent with the experimental observation of a high resistance to virus transport through laboratory columns of these soils.

The elution breakthrough calculation shown at the bottom of Figure 7 has not been observed experimentally but shows an interesting result. The *elution breakthrough calculation assumes* that initially the soil bed is saturated with virus in equilibrium with a solution at virus concentration C_o and that the influent composition and percolation rate are exactly the same as during the adsorption cycle except that it does not contain virus. The breakthrough curve shows that for these conditions, a long

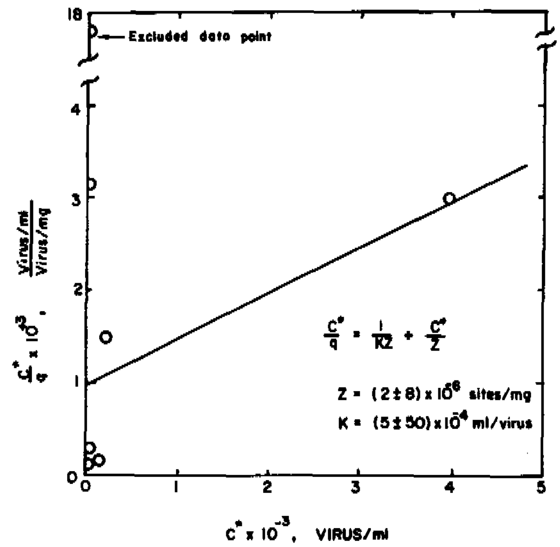

Fig. 6 Langmuir Adsorption of f_2-Phage to Arkansas Soil

time of about 5000 hours will be required before soil bed eluate concentration begins to decrease. Practically, this long purge time may not be too important since virus deactivation mechanisms would probably be significant contributions to the decrease of the number of infective particles in the bed eluate.

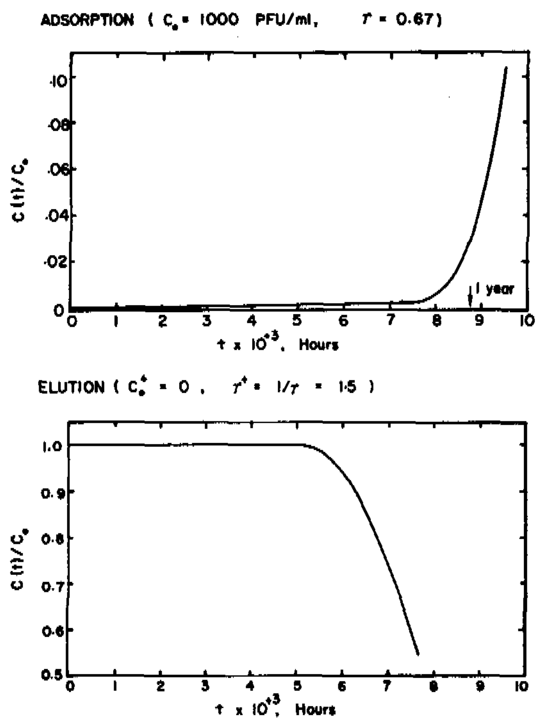

Fig. 7 f_2-Phage Breakthrough Prediction for Elution/Adsorption with Arkansas Soil. L = 2.3 cm, u = 0.05 cm/s

CONCLUDING REMARKS

Economic and operational optimization of sewage treatment facilities can best be achieved when the effect of natural processes on their effluents is known. Where landspreading of wastes is the disposal method, it has been shown that previous studies give equivocal results regarding the protection of water supplies from virus contamination. This paper is a first attempt to evaluate the concentration levels at which viruses in these effluents may be introduced safely to the environment. The percolation/adsorption model indicates that some solids such as activated carbon would be far less efficient for virus removal than some soil systems. This conclusion is in qualitative agreement with experimental observations reported in the literature.

ACKNOWLEDGMENTS

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USA-USSR WORKING AGREEMENT ON UTILIZATION OF WASTEWATER ON LAND

The International Symposium is a result, in part, of a cooperative program on Project 1-1, "Planning and Preparation of Measures for Rational Utilization of Water Resources" and Section I-I.3, "Uses of Industrial, Agricultural and Domestic Wastewater for Irrigation Purposes," between the USA and USSR.

In June 1977 Dr. Harlan McKim, Program Manager for the Wastewater Management Program, USACRREL, and A.F. Ponomarev, Head of Soviet Specialists, signed a protocol at CRREL agreeing that an international symposium on land treatment of wastewater would be held which would compile the state of knowledge on this subject. At this meeting the Soviet delegation presented the American delegation with 36 articles. The following nine Soviet articles have been translated for inclusion in this Symposium.

TREATMENT OF PIG OUTFLOW BY ALGOLIZATION FOR IRRIGATION USE

PODGOTOVKA SVINOSTOKOV PUTEM IKH AL'GOLIZATSII S TSELBYU ISPOL'ZOVANIYA NA OROSHENIYE

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Kupavna, Ministry of Amelioration and Water Utilization, All Union Scientific Research Institute for Utilization of Waste Water in Agriculture, 1975.

Translated by Office of the Assistant Chief of Staff for Intelligence for the Office, Chief of Engineers, 1978.

Under the conditions of the expanded conversion to industrial methods in the production of animal husbandry goods, animal care technology and the nature of manure accumulation and utilization change substantially. As a result of the high concentration of bedless cattle raised in stalls the animal husbandry complexes accumulate large quantities of manure outflow which must be treated, decontaminated, and used. Animal husbandry outflow contains the agents of salmonellosis, tuberculosis, and some leptospiroses [7]. Penetrating into the soil, most of them retain their viability from several months to 1 year. They may migrate in the water and penetrate into plants and animal organisms and, through the milk and meat (in the absence of heat processing), in the human body. Consequently, the sanitary-epidemiological danger of animal husbandry outflow is entirely obvious. Therefore, such outflow must be decontaminated prior to its utilization on irrigation fields.

In our view the decontamination of the liquid phase of animal husbandry outflow would be very effective in natural ponds and, particularly, in biological oxidation contact stabilization (BOCS) ponds which offer unquestionable advantages in terms of effect of decontamination, economy, simplicity of construction, and operational reliability, compared with other purification methods currently used [3, 6].

The microalgae play the basic decontamination function in BOCS ponds. According to the Provisional Recommendations [2], formulated and issued in 1971, they are set in the ponds before the latter become operative. Thanks to this we note in the ponds a total elimination of the group E. Coli bacteria and of the tuberculosis agents. Extensive studies of the effectiveness of the biological decontamination of industrial-residential waste water in BOCS ponds have indicated that the microalgae used in the inoculation of the waste water are greatly adaptable to changes in nutrition conditions and are extremely undemanding concerning nitrogen and carbon sources. It may be assumed, therefore, that cultivated in animal husbandry outflow as well they would be supplied with their basic nutritional elements to an adequate extent. However, since animal husbandry outflow contains a number of components which oxidize with difficulty, as well as specific compounds which are the metabolic products of the animal organism, the growth and development of the microalgae may be inhibited.

For this reason the purpose of this study was the development of the type of technology for the treatment of animal husbandry outflow which would guarantee its decontamination in BOCS ponds. This involved the use of a number of technological methods which are currently extensively used in treatment systems: outflow precipitation, aeration, or dilution. Hog outflow was precipitated in the course of 1 month at a temperature of 18-20°C with subsequently hourly aeration; the dilution involved the used of industrial-fecal waste water in a ratio of 1:1 and 1:3. The hog outflow thus treated was poured into algologic reactors and inoculated with an aerobic culture of microalgae. The experiments took place under 24-hour fluorescent light and a 20°C temperature and lasted 15 days.

The following was determined in the course of the experiments: amount of microalgae productivity, effectiveness of biological decontamination (elimination of group E. Coli microbes in the outflow), and concentration of organic pollution (five of biochemical and chlorine oxygen requirements, nitrogen, ammonium salts, pH, and transparency).

The result of the experiments of cultivating the aerobic suspension of phytoplankton in hog outflow subjected to additional preliminary treatment indicated that a very effective growth of microalgae is possible (Table 1).

With an unquestionable increase in the productivity of microalgae a considerable reduction of bacterial pollution could be expected as well (Table 2).

The hydrochemical indicators (Table 3) correlated entirely with the results of hydrobiological and bacteriological studies.

Table 1. Growth of the Biomass of Microalgae Cultivated in Hog Outflow.

Variant	Exposure, days	Raw weight, mg/liter
Hog outflow subjected to mechanical sedimentation and aeration	0	100
	7	144
	10	185
Hog outflow diluted with industrial-fecal waste water in a 1:1 ratio	0	100
	3	115
	15	200
Hog outflow diluted in a 1:3 ratio	0	100
	3	120
	15	235

Table 2. Effectiveness of Elimination of E. Coli.

Variant	Exposure, days	Percentage of disappearance of E. Coli.II
Hog outflow subjected to mechanical sedimentation and aeration	0	0
	7	72
	15	98.8
Hog outflow diluted with industrial-fecal waste water in a 1:1 ratio	0	0
	12	77
	15	100
Hog outflow diluted in a 1:3 ratio	0	0
	12	98.2
	15	100

Assessing the methods used as technological means for treating hog outflow for the cultivation of aerobic suspensions of microalgae, let us note that both could be used. Clearly, both the method of mechanical precipitation of outflow with subsequent aeration, as well as dilution with industrial-fecal

Table 3. Hydrochemical Indicators of Self-Purification Processes

Variant	Exposure, days	Indicators: pH	Transparency, cm	Biochem O_2 requirements, mg/liter	Chlorine requirements, mg/liter	NH_4 mg/liter
Hog outflow subjected to mechanical sedimentation and aeration	0	7.3	5	345	480	69
	7	8.4	7	155	309	31
	15	8.9	10	75	202	18
Hog outflow diluted with industrial-fecal waste water in a 1:1 ratio	0	7.6	5	212	552	43
	12	8.6	8	93	322	15
	15	9.0	11	20	180	5
Hog outflow diluted in a 1:3 ratio	0	7.9	9	200	620	40
	12	8.9	9	60	224	12
	15	9.2	12	15	118	3

waste water lower the concentration of matter difficult to decompose in hog outflow as well as the specific results of the metabolic activities of the animal organism, inhibiting the growth and development of the phytoplankton.

