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Design and Operation of Land Treatment Systems for Minimum Contamination of Ground Water[a]

by Herman Bouwer[b]

ABSTRACT

Low-rate or irrigation-type systems for land application of sewage effluent or similar wastewater are often used in humid areas because they have a small impact on the underlying ground water. In arid areas, low-rate systems cannot be used to produce renovated water for ground-water recharge, because the renovated water will have a much higher salt content than the effluent. Renovated water of relatively low salt content can only be produced with high-rate systems. Such systems, which require permeable soil, can also be used in humid areas to reduce the land requirements. To minimize the impact of high-rate systems on ground-water quality, the system should be managed to remove as much of the pollutants (particularly nitrogen and phosphorus) as possible from the wastewater as it seeps through the soil, and to restrict the spread of renovated wastewater in the ground-water basin. Nitrogen removal can be maximized by stimulating denitrification in the soil. Certain soils can store large quantities of phosphate. The spread of renovated water in the ground water can be controlled by intercepting the flow of renovated water with wells or drains for reuse or discharge into surface water. Techniques for predicting the underground flow system are presented.

INTRODUCTION

The "no-discharge" policy of the Federal Water Pollution Control Act Amendment of 1972 will undoubtedly cause a stepped up interest in land disposal of liquid wastes such as conventionally treated sewage effluent, processing-plant wastes, animal wastes, etc. Although the soil mantle is an effective filter system that causes considerable improvement in the quality of the wastewater as it moves down to the ground water (McGauhey and Krone, 1967; Kardos, 1967; Bouwer *et al.*, 1974b), the quality of the resulting "renovated" water usually will not be as good as that of the native ground water. Thus, the degree and spread of contamination of the existing ground-water resources by renovated wastewater must generally be minimized.

One way to minimize contamination of existing ground water is to apply only small amounts of wastewater per unit area and grow a crop or other vegetation on the disposal field. If, for example, secondary sewage effluent is applied to vegetated land at a rate of about 1 inch/week, the total nitrogen load is not much more than the amount removed from the soil by crop uptake and subsequent harvest (Kardos, 1967). Other elements, such as phosphorus, metals, etc., are also taken up by the crop, which reduces their rate of accumulation in the soil. Thus, the effluent water that seeps beyond the root zone and reaches the ground water as renovated water is of fairly good quality. In humid areas, dilution of the renovated water by rainfall may also be a significant factor.

The disadvantage of these so-called "low-rate" systems is that they require large land areas (at 1 inch/week, about 260 acres are required per 1 million gallons/day of wastewater). Where large volumes of wastewater are to be disposed, the social obstacles associated with large disposal areas

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may be insurmountable. Thus, it is tempting to apply more wastewater per unit area, particularly if permeable soils are available that can take the wastewater at high infiltration rates. With such "high-rate" systems, wastewater may be applied at rates of several feet to several yards per week.

In arid regions, low-rate or irrigation-type systems are not suitable to produce renovated water for ground-water recharge. This is because the salt content of the renovated water passing through the root zone and seeping deeper into the ground to the water table contains much more salt than the wastewater used for irrigation. The salt content of the renovated water can be estimated with the salt balance equation (Bouwer, 1969), which in its simplest form can be written as

$$C_i D_i = C_d(D_i - D_e) \tag{1}$$

where

C_i = salt concentration of wastewater used for irrigation,

D_i = amount of irrigation water applied,

C_d = salt concentration of water percolating from the root zone to the ground water, and

D_e = amount of water used by crop for evapotranspiration.

The amount of water percolating to the ground water, or renovated water, is $D_i - D_e$. With normal irrigation practices, D_i is about 30 percent more than D_e. According to equation (1), this yields a salt concentration in the renovated water that is 4.3 times higher than that in the irrigation water. Even if 3 times as much water is applied than needed by the crop, C_d would still be 1.5 C_i.

The soluble salts of the renovated water include nitrogen, which will mostly be in the nitrate form because of the predominantly aerobic soil conditions in irrigated fields. If the salt concentration in the renovated water is much higher than that of the wastewater, as can be expected for low-rate systems in warm, arid climates, the nitrate concentration may also be higher than in the original wastewater, particularly during times when crop uptake of nitrogen is low. Increased nitrogen and salt concentrations in the ground water below sewage-irrigated fields are commonly observed (Schmidt, 1972; Wells and Sweazy, 1973).

In view of the possible high salt and nitrate contents, water draining from sewage-irrigated fields in arid regions is not suitable for ground-water recharge. To obtain renovated water that does not have a significantly higher salt concentration than the effluent, much more water must be applied than needed for evapotranspiration of the crop. This is achieved with high-rate infiltration systems where effluent applications may be 10 to 100 times as much as the evapotranspiration.