A comparison of results shows the higher productivity of microalgae in hog outflow diluted with industrial-fecal waste water. Bacteriological self-purification processes developed more intensively in accordance with the rates of growth of the phytoplankton mass in the diluted outflow.

This is obviously due to the specific nature of industrial-residential waste water which, as studies have indicated [1, 4, 5], contains a practically complete set of micro- and macroelements insuring the normal life of microalgae. Obviously, this affected subsequent self-purification processes of hog outflow in concentrated algolization and was accompanied by the formation of physiologically active substances stimulating the growth and development of autotrophic microorganisms.

Obviously, on the basis of the acquired practical experience in building animal husbandry complexes around big cities and settlements [6], the technology of decontamination of hog outflow through biological methods should stipulate, above all, its preliminary dilution with industrial-fecal waste water.

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USA-USSR WORKING AGREEMENT ON UTILIZATION OF WASTEWATER ON LAND

USE OF LIQUID FRACTION OF BEDDINGLESS MANURE AT THE VORONOVO ANIMAL HUSBANDRY COMPLEX

ISPOL'ZOVANIYE NA OROSHENIYE ZHIDKOY FRAKTSII BESPODSTILOCHNOGO NAVOZA ZHIVOTNOVODCHESKOGO KOMPLEKSA "VORONOVO"

V.I. Dmitriyeva and N.A. Lapshina

Kupavna, Ministry of Amelioration and Water Utilization, All Union Scientific Research Institute for Utilization of Waste Water in Agriculture, 1975.

Translated by Office of the Assistant Chief of Staff for Intelligence for the Office, Chief of Engineers, 1978.

The successful conversion of animal husbandry to an industrial base requires the operative solution of the problems of removal, processing and utilization of considerable quantities of animal husbandry outflow. A complex raising 70,000 hogs simultaneously produces about 3,000 cubic meters of outflow per day. Frequently big livestock farms become sources of pollution of the surrounding territory, rivers and water bodies.

One of the efficient means for the utilization of liquid manure is its use as crop fertilizer. In turn, this would contribute to the creation of its solid fodder base and to upgrading the effectiveness of the complexes. According to domestic and foreign researchers in terms of fertilizing qualities liquid manure is as good as conventional bedding manure and, in many cases, superior [1-10]. The effectiveness of liquid manure is determined, above all, by its high nitrogen content in its ammoniacal form, easily accessible to the plants.

Extensive experience in the use of liquid manure has been acquired in Switzerland, Austria and the GDR. Here application doses are based on determining the extent of removal of nutritive elements by the plants from the soil and the amounts of nutritive substances contained in animal droppings. Experiments conducted in the GDR on the study of the effectiveness of maximum liquid manure norms (400 kg per hectare of nitrogen) on the sizes of various farm crops showed [8] that it could be used productively. For example, with such a dose of nitrogen potato yields reached 300 quintals per hectare.

It was established (GDR) that it would be more expedient to apply beddingless manure to corn and feed cabbage, undiluted with water, in doses of 20-30 cubic meters per hectare; to beets and potatoes, 10-20; and to perennial grasses, up to 30 cubic meters per hectare. The extent of liquid manure water dilution depends on the season. In spring it is recommended to dilute it (depending on the crop) in a 1:3-1:5 ratio; in the summer -- 1:8-1:20. Here we must take into consideration the ratio among nutritive substances in liquid manure. The addition of the deficient component through chemical fertilizers considerably upgrades its effectiveness. Consequently, beddingless manure is an addition to organic fertilizer and its utilization will enable us to increase farm crop yields and systematically keep the soil at a certain soil fertility level.

However, beddingless manure is scarcely used as fertilizer because of the inadequate development of technological methods for its preparation and utilization as organic fertilizer. Furthermore, under the existing methods for the removal of livestock excrements from the premises through washing the manure's moisture rises to the 98% level. This considerably increases the amount of manure outflow and calls for its separation and utilization on a year-round basis.

Table 1. Changes in Outflow Structure at Time of Irrigation, mg/liter.

Outflow type	pH	HCO_3	General nitrogen	Ammoniacal nitrogen	BOD
Outflow from hydrant	6.85	4800	1054	665	7830
Outflow after sprinkling	7.25	4500	989	602	6550

One of the promising methods for the utilization of manure outflow is for the irrigation of farm crops. However, so far this problem has been insufficiently studied. The purpose of our studies was the following: to determine the suitability of the liquid fraction of cattle manure outflow for the irrigation of farm crops based on land reclamation indicators, and to study irrigation influence on farm crops and soil fertility. The experiments were conducted at the Voronovo Sovkhoz in Podol'skiy Rayon, Moscow Oblast. The soil of the plots was soddy-podzolic and heavy loam. The agrochemical characteristics of the soil are shown in Table 1.

Due to the fact that the irrigation network and field manure containers were in the construction stage, experimental-industrial irrigation with animal husbandry outflow was accomplished with the help of a temporary irrigation network consisting of the RT-180 model type. Irrigation was used on perennial first-year grasses. The outflow was fed from the first container of the horizontal sedimentation collector with the help of the mobile SNP-50/80 pumping station. The distribution of the waste water on the area was accomplished with a DDN-45. The hydrants for the collection of the liquid were placed in such a way that no liquid could fall on the adjacent irrigated section. Before irrigation sample soils were taken to determine the agrochemical characteristics of the soil. The animal husbandry outflow application norms were based on the content of basic nutritive elements in the outflow and their removal with the planned perennial grass crops.

The composition of the outflow was determined on the basis of all reclamation indicators: color, pH, suspended sedimentation content, thick and hardened residue, HCO_3, SO_4, Cl, Ca, Mg, Na, K, P_2O_5, general and ammoniacal nitrogen, and BOD. The computations were based on Yu. Yu. Lur'ye's method, as modified by the water and soil chemistry laboratory of the VNIISSV. Changes in the composition of the outflow and the time of irrigation were determined with their applications with a sprinkling system.

The animal husbandry outflow of the Veronovo Sovkhoz contains great quantities of suspended sediment -- up to 14,000 mg per liter; organic substances (bichromatic oxidizing) reach 19,000 mg per liter. Characteristic of such waste water is a low alkaline reaction, close to neutral, a high concentration of soluble matter (thick residue) 8300 mg per liter, half of which consists of organic matter (as confirmed by the relatively low content of solidified residue of 3500 mg per liter), and a clearly expressed bicarbonate structure. The bicarbonates are represented mainly by nitrogen, potassium, and calcium salts. Chlorides and sulfates are preset in significant quantities. The animal husbandry outflow has high fertilizing value (the nitrogen content reaches 1500 mg per liter; the potassium content reaches 940, and the phosphorus content 620 mg per liter), and could be successfully used for fertilizer irrigation of farm crops.

The studies indicated that in the course of the year the composition of the outflow changes substantially. Particularly great fluctuations were noted in the content of the suspended sediment, the organic matter, and the nitrogen, potassium, and phosphorus.

The changes in the outflow structure at the time of irrigation are determined by the presence of a large quantity of nutritive substances in a very mobile form. Thus, the content of ammoniacal nitrogen reaches 700 mg per liter and it is volatilized essentially as ammonia. Table 1 shows the changes

Table 2. Changes in Soil Agrochemical Characteristics in Irrigation with Animal Husbandry Complex Outflow (Spring Irrigation).

Genetic level, cm	Organic matter, %	pH (KCl)	Hydraulic acidity (mg equivalent per 100 grams soil)	Total absorbed alkali (mg equivalent per 100 grams soil)	Absorption capacity (mg equivalent per 100 grams soil)	Level of alkaline saturation	Hydrolytic nitrogen (mg per 100 grams soil)	P_2O_5 (mg per 100 grams soil)	K_2O (mg per 100 grams soil)
Initial soil condition									
An 0-10	2.8	6.4	0.9	31.0	31.9	97.0	5.4	100.0	27.9
An 10-20	2.8	6.2	1.96	21.5	23.46	91.5	4.44	79.2	27.1
A_1A_2 20-30	0.63	4.05	3.74	7.4	11.11	66.5	2.8	9.0	14.2
A_2 30-40	0.44	3.65	5.04	8.6	13.64	64.0	2.5	3.0	11.1
A_2B 40-50	0.59	3.25	6.65	13.0	19.65	66.1	3.4	7.0	7.44
After irrigation with 300 cubic meters per hectare on 18 May 1973									
An 0-10	3.21	6.6	0.87	31.2	32.07	97.2	-	150.0	27.6
An 10-20	3.16	6.0	2.1	26.4	28.5	92.5	-	110.0	27.0
A_1A_2 20-30	1.73	5.7	2.1	15.8	17.9	88.3	-	90.0	16.3
A_2 30-40	1.16	4.35	2.8	12.4	15.2	81.5	-	12.0	9.84
A_2B 40-50	0.69	3.85	2.65	13.0	15.65	83.0	-	11.5	7.68

in the composition of the outflow at the time of irrigation.

Table 1 shows that the structure of the outflow changes substantially at the moment of irrigation: the content of general and ammoniacal nitrogen and hydrocarbonates declines and the reaction of the medium becomes alkaline. Thus, the loss of ammoniacal nitrogen from sprinkled outflow equals 63 m per liter or 10% of the initial composition; the content of hydrocarbonates declined by 7% while BOD declined by 16%.

Irrigation with animal husbandry outflow has a positive influence on soil fertility (Table 2). As a rule, following waste water irrigation, the content of organic matter rises; absorption capacity and level of alkaline saturation rise while acidity declines somewhat. The nutritive system improves with wastewater irrigation: the content of nitrogen, phosphorus and potassium accessible to the plants rises. Particularly clear differences in the content of the individual elements were manifested in the upper strata.

Compared with the control group irrigation of perennial grasses with outflow increases their yields. Yields are not increased only with full norm wintertime irrigation (300 cubic meters per hectare). This is explained by the partial perishing of the plants caused by the ice cover. In the variants irrigated in winter at the rate of 150 cubic meters per hectare grass survivability was higher compared with the application of the full norm since no ice covering was formed.

Following the first mowing of perennial grasses the field was replowed because of the thinning of the grasses and seasonal grasses were planted (mixed peas and oats). The seasonal grasses were not irrigated but the aftereffect of animal husbandry outflow was determined.

The fertilizing effect of animal husbandry outflow was manifested on the second crop as well (vetch-oat mix). A higher aftereffect of the spring irrigation was achieved, since in the short period between the application of the waste water and the harvesting of the perennial grasses, naturally, the nutritive substances contained in the outflow were not fully used which influenced the following crop yield.

Throughout the vegetation period, thanks to the irrigation with animal husbandry outflow (Table 3) additional crops were grown by the sovkhoz averaging 1570 fodder units per hectare. These data confirm the high fertilizing value of the clarified fraction of cattle outflow of the Voronovo complex used for irrigation. Irrigation with waste water has a positive influence on the fertility of soddy-podzolic soils and on crop yields.

Table 3. Influence of Animal Husbandry Outflow Irrigation on Crop Yields.

Variant	Green mass yield, quintals/ha		Total fodder units	Addition to control figure, fodder units
	Perennial grasses	Pea-oat mix		
Control without irrigation	139	222	6330	-
Winter outflow irrigation, 300 m^3/ha	137	272	7090	760
Winter outflow irrigation, 150 m^3/ha	202	252	8070	1740
Spring outflow irrigation, 300 m^3/ha	187	300	8540	2210
Average				1570

The use of animal husbandry outflow for irrigation contributes not only to higher fodder crops but to the protection of water sources from pollution.

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USA-USSR WORKING AGREEMENT ON UTILIZATION OF WASTEWATER ON LAND

NEUTRALIZATION OF ORGANIC SUBSTANCES IN WASTE WATER BY PLANTS

OBEZVREZHIVANIYE ORGANICHESKIKH VESHCHESTV STOCHNYKH VOD V RASTENIYAKH

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The question of the purification of waste water is becoming ever more important in connection with the fast development of the chemical industry. Artificial biological irrigation with the help of water microorganisms is the most widespread method for the purification of waste water contaminated by organic matter. Thanks to the existence of a great variety of microorangisms in the soil, irrigation fields could also be used as a natural method for the biological purification of waste water. Here, along with the protection of water reservoirs from pollution, we resolve the problem of obtaining high crop yields. The danger arises, however, that the organic matter coming from the waste water may accumulate in the plants.

The works of Yu. V. Rakitin at the Institute of Plant Physiology of the USSR Academy of Sciences, as well as a number of other works published domestically and abroad, have proved that, penetrating the plants, herbicides and other phytotoxic substances may disturb one or another metabolic process in the course of which, however, they themselves become detoxified. Only in rare cases do compounds of higher toxicity develop as a result of the toxic substances introduced into the plants through biochemical processes. However, should the plant survive, the existence of new toxic substances is only temporary, for, involved in the metabolic processes, they inevitably lose their toxicity.

The VNIISSV (L. Ye. Kutepov and S.I. Khramova) has studied the neutralization of organic matter in the plant using the highly sensitive method of gas chromatography with a flame-ionizing detector.

It was determined that plants play an important role in the biological neutralization of organic matter in the utilization of waste water on irrigated fields. The organic substances introduced with the irrigation water were neutralized in the plants, depending on their chemical structure, over different time periods ranging from 1 to 16 days (Table 1). As a rule, the neutralization process in root and tuber crops developed somewhat more slowly than in the aboveground part of the plants. That is why it would be desirable to restrict the use of waste water for irrigation to root and tuber crops.

Aniline, acetone, benzaldehyde, dimethylamine, diethylamine, crotonaldehyde, furfural, cyclohexanol and cyclohexanone were totally neutralized in the plants within a 5-day period; acetaldehyde, benzene, butanol, p-xylene, methanol, n-propanol, toluene, and ethanol were neutralized in 5 to 10 days; caprolactam, dichlorethane, and carbon tetrachloride were neutralized in 10 to 16 days.

In the presence of such organic substances in waste water used for irrigation we must observe the quarantine period between the last watering and the harvest ranging from 2 to 4 weeks depending on the nature of waste water pollution.

Name of substance and chemical formula	MPC b.p.s.* mg/liter	Concentration in irrigation water, mg/ℓ	Irrigation norm	Plant type	Period of total decomposition in plant, days
1	2	3	4	5	6
Aniline $C_6H_5NH_2$	0.5	300	50	Corn	1
				Grasses	1
Acetaldehyde CH_3CHO	1000	100	100	Corn	8
				Grasses	8
				Potatoes (tubers)	3
				Potatoes (plants)	4
Acetone CH_3COCH_3	800	300	50	Corn	2
				Grasses	2
				Leguminous grasses	2
Benzaldehyde C_6H_5CHO	-	100	100	Corn	3
				Grasses	3
				Potatoes (tubers)	6
Benzene C_6H_6	100	100	70	Corn	5
				Grasses	4
				Carrots (root crop)	6
Butanol $CH_3(CH_2)_2CH_2OH$	420	100	70	Corn	6
				Grasses	3
				Celery (plant)	7
				Celery (root)	8
Dimethylamine $(CH_3)_2NH$	0.7	300	70	Corn	1
				Grasses	1
Dimethyldioxane	-	500	200	Corn	4
Diethylamine $(C_2H_5)_2NH$	10	300	70	Corn	1
				Grasses	1
Caprolactam	100	500	200	Corn	16
Crotonaldehyde $CH_3CH{=}CHCHO$	250	100	100	Corn	2
				Grasses	2
				Potatoes (tubers)	5
p-Xylene $C_6H_4(CH_3)_2$	1.0	100	100	Corn	5
				Grasses	4
				Carrots (root crop)	7
Methanol CH_3OH	200	100	70	Corn	2
				Celery (root)	7
				Celery (plant)	3

1	2	3	4	5	6
n-Propanol $CH_3CH_2CH_2OH$	62	100	70	Corn	4
				Grasses	3
				Celery (root)	10
				Celery (plant)	9
Toluene $C_6H_5CH_3$	200	100	70	Corn	5
				Grasses	3
				Carrots (root)	6
Furfural CHO	1.0	400	50	Corn	2
				Grasses	2
Cyclohexanol $C_6H_{11}OH$	1.0	300	50	Corn	4
				Grasses	3
				Leguminous grasses	3
Cyclohexanone $C_6H_{11}O$	50	300	50	Corn	5
				Grasses	2
				Leguminous grasses	4
Dichlorethane $C_2H_4CI_2$	200	100	50	Grasses	13
				Celery (plant)	8
				Celery (root)	12
Carbon Tetra-chloride CCI_4	50	100	50	Grasses	13
				Celery (plant)	16
				Celery (root)	16
Ethanol C_2H_5OH	-	100	70	Corn	5
				Grasses	4
				Celery (plant)	5
				Celery (root)	10

*Remark: MPC b.p.s. stands for maximum admissible concentration of matter in artificial biological purification systems.

USA-USSR WORKING AGREEMENT ON UTILIZATION OF WASTEWATER ON LAND

EFFECTIVENESS OF APPLICATION OF ORGANIC AND MINERAL FERTILIZERS DURING WASTE WATER IRRIGATION

EFFEKTIVNOST'VNESENIYA ORGANICHESKIKH I MINERAL'NYKH UDOBRENIY PRI OROSHENII STOCHNYMI VODAMI

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The waste waters used for irrigation may contain different quantities of nutritive substances. I.P. Kanardov [1] writes that the NPK ratio in waste water is usually 5:1:2, whereas the optimal ratio of such nutritive substances for plants averages 2.5:1:2.3 according to D.N. Pryanishnikov. Consequently, soil irrigated with waste water becomes enriched, first of all, with nitrogen. It would be expedient to compensate for phosphorus and potassium shortages through the additional application of fertilizers. The nutritive substances which enter the soil with waste water are used most fully in the course of the seasonal irrigation of soils possessing good absorption capacity. Soils with a low absorption capacity such as sandy and large grained soils with small quantities of organic substances have a low coefficient of utilization of nutritive subtances. On such soils extensive sprinkling and irrigation norms are frequently paralleled by losses of nutritive substances. Peat, compost, manure and others applied to the soil raise the utilization coefficient of nutritive substances entering the soil with the waste water. A great variety of conflicting data exist on the utilization coefficient of nutritive substances in irrigated fields. For example, at communal irrigated fields with an annual utilization of waste water per unit area ranging from 10,000 to 25,000 cubic meters per hectare, and a considerable denitrification and extensive seepage of irrigation water in the ground water, the plants' utilization coefficient is 0.5 for nitrogen, 0.95 for phosphorus, and 0.7 for potassium [2].

Losses of nutritive substances occur also in irrigated fields using higher sprinkling and irrigation norms or where waste water is unevenly distributed. Where such water accumulates, a considerable percentage of the nutritive substances, nitrogen above all, seeps into the ground waters. Such losses of nutritive substances could be reduced through the proper organization of irrigation and water distribution. In terms of nutritive content, waste water is fertilizing. However, the additional application of organic and chemical fertilizers upgrades the effectiveness of its utilization. This problem was studied at the Noginskiy Sovkhoz, Moscow Oblast, and at the experimental farm of the VNIISSV.

The residential waste water of the neighboring settlement was used for irrigation. One liter of waste water contained the following: from 59 to 156 milligrams general nitrogen; 13.6-47 milligrams P_2O_5; 16.2-38.5 milligrams K_2O; 83.5-155.0 milligrams Ca+Mg; the pH was 6.5-7.1. The amount of water per sector was determined through control measurements of debits and length of water supply.

The study of waste water indicated a relative shortage of nutritive substances. The need arose, therefore, for field experiments to determine the effectiveness of organic and chemical fertilizers in a waste water irrigation system.