High-rate systems require only a fraction of the land area needed for low-rate systems. Being essentially a point source, however, high-rate systems have a much greater impact on the ground water. To minimize the effect on the ground water, high-rate systems should be designed and operated (a) to obtain renovated water of the best possible quality (particularly as regards the nitrogen and phosphorus content), and (b) to restrict the spread of renovated water into the native ground water.

For both low- and high-rate systems, the wastewater is applied to the land in intermittent fashion, rotating infiltration periods with dry or "resting" periods to allow recovery of the infiltration rates and entry of oxygen into the soil (Bouwer *et al.*, 1974a). The wastewater may be applied to the land with sprinklers or, if the topography permits, with basins or furrows (Kardos, 1967; Bouwer, *et al.*, 1974a).

MAXIMIZING NITROGEN REMOVAL

If sewage effluent is applied to land with a high-rate system, the nitrogen load may be much greater than the few hundred pounds that can be removed per acre per year from the soil by growing and harvesting a crop (at an application rate of 1 foot/day, the nitrogen load may be 25,000 pounds/acre per year!). The only process whereby nitrogen can be removed from the soil in quantities much greater than what crops can take out is denitrification. This is a microbiological process whereby nitrate in the soil is reduced mainly to free nitrogen gas, which returns to the atmosphere. The process requires anaerobic conditions and the availability of organic carbon as an energy source for the denitrifying bacteria. Thus, where the nitrogen load exceeds the amount that can be removed by a crop, the system should be designed and managed to stimulate denitrification. How this should be accomplished depends on the form of the nitrogen in the wastewater and on the amount of organic carbon available in the water or in the soil. If the nitrogen occurs as nitrate in the wastewater and there is sufficient organic carbon, denitrification in the soil can be promoted by continuing the wastewater application sufficiently long to cause oxygen depletion in the soil.

For conventionally treated sewage effluents, the nitrogen is mostly in the ammonium form at concentrations of 20 to 40 mg N/liter. If this effluent is frequently applied in small amounts, the upper portion of the soil profile will be sufficiently aerobic for essentially complete conversion of the nitrogen to the nitrate form (Bouwer *et al.*, 1974b; Lance and Whisler, 1972; Lance *et al.*, 1973). At the same time, the organic carbon in the wastewater, which is generally at low concentrations for secondary effluent, will also become completely oxidized. Thus, little or no denitrification can be expected as the nitrates move down with the water to anaerobic zones, because there is no organic carbon available for the denitrifying bacteria. The nitrogen will then remain in the highly mobile nitrate form, which can result in nitrate levels in the ground water that exceed the maximum permissible limit of 10 mg/liter nitrate-nitrogen for drinking water. If the organic carbon level in the wastewater is high, however, such as in certain liquid animal wastes, sufficient organic carbon can be left after the wastewater passes through the aerobic zone to stimulate active denitrification further down (Erickson *et al.*, 1971).

To maximize denitrification in soil receiving secondary sewage effluent in which the nitrogen is mostly in the ammonium form and the organic carbon levels are usually fairly low, the effluent should be continuously applied for a sufficiently long period to cause oxygen depletion in the soil. The ammonium is then no longer converted to nitrate, and it can be adsorbed by the clay and organic matter in the soil. Before this cation-exchange complex is saturated with ammonium, the application of wastewater should be stopped so that the soil can drain and dry. Oxygen entering the soil will then cause the adsorbed ammonium to be nitrified, after which denitrification occurs in (micro) anaerobic zones. If wastewater is then applied again, the nitrate-enriched capillary water mixes with the incoming wastewater which contains organic carbon, and denitrification can further occur when anaerobic conditions are reached.

Thus, whereas short, frequent flooding (2 days wet, 3 days dry, for example) of infiltration basins with secondary sewage effluent gave essentially complete conversion of the nitrogen to nitrate, flooding periods of 2-3 weeks alternated with drying periods of about the same length gave approximately 30 percent removal of nitrogen in a pilot high-rate land filtration system west of Phoenix, Arizona (Bouwer *et al.*, 1974b; Lance and Whisler, 1972). This amounted to a total nitrogen removal of about 8,000 pounds/acre per year, which is much more than can be removed by a crop. Additional research has shown how the systems should be managed to obtain even greater removal of nitrogen by denitrification (Lance and Whisler, 1973). This may require more careful manipulation of application rates, adding organic carbon to the effluent or to the soil, growing certain crops and incorporating the crop residue into the soil, or additional treatment of those portions of the renovated water that contain most of the nitrate (including recycling through the infiltration system).