The soils of the experimental sector of Noginskiy Sovkhoz -- consisting of light soddy-underpodzolic sand -- have a

Table 1. Agrochemical Characteristics of the Soil of the Experimental Section (Noginskiy Sovkhoz)

Depth of soil sample, cm	pH of salt extract	Humus according to Tyurin, %	Total absorbed foundation, mg equiv to 100 g soil	Hydrolytic acidity, mg equiv to 100 g soil	Mobile forms, mg per 100 g of soil	
					K_2O	P_2O_5
0-20	5.7	2.53	20.0	2.7	9.2	7.0
30-40	6.1	0.56	6.4	-	7.7	7.0
40-50	4.9	0.63	5.6	-	7.0	5.0
50-75	4.2	0.17	-	-	-	5.0

bulk density of 1.5-1.72, and a density of 2.5-2.67 grams per cubic centimeter. The filtration coefficient of these soils is 10 meters per 24 hours. The sector is well drained and has a discharge of ground waters located at a depth of 9.33 meters (square No. 269). The chemical analysis of the soils of the irrigated section is given in Table 1.

Despite the low effective and potential fertility, sandy soils have favorable thermal characteristics and good aeration, and yield to cultivation quite rapidly. The need for additional application of organic and chemical fertilizers in year-round irrigation with waste water was studied in accordance with the following system:

Control without irrigation and fertilizing;

Control -- irrigation with industrial and residential waste water;

Manure 40 tons per hectare + irrigation with industrial-residential waste water;

Peat 40 tons per hectare + irrigation with industrial-residential waste water;

$N_{80}P_{100}K_{100}$ + irrigation with industrial and residential waste water;

$N_{80}P_{100}$ + irrigation with residential waste water;

N_{80} + irrigation with industrial and residential waste water;

P_{100} + irrigation with industrial-residential waste water;

K_{100} + irrigation with industrial and residential waste water.

The irrigation was made with a 60% maximum soil moisture capacity. The crop was the Krasnodarskiy 1/49 corn strain. Chemical fertilizers were applied during the spring plowing. The amount of fertilizer to be applied per lot was computed by the formula:

$$y = \frac{d.S\cdot 100}{P}$$

in which d is the dose of fertilizer in active matter per hectare, in kg; S is the lot size in square meters; and P is the content of active substance in the fertilizer, in percent.

The experiment was repeated three times on 300-312 square meter lots. The crop was estimated on the basis of the continuous method.

At the same time organic fertilizer (manure and peat) was applied with manure spreaders mounted on a Belarus' tractor.

The autumn plowing of the section was 22-23 centimeters deep; the early autumn plowed field was harrowed in early spring, followed by a replowing at a 20-22 centimeter depth with the application of fertilizers based on the experimental system with presowing leveling of the soil with a cultivator. The corn was planted with a SKPN-8 drill with a density of 30 kg per hectare. In the vegetation period irrigation and a single application of herbicides was followed by inter-row cultivation. The resulting data indicated that the additionally applied organic and chemical fertilizers were highly effective with waste water irrigation (Table 2).

Similar studies were made at the experimental sector of the VNIISSV with the irrigation of potatoes with the waste water of a fine woolen cloth factory. The following alternatives were studied:

Control without irrigation;

Control with water supply irrigation averaging 3,000 cubic meters per hectare;

Control with vegetational irrigation of 600 cubic meters per hectare;

Application of peat, 50 tons per hectare, with water supply irrigation of 3,000 cubic meters per hectare;

Application of 50 tons per hectare of peat + $N_{90}P_{40}K_{90}$ + water supply irrigation of 3,000 cubic meters per hectare.

Table 2. Influence of Organic and Chemical Fertilizers in Waste Water Corn Crop Irrigation
(2-year average)

Variants	Yield, quintals per hectare	Additional yield and control with waste water irrigation with 60% maximum soil moisture: quintals per hectare	%
Control without irrigation and fertilizing	154.1	-	-
Control, irrigation with 60% of maximum soil moisture	274.6	-	-
Manure 40 tons + irrigation with 60% of maximum soil moisture	432.1	157.5	57
Peat 40 tons + irrigation with 60% maximum soil moisture	357.8	101.2	36
$N_{80}P_{100}K_{100}$ + irrigation at 60% of MSM	345.4	70.8	25
$N_{80}P_{100}$ + irrigation with 60% MSM	295.5	21.0	7
N_{80} + irrigation with 60% MSM	282.9	8.3	3
K_{100} + irrigation with 60% MSM	208.1	50.0	13.6
Experiment accuracy, %		2.67	
Least essential disparity ($LED_{0.95}$)		19.836	

The experimental lots averaged 250 square meters and the experiment was repeated three times.

The studies indicated that the additionally applied fertilizers increased potato yields. Thus, the yield averaged 143.6 quintals per hectare with irrigation with waste water and application of 50 tons per hectare of peat + $N_{90}P_{40}K_{90}$. At the control plot irrigated with the same amount of waste water (76.6 quintals per hectare), with the application of total chemical fertilizer of $N_{90}P_{40}K_{90}$ the potato crop averaged 137.7 quintals per hectare, or was 61.6 quintals above the control figure.

The additionally applied organic and chemical fertilizers, with waste water irrigation, would be economically justified: production cost per quintal of grain mass of corn without irrigation equalled 1.22 kopek; with irrigation without fertilizer, 0.83; and with additional application of 40 tons per hectare of manure, 0.86 kopek; of peat, 0.87 kopek; and $N_{80}P_{100}K_{100}$ -- 0.70 kopek.

The high effectiveness of the additional application of organic and chemical fertilizers in irrigation with waste water is based not only on the content of their nutritive substances but their favorable impact on the soil structure. Furthermore, the application of manure and peat improves the aeration system of the soil which is very important for the decomposition of the organic matter which penetrates with the waste water.

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NUTRITION VALUE AND FODDER HARMLESSNESS OF PLANT OUTPUT GROWN WITH TEXTILE INDUSTRY WASTE WATER IRRIGATION

PITATEL'NAYA TSENNOST' I KORMOVAYA BEZVREDNOST' RASTENIYEVOLCHESKOY PRODUKTSII, VYRASHCHENNOY PRI OROSHENII STOCHNYMI VODAMI TEKSTIL'NOGO PROIZVODSTVA

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The intensive development of Soviet industry, the chemization of all economic sectors, urbanization, and the construction of big livestock complexes on an industrial basis have led to the formation of huge quantities of waste water which are a serious source of pollution of open reservoirs. Irrigation farming fields (IFF) play an important role among the existing methods for the soil purification of waste water. The use of waste water for farm crop irrigation enables us to resolve simultaneously a number of most important national economic problems related to the purification and additional purification of waste water, and the protection of water sources from pollution. It offers extensive opportunities for raising high and stable harvests. Particularly great possibilities develop with the waste water irrigation of fodder and pasture farmland. The long experience gained by Moscow Oblast sovkhozes (Noginskiy, Znamya Oktyabrya, Zarya Kommunizma, and Im. Timiryazev) proves that with waste water irrigation pasture land productivity increases 3-4 times, which makes possible the production of 7,000 to 10,000 fodder units per hectare.

It is believed that feed production on IFF will play a considerable role in the future overall animal husbandry feed balance.

The toxicological-sanitation studies aimed at preventing any possible harmful influence of IFF output on the organisms of warm-blooded animals become particularly topical in this connection, since a variety of harmful and toxic substances of no nutritional or food value may find their way from waste water to farm crops (detergents, chromium, organic dyes, and others).

Studies were made by the VNIISSV (V.P. Sayapin, L.N. Tsvetnova, I.A. Kas'yanov, T.L. Kosareva, N.I. Matulyavichene, and Yu. I. Tararin) on the influence of long-term feeding with perennial grasses and hay grown in sectors irrigated with waste water from the Kupavna fine woolen fabric factory, and of the basic physiological and biochemical indicators of the livestock, their productivity, reproductive functions, and nutritive value of animal husbandry output (meat and milk). The experiments were conducted at the laboratory for veterinary hygiene research and the Noginskiy Sovkhoz, Moscow Oblast. A complex set of biological indicators was studied in the course of chronic experimentation with rabbits, calves and cows, making possible the objective determination of the physiological condition of the organism.

Preliminary organoleptic studies of feeds indicated that perennial grasses and hay grown on IFF were not distinct from control feeds in terms of color and odor. In terms of their chemical composition, the tested feeds were characterized by a considerably higher content of raw protein and fat, and a lower amount of cellulose. This proved their relatively good nutritive value. The degree of eatability of the studied hay,

Table 1. Some Morphological and Biochemical Indicators in Experimental Rabbits.

Animal Group	Studied Indicators						
	Erthrocytes (million /mm^3)	Leukocytes (thousand /mm^3)	Hemoglobin (Sal units)	General Portein (grams %)	ALT (units /mg)	Bilirubin (mg %)	General Glutathione (mg %)
Control	6.8±0.6	7.4±0.6	71.7±1.6	7.0±0.2	47.8±2.5	0.12±0.01	41.75±1.81
Experimental	6.7±0.3	7.9±0.8	75.8±2.6	7.2±0.1	41.8±0.8	0.1±0.01	41.05±1.07

Table 2. Cow Milk Chemical Composition

Animal Group	Studied Indicators (grams %)				
	Acidity (°T)	Density	General Nitrogen	Fat	Sugar
Control	18.4±0.2	1.03±0.001	3.2±0.1	3.0±0.2	4.3±0.2
Experimental	16.6±0.8	1.03±0.001	3.9±0.03	3.5±0.2	4.7±0.1

compared with the control, averaged 94.3% ($P>0.05$). Adding to the ration of rabbits' hay from IFF resulted in the following digestibility coefficient of the nutritive substances in the feed mix: protein, 75.9%, compared with 68.1% in the control; fat, respectively, 95.3 and 94.9%; and cellulose, 61.9 and 56.7%.

Throughout the experimental period (2.5 years) the general condition of the experimental animals remained within the range of the physiological norm. All animals adequately reacted to their environment, retained their appetite, and added weight. In the course of the experiment no substantial deviations were noted in the experimental animals in the temperature reaction of the body, and frequently of the pulse beat and respiration.

No changes were noted in the quantitative content of erythrocytes and leukocytes or in the percentile ratio among the individual types of leukocytes and the erythrocyte sedimentation rate between the experimental and control animals.

The experimental data show that adding IFF grown fodder to animal rations has no adverse effect on the protein, carbohydrate, mineral and pigment metabolisms, the level of oxidation-reduction processes in the body, or the activeness of the most important blood ferment systems (Table 1).

The use of experimental rations had no harmful effect on the genetic apparatus of the animals and the immunological reactiveness (complementary, bacteriocide, lysozymic activeness of the blood serum and the manufacturing of specific agglutinins). Morphological studies showed no changes in the macro- and micro-morphological structure of the internal organs of the experimental animals.

The study of cow milk productivity indicated that the average milking per forage cow over 300 days of lactation in the experimental group was 3,230 kilograms of milk, compared with only 3056 kilograms in the control group. The organoleptic study of the milk of the experimental cows was totally identical to the milk of the control cows in terms of color, taste, and odor. Throughout the experimental period (2.5 years) no negative changes were noted in the chemical composition of the milk of the experimental cows (Table 2).

In terms of chemical composition the meat of the experimental rabbits as well showed no major differences compared with the control group (Table 3).

Table 3.

Animal Group	Studied Indicators (grams % of dry matter)						
	Protein Nitrogen	Fat	Ash	Na	K	P	Ca
Control	11.71±0.20	5.78±0.38	4.17±0.07	0.14±0.01	1.20± 0.10	0.65± 0.07	0.47±0.05
Experimental	11.89±0.30	6.69±0.96	4.34±0.21	0.19±0.05	1.14± 0.04	0.73± 0.09	0.50±0.04

Therefore, the conducted studies prove that the extensive use of perennial grasses and hay grown on sectors irrigated with the waste water of the Kupavna fine woolen fabrics factory in feeding a variety of animals had no adverse effect on the health conditions of the animals, their productivity, nutritive value of animal husbandry output, animal reproductive function, and quality of offspring.

USA-USSR WORKING AGREEMENT ON UTILIZATION OF WASTEWATER ON LAND

IRRIGATION USE OF FOOD INDUSTRY WASTE WATER

ISPOL' ZOVANIYE NA OROSHENIYE STOCHNYKH VOD PREDPRIYATIY PISHCHEVOY PROMYSHLENNOSTI

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Our country has a number of food industry enterprises. The principal among them are sugar, starch, and hydrolysis plants. Most of them release large amounts of waste water in filtration fields or surface waters (rivers, ponds and others). In the first case a large area of fertile soil is removed from farm use; in the second, surface water sources become polluted.

Studies conducted by the All-Union Scientific Research Institute for the Agricultural Utilization of Waste Water [VNIISSV] have established that the waste water of starch plants and sugar refineries could be used for irrigation. Such waste water is of considerable and, frequently, high fertilizing value. After treatment such water is suitable for the irrigation of meadows, pastures and fodder, grain, and industrial crops.

The waste water of starch plants is characterized by a low acid reaction (pH=5.3-5.6), a relatively low concentration of dissolved substances (2.0 grams per liter), and bicarbonate-sulfate composition. Frequently salts of univalent cations (K, Na) predominate. Inherent in such water is a high content of organic matter. This is indicated by the level of BOD which ranges from 3 to 5 grams per liter of O_2 depending on production technology and the nature of the preliminary waste water treatment. The waste water has a high fertilizing value. It has a content of 30-150 milligrams per liter nitrogen, 85-180 milligrams per liter potassium, and 20-40 milligrams per liter phosphorus. Studies have shown that, compared with waste water from the production of corn starch, waste water from the production of potato starch is better in terms of structure and fertilizing value. With an irrigation standard of 3,000 cubic meters per hectare, considered average for a number of farm crops, the soil receives 7 to 12 quintals per hectare of nitrogen, 6 to 13 quintals per hectare of potassium, and 3 to 5 quintals per hectare phosphorus fertilizers converted in terms of chemical fertilizer. The high content of nutritive elements in waste water released by starch-making plants, used for irrigation, raises the fertility of the soils and farm crop yields. Perennial grasses and corn are particularly responsive to such irrigation. In Ryazanskaya Oblast, corn yields on gray forest soils irrigated with waste water of the Zadubrovskiy Dry Starch Plant were 2-3 times higher compared with nonirrigated areas and averaged 420 quintals per hectare of green mass.

Currently an irrigated area covering 600 hectares has been established at that site, 200 of which are in use. Most of the waste water is used to irrigate fodder and grain crops. As a result of this untreated waste water is no longer released into the Yaroslavna River which flows into the Oka River. The establishment of an irrigated field at the site has blocked the pollution of the basin of the Oka River. Before it can be used for irrigation, the waste water of starch plants must undergo treatment consisting of homogenizing and settling. Starch plant waters may be used for the irrigation of farm crops in all soil and climatic zones: Chernozem, gray forest, and soddy-podzolic soils of various mechanical compositions, or similar soils.

The waste water of sugar refineries has a lesser fertilizing value and higher suspended sediments.

Currently the waste water of sugar refineries is treated through filtration systems. Such systems consist of sedimentation ponds and filtration fields which remove from cultivation huge areas of fertile Chernozem soil within whose area the bulk of sugar refineries is located.

The studies made by our institute have shown that the waste water of sugar refineries is suitable for the irrigation of farm crops and Chernozem soils.

The waste water of sugar refineries is characterized by an alkaline reaction (pH=7.5-8.5), a bicarbonate content, high content of organic matter, and average fertilizing value. On an average it contains 45 milligrams per liter nitrogen, 75 milligrams per liter potassium, and very little phosphorus (2.5 milligrams per liter).

With an irrigation norm of 3,000 cubic meters per hectare the soil receives 4 quintals per hectare of nitrogen, 5.6 quintals per hectare of potassium, and 0.5 quintal per hectare of phosphorus fertilizers.

The waste water of sugar refineries has a high content of suspended sediment, averaging 50 grams per liter. The sediment contains lime and soil particles. Its high content hinders the utilization of waste water for irrigation. This calls for advanced treatment which includes averaging and sedimentation. The sedimentation process takes place in the sedimentation ponds which are always found on filtration fields. The clarified waste water has a better chemical composition. However, its fertilizing value is 15 to 20% below that of unclarified waste water.

The results of the studies made by the VNIISSV at sugar refineries of Krasnodarskiy Kray showed that irrigation with clarified waste water has a positive influence on the fertility of Chernozem soils and on farm crop yields. The content of mobile forms of nitrogen, phosphorus and potassium rises in the soils. Such irrigation does not cause soil salinization or solonetzization processes. Extensive studies conducted at the Timashevsk Sugar Refinery in Krasnodarskiy Kray proved the high effectiveness of the utilization of waste water in the irrigation of corn, sudan grass, sugar beet, wheat and stubble crops. With irrigation, crop yields rose from 50 to 150% compared with unirrigated areas. In 1970 yields of controlled sudan grass averaged 115 quintals per hectare; irrigated with clarified waste water at the rate of 2000 cubic meters per hectare, yields reached 165 quintals per hectare; irrigated with unclarified waters in the same, yields reached 215 quintals per hectare. Stubble crop yields (peas with oats) increased with higher irrigation norms. Without irrigation they averaged 58 quintals per hectare; they rose to 94 quintals per hectare with an irrigation norm of 1000 cubic meters per hectare, and 145 quintals per hectare with 2,000 cubic meters per hectare irrigation.

Practical experience has been developed in the utilization of the waste water of sugar refineries by the Kollektivist Plant in Kurskaya Oblast, and the Nizhniy Kislyay Sugar Refinery in Voronezhskaya Oblast. The results of the studies were tested under industrial conditions. At these projects sovkhoz land was used for irrigation with waste water. The irrigated areas ranged from 20 to 150 hectares.

In 1975 irrigation with clarified waste water of the Timashevsk Sugar Refinery was used by the Timashevskiy Sovkhoz over an area of 500 hectares. The waste water from the collector was fed to the existing sovkhoz irrigation network. In the course of 1975 corn and alfalfa were irrigated with waste water.

Waste water was simultaneously treated at the irrigation fields. Water was promptly received from filtration fields which enabled the plant to operate normally and rhythmically in the new production season. The use of waste water for irrigation made it possible to resume crop cultivation on some of the filtration area.

Calculations showed that the treatment and utilization of waste water from the Timashevsk Sugar Refinery yielded tremendous results. Compared with total biological treatment, now planned for many sugar refineries, the annual savings from the utilization of waste water for irrigation equalled 833,000 rubles.

Scientific research and practical experience showed that the use of waste water from the sugar refinery for irrigation is a promising measure which must be extensively applied in agricultural production practice.

USA-USSR WORKING AGREEMENT ON UTILIZATION OF WASTEWATER ON LAND

METHODS FOR PREVENTING THE CONTAMINATION OF WATER BODIES BY WASTE WATER OF ANIMAL HUSBANDRY COMPLEXES

SPOSOBY PREDOTVRASHENIYA ZAGRYAZNENIYA VODOYEMOV STOCHNYMI VODAMI ZHIVOTNOVODCHESKIKH KOMPLEKSOV

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The building of big state industrial type animal husbandry complexes demands the urgent solution of problems of removal, processing, and rational utilization of liquid manure whose overall production is about 200 million tons per year.

As a result of the high concentration of cattle in livestock farms, the elimination of cattle bedding, and the total mechanization of the removal of the manure from the premises a new type of waste water has developed -- the outflow of animal husbandry complexes consisting of a mixture of droppings, and feed and water waste, used for industrial feeds. In connection with the reconstruction and building of new specialized animal raising complexes by 1980 the volume of animal husbandry outflow will more than double.

The volume of waste water depends on the cattle raising method, size of the herd, animal species and age, duration of stable period, manure removal system, type of forage, and other factors (Table 1).

In terms of concentration of organic matter and mineral salts, the waste water of livestock complexes exceeds by several hundred percent household and industrial waste water, for which reason livestock farms are far greater polluters of the environment than industrial projects (Table 2).