MAXIMIZING PHOSPHORUS REMOVAL

The phosphorus content in conventionally treated sewage effluent usually ranges between 5 and 20 mg/liter with 10 mg/liter probably a reasonable average. The phosphorus is mostly in phosphate form, which in acid soils is adsorbed by iron and aluminum oxides. After adsorption, the phosphates may slowly change into insoluble compounds (Ellis, 1973). Thus, evaluating the capacity of a soil to remove phosphate by quick adsorption tests in the laboratory may underestimate the long-term removal capability of the soil, and hence, the useful life of the land treatment system.

In calcareous sands and soils, phosphates may precipitate as calcium phosphate compounds. At the Flushing Meadows Project (Bouwer, *et al.*, 1974b), for example, phosphate removal was about 50 percent after 30 feet of underground movement and as much as 90 percent after 100 feet. Phosphate removal decreased with increasing application rate, particularly in the first 30 feet of underground travel. Phosphate removal efficiency was still stable after 5 years of operation of the project, during which a total of about 43,000 pounds of PO_4-P were applied per acre. Thus, the soil has a tremendous capacity to fix and store phosphates.

More research, however, is needed on the kinetics of phosphate immobilization in soil before the removal of P in high-rate land treatment systems can be accurately assessed.

Phosphate precipitation may be enhanced by adding lime to the soil. Also, it may be feasible to dose the effluent with iron chloride or other phosphate precipitant prior to infiltration. The phosphate precipitates will then accumulate on the soil surface from where they can be removed by periodic scraping, or worked into the soil by cultivating.

Sometimes it may be desirable to reduce the nitrogen and phosphorus content of the effluent

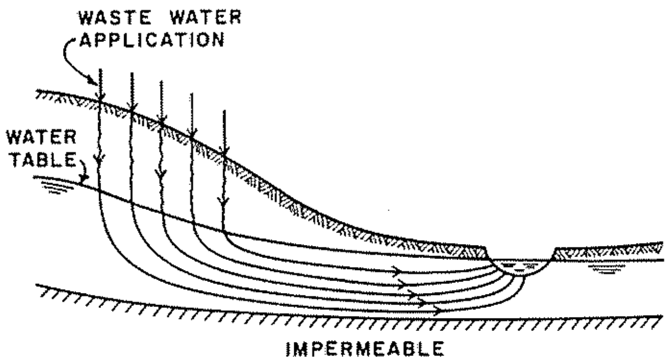

Fig. 1. Renovated wastewater draining naturally into surface water.

prior to land application. This "pretreatment" may range from lagooning for nutrient stripping to in-plant tertiary treatment. In the latter case, land application would mainly be used for polishing treatment and underground storage.

RESTRICTING SPREAD OF RENOVATED WATER

To minimize the spread of renovated wastewater into a ground-water basin, the renovated water should be taken out of the aquifer at some distance from the place where it reaches the ground water. This may happen naturally if the ground water drains to a stream or lake (Figure 1). If the renovated water does not leave the aquifer in a natural manner, it should be collected by drains (for shallow aquifers) or wells (for deep aquifers) to limit its spread into the aquifer. After collection (and additional treatment if necessary), the renovated water may be used for irrigation, recreation (including lakes), industrial, and perhaps municipal purposes, or it may be discharged into surface water. With such a system, a portion of the aquifer is essentially used as a natural filter.

The proper distance between the points where the wastewater enters the soil and where it leaves the aquifer as renovated water depends on the type of wastewater, the desired quality of the renovated water, and the nature of the soil and aquifer materials. For granular soils and aquifers, underground travel distances of several hundred feet and underground detention times of several weeks may be sufficient to yield a renovated sewage effluent that is free from microorganisms, BOD, suspended solids, and taste and odor, and that is reduced in nitrogen, phosphorus, fluorine, metals, and other substances. The more time and distance allowed for underground travel, the better will be the quality of the renovated water, at least to a certain limit. Most of the quality improvement, however, takes place in the first 3 or 4 feet of the soil profile.

The simplest protection against pathogenic organisms in renovated water, however remote the chance for long-distance transport of such organisms in well-selected soil and hydrogeologic formations would be, may be to chlorinate or otherwise disinfect all ground water for human consumption that is collected within underground traveling distance from land treatment sites. Most water-borne diseases in the U.S. are caused by undisinfected ground water anyway, and disinfection of all ground water for human consumption has been recommended as a simple and effective means to reduce the incidence of such diseases (Craun and McCabe, 1973).