The amount of nutritive elements contained in the overall amount of livestock outflow is the equivalent of 2.2 million tons of nitrogen, 1 million tons of phosphorus, and 2 million tons of potassium. This is 4 times the amount of fertilizers contained in the waste water of the food industry and industrial-residential volumes which total 11.8 million cubic meters per year and is usable for irrigation.

Penetrating water bodies, animal husbandry waste water triggers nitrification. It drastically worsens the quality of water, making it unsuitable for use.

Specialists in different countries have developed over 20 methods for the neutralization of liquid manure -- chemical, thermal, biological and others. As a rule, all of them are expensive and do not insure adequate purification. Thus, in the GDR formaldehyde processing based on 3 kg per cubic meter of liquid manure is recommended for its chemical disinfection. However, bearing in mind that the discharge of liquid manure by big animal husbandry complexes exceeds 2000 cubic meters per day, such neutralization would require a great volume of disinfecting substances, for which reason the chemical disinfection method is considered unsuitable.

Collectors (frequently earthen) may be of considerable size and capable of containing up to 100,000-500,000 cubic meters. As a result of its stay in the collector the animal husbandry waste water is cleaned from the harmful microflora. However, the concentration of organic matter remains quite high as a result of which such outflow cannot be channeled into water bodies.

Table 1. Volume of Livestock Farm Outflow Based on Collection System.

Type of complex	No. of head raised together	Animal excrement output (10^3 m^3/yr)	Waste water outflow from the complex (10^3 m^3/yr) Gravity flow	Water flushing
Hogs				
for 108,000 head	73000	239.0	321.0	940.0
for 54,000 head	37000	114.0	181.0	332.5
for 24,000 head	24000	70.5	96.8	195.5
for 12,000 head	12000	36.0	52.4	101.0
Young cattle offspring				
for 10,000 head	9883	94.8	113.0	-
for 20,000 head	20000	328.0	-	-
for 30,000 head	30000	493.0	-	-
for 600 cows	600	12.0	14.2	20.8
Dairy cattle				
for 800 cows	800	16.0	18.9	30.6
for 1200 cows	1200	24.0	28.5	46.0

Table 2. Chemical Composition of Waste Water of Livestock Complexes, Industrial-Residential, and Food Industry Waste Water, mg/liter.

Waste water type	Dry matter content	General nitrogen	P_2O_5	K_2O
Livestock cattle, 90% moisture	12000	4000	2000	4500
Hogs	10500	5000	2500	2400
Industrial-residential	550	35	6	20
Food industry	1500	80-100	10	240

The two-stage biological purification of livestock waste water used in the sovkhoz imeni 50-letiya SSSR (Moscow Oblast) lowers the concentration of organic substances for the basic BPK_5 [biochemical oxygen requirement] to 170 mg per liter. However, even in the course of the totally satisfactory reduction of the BPK_5, biogenic substances such as nitrogen, phosphorus, and potassium find their way in the waste water channeled into water bodies. The nitrogen nitrification and phosphorus breakdown processes are accompanied by a high water bloom. For this reason the livestock waste water must be channeled into irrigation fields even after biological purification.

The experience of irrigation with liquid manure used by the farm of the Belgorodskaya Oblast may be cited as an example. Until 1970 29 specialized farms raising and feeding hogs and cattle stored waste water in earthen containers over a period of years. Periodically, with an overflow, particularly in spring and autumn, the water was channeled into reservoirs (Druzhba and imeni Frunze Sovkhozes, Belgorodskiy Rayon, imeni Zhdanov, Rakityanskiy Rayon, and others). Furthermore, considerable farming areas had to be set aside as liquid manure collectors. In some farms this totaled 40 to 50 hectares. The industrial type farms make full use of liquid manure on irrigation fields as crop fertilizer. This leads to stable yields and considerably strengthens the animal husbandry fodder base.

In the sovkhoz imeni Zhdanov in Rakityanskiy Rayon, since 1971 a 210-hectare area has been regularly irrigated with the waste water of a livestock complex raising 24,000 hogs. Outflow clarified in advance in manure sedimentation tanks of the Rosgrprosel'khovstroy system and clean pond water are used for irrigation. The release of waste water from the collectors was stopped.

A number of kolkhozes have developed irrigation fields using animal husbandry waste water after purification in manure collector sedimentation tanks. At the end of the Ninth Five-Year Plan the overall area of such irrigated fields in Belgoro-skaya Oblast will reach 9,400 hectares.

Initially the hog breeding complex of the sovkhoz imeni 50-letiya SSSR in Kalininskaya Oblast built filtration fields covering 9.7 hectares with five sedimentation tanks. The liquid manure clarified in the sedimentation tanks was released in the Ol'khovka stream and, subsequently, in the Mezhurka and Volga Rivers. As a result the bacterial pollution of Volga water at the Mezhurka confluent rose 50-70 times. The completion of a 200-hectare drained irrigation field made it possible to reduce the BPK_5 to 3 mg per liter. Such waste water could be discarded in water channels.

The studies conducted by the VNIISSV of the process used for the purification of animal husbandry outflow on irrigation fields of sovkhozes in the Belgorodskaya and Moscow Oblasts indicated that the upper 30-cm-thick soil stratum absorbs almost entirely the nitrogen, phosphorus, and potassium contained in the waste water, reduces the amount of suspended matter 90-99%, and lowers by 99% the BOD in lysimetric waters.

Tables 3 and 4 show the degree of purification of cattle manure waste water of the Voronovo complex and of the biologically purified outflow of the hog breeding complex of the sovkhoz imeni 50-letiya SSSR in Moscow Oblast.

It would be expedient to plant feed crops on sectors irrigated with animal husbandry waste water with a view to the development of the solid fodder animal husbandry base. Crops such as perennial and seasonal grasses can increase yields greatly and, with the production of grass vitamin-enriched yield, provide hygienically impeccable products.

One of the determining prerequisites in selecting the place for the building of a livestock complex should be the availability of land suitable for the totally effective utilization of liquid manure on irrigated crop land. In order to protect ground water from pollution, the level of the highest subsoil water should be at a depth of no less than 2 meters.

Roughly, the size of the irrigated farm land could be based on the ratio of 1500 kg of live animal weight per hectare. This value is 6 head of cattle or 15 hogs per hectare in the United States, no more than 6 head of cattle or 30 hogs per hectare in the GDR, and no more than 4 conventional (500 kg) head in the FRG.

The studies conducted by the institute in Belgorodskaya, Kalininskaya and Moscow Oblasts indicated that irrigation with animal husbandry waste water insures high and stable fodder crop yields. Thus, at the specialized Druzhba farm in Belgorodskaya Oblast, additional sugar beet crops as a result of irrigation with clarified hog farm waste water ranged from 300 to 400 quintals per hectare, and of silage corn, 60 to 160 quintals (Table 5). In Moscow Oblast irrigation with cattle waste water of perennial grasses and a mixture of peas and oats on heavy loamy soil increased yields by 20 quintals per hectare in terms of fodder units (Table 6).

Irrigation with animal husbandry waste water is done most frequently with the help of long spraying DDN-45 and DDN-70 sprinklers which make possible the application of liquids containing a substantial amount of suspended matter.

In order to increase the winter application norms and avoid the collection of outflow the soil must be prepared in the autumn before the irrigation: it must be plowed at a depth of 25-27 cm with a 25-cm ridging, slotting and moling. Furthermore, the edges of the most slanted irrigated sectors should be surrounded by small 30-40 cm high ridges.

Of late the Belgorodskaya Oblast sovkhozes have used for irrigation animal husbandry outflow clarified in precipitation tanks using quite adequate medium jet Volzhanka sprinklers. Such systems insure high irrigation quality. The odor does not spread over wide areas as when long range sprinklers are used.

Table 3. Level of Purification of Waste Water at the Voronovo Animal Husbandry Complex, Soddy-podzolic Soil (Irrigation norm 750 m^3/hectare), mg/liter

Indicator	Color	pH	Suspended precipitate	Residue dry	Residue hardened	HCO_3	SO_4	Cl	Ca	Mg	K	Na	Nitrogen General	Nitrogen Ammoniacal	P_2O_5	BOD
Initial waste water	Green-gray	7.7	7482	5500	2700	6710	200	532	312	240	1230	300	1143	941	100	22400
Lysimetric waters from the 0-30 cm stratum	Light-yellow	7.6	885	1245	830	1220	80	130.8	210	162	18.5	95	71	42	2.5	320
Lysimetric waters from the 0-50 cm stratum	Ibid	7.5	115	640	450	152.5	90	52.3	40	24	2.5	45	10	7	1.5	200
Lysimetric waters from the 0-100 cm stratum	Ibid	7.4	100	605	455	141	75	48	38	22	3.0	50	10	5	1.0	180
% of purification			98.7	89	83.2	97.9	62.5	91	81.8	90.9	99.8	83.3	99.1	99.5	99	99.2

Table 4. Degree of Purification of Biologically Purified Waste Water of the Animal Husbandry Complex imeni 50-letiya SSSR, Soddy-Podzolic Loamy Soil, mg/liter

Indicator	pH	Suspended precipitate	Residue dry	Residue hardened	HCO_3	SO_4	Cl	Ca	Mg	K	Na	Nitrogen General	Nitrogen Ammoniacal	P_2O_5	BOD
Initial condition of bulk waste water	7.6	477	1700	728	1205	84.3	106.4	96	67.2	205	220	142	112.6	82	800
Lysimetric water, 0-20 cm stratum	7.5	40	1472	580	322	52.2	105	160	55.2	10	40	12	0.7	-	320
Lysimetric water, 0-75 cm stratum	7.5	38	562	428	220	41.14	82.3	140	50.4	10	34	1	0.3	-"-	300
Percentage of purification		92	67	41.2	81.1	51.5	22.4		25	95.1	85.6	99.4	99.7	-	62.5

Table 5. Effect of Irrigation with Waste Water from the Hog Raising Complex on Crop Yields (Druzhba Kolkhoz, Belgorodskaya Oblast)

Crop	Norm of irrigation with clarified outflow, m^3/ha	Yield, quintals/ha	Additional yield, quintals/ha
Sugar beets	1900	830	455
Silage corn	1400	470	159
Barley (green mass)	600+100 kg/ha P_2O_5	239	75
Barley for grain	200+400 clean water	36.8	6.4

Table 6. Influence of the Irrigation with Cattle Animal Husbandry Outflow on Crop Yields under Moscow Oblast Conditions (Voronovo Sovkhoz)

Variant	Green mass crop, quintals/ha		In terms of fodder units	Increase, fodder units
	Perennial grasses	Pea-oat mix		
Control -- no irrigation	139	222	6330	-
Waste water winter irrigation -- 300 m^3/ha	137	272	7090	760
Waste water winter irrigation -- 150 m^3/ha	202	252	8070	1740
Spring waste water irrigation (18 May) -- 300 m^3/ha	187	300	8540	2210

The most sanitary irrigation methods with animal husbandry waste water are those of gravity flow and intrasoil systems which are particularly effective in areas where no deep freezes occur in the nonvegetation period. The VNIIGIM and VNIISSV developed a machine for the intrasoil introduction of liquid manure. One such machine could haul and apply to the soil the manure produced by up to 5,000 hogs or 500 cows.

The economic study of agricultural production on irrigated farm land has shown that with an increased volume of additional outlays for irrigation the cost of output at irrigated sectors is considerably lower than on non-irrigated land (Table 7).

Table 7 shows that at the irrigated section of the kolkhoz imeni Zhdanov (a complex for 24,000 hogs) production cost per quintal of fodder units ranged from 0 to 55 rubles for irrigated corn and seasonal grasses and 2-74 rubles for beets compared with unirrigated land.

Table 7.

Crop	Irrigated area, ha	Yield, quintals/ha		Cost per unit of output in rubles & kopeks		Cost per quintal fodder units in rubles & kopeks	
		Irrigated	Nonirrigated	irrigated	nonirrigated	irrigated	nonirrigated
Corn or fresh fodder	52	264	191	0-38	0-49	1-90	2-45
Fresh seasonal grasses	86	174	112	0-36	0-50	2-25	3-00
Sugar beets	50	355	207	0-82	2-15	15-17	17-91

USA-USSR WORKING AGREEMENT ON UTILIZATION OF WASTEWATER ON LAND

ON A METHOD FOR DETERMINING THE USEFULNESS OF WASTE WATER IRRIGATION

ISPOL'ZOVANIYE BYTOVYKH I PROM'SHLENNYKH STOCHNYKH VOD V SEL'SKOM KHOZYAYSTVE

L. Ye. Kutepov

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STUDY OF THE CHEMICAL COMPOSITION OF WASTE WATER

The utilization of waste waters in irrigated fields requires a knowledge of the chemical composition. Its study requires the separate consideration of its mineral structure and pollution with organic matter.

In most types of waste water the basic mineral salts include sulfates, chlorides, and the bicarbonates of sodium, potassium, calcium, and magnesium. These salts are basic in soils and in natural waters. Data are available on soil-hydrogeological conditions and the irrigation system, and the admissibility of the concentration of such salts in irrigation waters.

Irrigation water must have a reaction with the environment close to neutral (pH 6-8). In the irrigation of acid soils it would be desirable to change the pH value toward the alkaline side (pH 7-9); in the case of alkaline soils the acidity should be higher (pH 5-7). Obtaining the necessary pH value for waste water prior to its utilization for irrigation presents no technical difficulties. The quantity of nutritive elements needed by the crops, applied with the irrigated water, should not substantially exceed specific plant requirements.

A number of elements (heavy metals, arsenic, selenium) may have a toxic effect on the soil microflora or the plants, or accumulate in the plants and, in turn, thus adversely affect animals and humans. Such waters are either unsuitable for irrigation or could be used after preliminary treatment.

Usually, the salt composition of industrial waters is limited to sulfates, chlorides, and bicarbonates of calcium, magnesium, sodium and potassium. The existence of bicarbonates and carbonates should be determined only in the case of alkaline waters. Thus, irrigation water should have a pH value close to neutral and carbonates and bicarbonates should be almost nonexistent.

The simplest analytic system which would meet the reclamation requirements would be to determine the overall mineralization and cation structure in the water samples (sodium, potassium, calcium, magnesium). General mineralization must equate the sum total of equivalent concentration of cations and could be determined through conductimetry.

The presence of organic matter in water is determined, above all, on the basis of overall BOD. Should BOD exceed 200 mg per liter of O_2 we must determine the organic substances which pollute the waste water. According to the type of production technology the basic organic pollutants are identified and the substances whose concentration, based on production technology, could exceed average daily samples by 20-30 milligrams per liter must be analytically identified.

A considerable biological mass of microorganisms is found in the effluent following biological treatment and, in summer, in storage ponds. Such microorganisms are oxidized bichromatically and substantially exceed the determined quantity of organic pollutants. The biomasses in the water could be considered only as fertilizers. Therefore, prior to the analysis of such water samples, the biomass must be separated through centrifugal separation, or filtration using a membrane or Selzer filter.

IRRIGATION REQUIREMENTS CONCERNING THE MINERAL COMPOSITION OF THE WATER

The optimal concentration of salts in soil solutions is within the range of 45 to 75 milligrams-equivalent/liters (3-5 grams per liter). With this concentration the amount of salts released with the crop is considerably lower than the gross salt content in the soil water used for transpiration. Therefore, in irrigation practice water containing no more than 15 mg-equivalent per liter salts is considered suitable while water containing salts in excess of 45-105 mg-equivalent per liter could be used only on an exceptional basis in the presence of an ideal draining system and with irrigation flushing.

In his report presented at the UNESCO symposium on the salinization of soils and water sources, V.A. Kovda drew attention to the fact that irrigation with mineralized waters is possible through flushing, i.e., by maintaining the outflow of water and the elimination of accumulating salts from the root stratum and only with intensively operating draining systems.

The report stated the following:

An annual extra-vegetational flushing is required in the case of a 2-3 grams per liter mineralization of irrigation waters.

One flushing irrigation after 4-5 conventional waterings is required with a 4-5 grams per liter mineralization of irrigation water.

Each second or third watering should be a flushing in the case of 7-8 grams per liter mineralization of irrigation waters.

Frequent high volume watering, exceeding the water retaining capacity of the soil, with an intensively operating draining system, is necessary with a 10-12 grams per liter mineralization of irrigation waters.

However, even soils with good drainage irrigated with water containing essentially sodium salts could solonetzize and become impermeable to water. Circular No. 969 of the U.S. Department of Agriculture states that water in which

$$\frac{Na}{\sqrt{\frac{Ca+Mg}{2}}} < 8 \qquad (1)$$

presents no soil solonetzization threat.

The cation content in formula (1) is in milligram-equivalents per liter.

O. Izrael'sen suggests the following classification of irrigation waters based on the percentage of sodium content in the sum total of cations (in milligram-equivalents per liter): good, less than 65%; satisfactory, 66-75%; poor, over 75%. Using the same principle A.M. Mozheyko and T.K. Vorotnik suggest a similar classification of irrigation waters: good, under 65%; satisfactory, 66-75%; poor, over 75%.

M.F. Budanov recommends for irrigating the territory of the southern part of the Ukrainian SSR the use of waters with a mineralization not exceeding 3 grams per liter providing that the overall mineralization (in milligram-equivalents per liter) divided by the value of the hardness should not exceed 2 for medium and heavy loamy soils, 2.5 for light loamy soils, and 3 for sandy loam and sandy soils.

I.N. Antipov-Karatayev and G.M. Kader believe that the strictest possible determination of constants of adsorption exchange between the soil absorbing complex and soil solution should be computed on the basis of the following equation:

$$\frac{\sqrt{x(Ca+Mg)}}{X_{Na}} = \frac{K\sqrt{C(Ca+Mg)}}{C_{Na}} \qquad (2)$$

in which X is the content of absorbed cations, and C is the content of cations in irrigation water. This equation is the basis for formula (1) for the irrigation rating of the water. Should the water contain a substantial amount of potassium salts it is added to the sodium content in determining the quality of the irrigation water.

Experience in irrigating with waste water with a mineralization of up to 3 grams per liter in different soil and climatic zones in the USSR has shown that all the above-enumerated methods could be used in assessing the quality of irrigation water.

POSSIBILITY FOR THE DECONTAMINATION OF WASTE WATER POLLUTED WITH ORGANIC MATTER ON IRRIGATED FIELDS

The decontamination of waste water on irrigated fields and at artificial biological treatment systems consists of developing living organisms of organic matter as a source of carbon nutrition. This offers a number of advantages compared with artificial treatment systems, based on the following principal conditions of irrigated fields:

a) Organically polluted waste water goes through 2-3 biological barriers (soil microorganisms, plants, domestic animals).

b) With its tremendous surface of soil particles, colloidal in particular, the soil has great absorbing capacity as a result of which standard irrigation and sprinkling norms exclude the pollution of ground waters and lower the concentration of organic matter in the soil solution of ground waters and lower the concentration of organic matter in the soil solution. This considerably facilitates the development of biochemcial processes.

c) the existence in the soil of a tremendous quantity and great variety of microorganisms under normal water-air and food systems, required for the cultivation of farm crops, is a guarantee of the fast mineralization of water soluble organic matter contained in waste water.

d) A sufficient period of time of 20 to 30 days is available for biochemical contamination in the use of waste water for irrigation, between the watering and the harvest. Such a period of quarantine for all farm crops (fodder and grain), meeting the regulations on irrigated farmland, has no substantial influence on crop sizes. Whereas such a period of time results in the total self decontamination of waste water released in water reservoirs, this applies even more so to irrigated farmland.

e) Outlays for the building and operation of an irrigation network are compensated by higher yields.

Reaching the soil with the waste water, the organic pollutants may greatly harm the soil microflora and vegetation. It has been biologically established that the toxicity of substances is manifested through their dose or concentration. Therefore, it would be impossible to draw a line separating medicinal from toxic substances, since in insignificant amounts inhibitors act as stimulators while stimulators in excess doses have an inhibiting impact.

Certain chemical elements (arsenic, heavy metals, and polychlorine-containing organic matter) can be removed from a living organism exceptionally slowly, for which reason they may accumulate within it. Most organic compounds entering a living organism leave it gradually. This occurs either as a result of their breakdown or deactivation or as a result of their removal from the organism, or through a combination of these and other methods. Such disintoxication is identically present in animals, plants, and microorganisms.

Whereas the basic rules governing metabolism and disintoxication in the living organism in the soil have been sufficiently developed, such development in terms of plants has occurred quite recently, starting with the application of toxic chemicals in agriculture. It turned out that organic phosphorus poisons are more toxic to all biological species compared with organic chlorine poisons. The latter, however, have cumulative properties to one or another extent, whereas organic phosphorus toxins detoxify quite rapidly in the living organism. The mineral theory of plant nutrition has been universally recognized. It was the basis for the theory of plant fertilizing. In recent years information has been developed not only on detoxification but on the absorption by plants of a great variety of organic compounds. F. Nartey has reported that various plant species have effectively absorbed rather large quantities of cyanides. Marked carbon cyanide energetically participates in the metabolic process of substances of vital importance to plants. S.V. Durmishitdze and D. Sh. Ugrekhelidze have noted the active absorption by superior plant species of butane, ethane, propane, and even a chemically strong compound such as benzene. The marked carbon in these organic matters was separated from the carbon dioxide and linked with proteins and organic acids.

The studies conducted at the VNIISSV led to the determination of energetic processes of decontamination of farm plants from acetaldehyde, crotonaldehyde, benzaldehyde, dichloroethane, carbon tetrachloride, methanol, ethanol, butanol, propanol, dimethyldioxane, and caprolactam. The concentration of such organic matter in irrigation water not exceeding

400 milligrams per liter has no physiological influence on plants.

Nevertheless, a number of difficulties remain such as the multicomponent nature of pollution in waste water and the impossibility to establish all pollutants. Therefore, presently the following system is used in determining the suitability of waste water for irrigation:

The activities of soil microflora must not be suppressed.

Groundwater must not be polluted.

Farm crop yields must be close to yields of crops irrigated with clean water.

No pathological changes must develop in animals fed over long periods of time goods grown on such irrigated areas.

The products of animals fed vegetation grown on irrigated fields must be of good quality.

USA-USSR WORKING AGREEMENT ON UTILIZATION OF WASTEWATER ON LAND

METHOD OF WINTERTIME SPRINKLER DISTRIBUTION OF WASTE WATER

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The fertilizing value of waste water and its utilization on a year-round basis on irrigated farmland (IFL) is widely known.

In the course of the distribution of waste water on the fields through sprinkling, at below freezing air temperatures the water freezes and, in the course of the spring thawing period, the danger appears that the thawed waste water will be discharged from the fields and thus pollute surrounding waters.

The accumulation of waste water and its distribution in winter through other methods involves considerable capital outlays and technical difficulties. The specific nature of irrigation with waste water, particularly on the basis of a year-round system, calls for the creation of spraying systems with a maximum level of mechanization and automation of the irrigation process which could be achieved presently most completely through sprinkling.

The laboratory for the automation of spraying systems with the use of waste water of the VNIISSV has elaborated a method for the distribution of waste water through sprinkling in wintertime that reduces the outflow of thawed waste water from the irrigated lot and insures the even penetration of the thawed water in the soil along the entire irrigated sector.

This is achieved by distributing the sprinkled waste water at below freezing air temperatures not evenly on the entire area but only on the area where shafts of different configuration have frozen, distributed on the irrigated section where border strips have formed (a.s. No 480378).

The shape of the frozen shafts depends on the topography of the area, the sprinkling equipment used, the operational sequence and the design of the sprinklers.

This method for the distribution of waste water on irrigated fields in the winter may be implemented both through the use of DDN-70 sprinklers, as well as stationary sprinkling systems with single or group control of the sprinklers.

Unlike the dense irrigation, in the winter, at below freezing air temperatures, the sprinklers (DA-2 type) of the stationary sprinkling systems are turned on in a checkerboard sequence (every second sprinkler). The use of this method with stationary sprinkling systems is possible also through the utilization of special sprinklers for year-round irrigation with waste water and livestock farm effluents.

On the basis of the studies of serially produced sprinklers and of foreign designs the VNIISSV has developed a special sprinkler (a.s. No 412943) for year-round irrigation with waste and livestock effluents. The rotation of the machine is accomplished through the reactive action of the jet. The implementation of this method with the sprinkling apparatus (the freezing of shafts with the formation of closed basins) takes place by removing for the winter period the jet spraying system (close irrigation system). In other words, at below freezing air temperatures no close sprinkling takes place (see Figures 1-2). The ice coating is done when wind velocity does not exceed 2.5 meters per second.

In the spring the shafts of waste water ice are gradually thawed (thawing is up to 10 days slower compared with unirrigated sections) and the bulk of the thawed waste water penetrates the soil through the basins created by the ice.

The use of this method for the distribution of waste water through sprinkling in the winter offers the following advantages:

The distribution of waste water by sprinkling of below freezing air temperatures with a reduced thawing loss becomes possible.

The size of the storing areas for the accumulated waste water is reduced.

The sprinkling equipment for waste water is used on a year-round basis.

Conditions for the storing of moisture and fertilizers in the soil improve.

The possibility develops for the use of mechanization and automation in the year-round distribution of waste water, which improves environmental protection conditions and lowers the cost of utilization of waste water.

Personnel from the Laboratory for Automation of Sprinkling Systems Using Waste Water of the VNIISSV participated in the development of this method and experimentation with it over a 2-hectare lot (experimental stationary system).

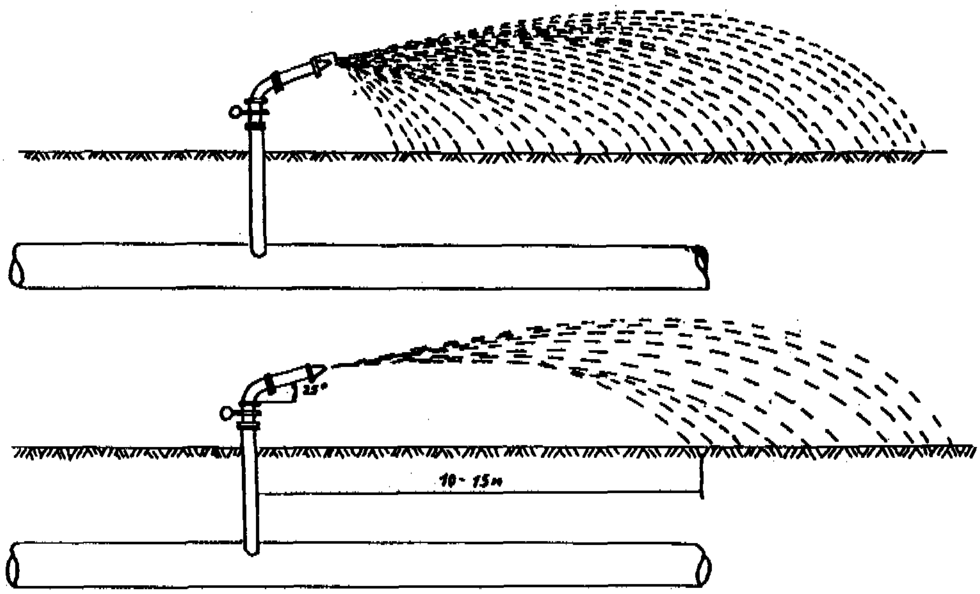

Figure 1. Schematic of the sprinker apparatus during accumulation of frozen waste water.

a. Summer spraying.

b. Spraying at below-freezing temperatures. Accumulation of frozen waste water.

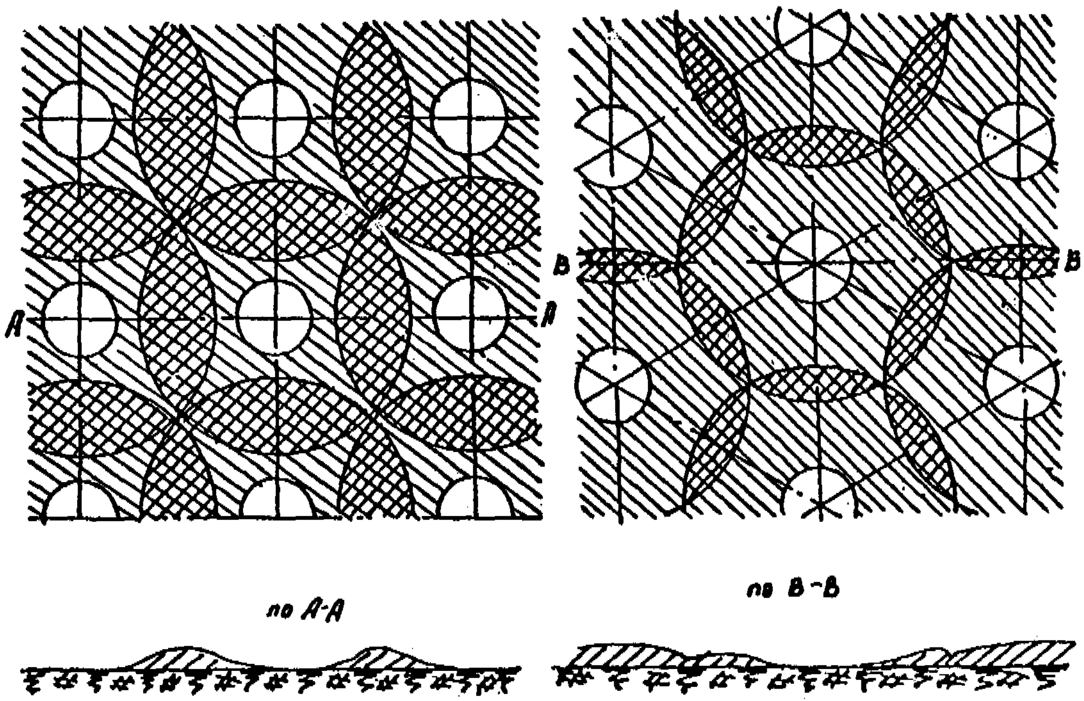

Figure 2. Frozen Waste Water accumulation due to spraying.

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