Deep Aquifers

When the aquifer is unconfined and relatively deep, a "closed" wastewater renovation system can be obtained by locating the areas where wastewater is applied in two parallel "infiltration strips" and pumping the renovated water from wells midway between the strips (Figure 2). Other possibilities are a single infiltration strip with wells on both sides, or a central infiltration area surrounded by a ring of wells (Figure 3). The wells in the two systems of Figure 3 will pump a mixture of native ground water and renovated wastewater. This may be desirable if the use of pumped water requires dilution of the renovated water anyway. However, if the wells should pump renovated water only, the additional native ground water also pumped in-

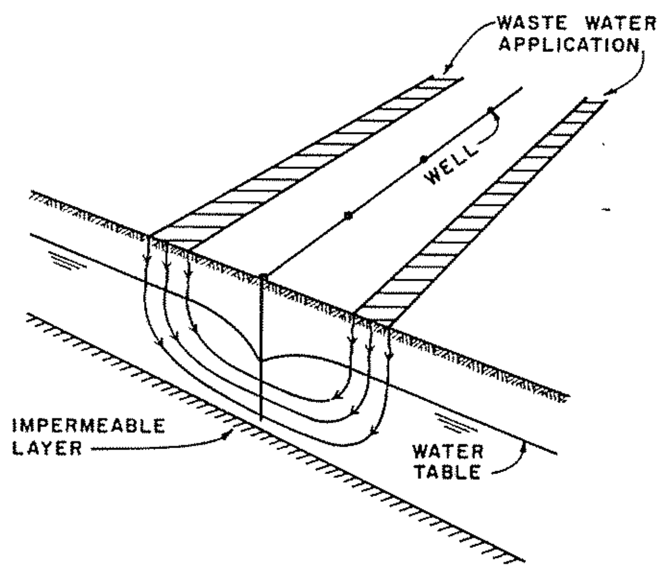

Fig. 2. Schematic diagram of two parallel strips (hatched areas) for applying wastewater, and wells midway between the strips for pumping renovated water.

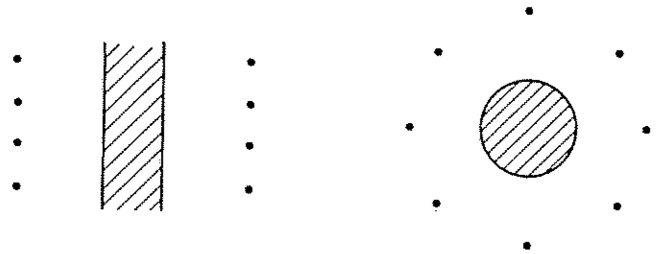

Fig. 3. Long infiltration strips (hatched area) with wells on both sides (left) and circular infiltration area surrounded by wells (right).

creases the pumping costs and constitutes an extra draft on the native ground water.

The design and operation of a wastewater renovation system consisting of two parallel infiltration strips and wells midway between the strips (Figure 2) should be based on the following three criteria:

1. The water table below the infiltration strips should not rise to field surface, where it can restrict the infiltration rates. The water table preferably should not come closer to field surface than a distance of about 4 feet. This depth enables rapid drainage of the soil profile, and thus entry of oxygen, when infiltration periods are rotated with dry or resting periods.

2. All wastewater that has infiltrated should be pumped as renovated water from the wells. No renovated water should move into the aquifer outside the system of infiltration areas and wells.

3. The renovated water should have traveled the proper time and distance underground when it reaches the wells.

In order to investigate whether a certain design meets these three criteria, the underground flow system must be predicted. This prediction will also yield an estimate of the pumping lift in the wells.

The prediction of the underground flow system for renovation systems such as those in Figure 3, requires knowledge of the rate of entry of wastewater into the soil and of the hydraulic properties of the aquifer. The infiltration rates may be evaluated by local experimentation to see what hydraulic loading rate can be maintained, regardless of whether the wastewater is applied with sprinklers, basins, or furrows. The main hydraulic property to be evaluated for the aquifer is the effective transmissivity for ground-water recharge, which will govern the flow system under and near the infiltration strips.

The effective transmissivity for ground-water recharge is less than the total transmissivity of the aquifer, particularly for relatively deep, unconfined aquifers, because recharge flow systems are characterized by an upper active zone and a lower passive zone (Bouwer, 1965, 1970). The effective transmissivity for recharge depends on the width of the infiltration strip. It increases essentially linearly with width until it has become equal to the total transmissivity of the aquifer. Once the underground flow has become mainly horizontal, as it does in the vicinity of the wells (Figure 2), the total transmissivity of the aquifer can be used to analyze the rest of the flow system (Bouwer, 1970). If the wells do not completely penetrate the aquifer, the appropriate correction factors should be applied to the total transmissivity.

A good way to evaluate the effective transmissivity of an aquifer for ground-water recharge is to determine the response of ground-water levels to infiltration, as may be done in an experimental recharge project. This was done for the Flushing Meadows Project in the Salt River bed west of Phoenix, Arizona, where renovation of secondary sewage effluent by land application is studied with six parallel recharge basins covering a block measuring 220 by 700 feet (Bouwer *et al.*, 1974a and b). Two observation wells, one 30 feet deep and the other 100 feet deep, were installed in the center of this block. The observed response of the water levels in these wells to infiltration was simulated on an electric analog, which then indicated the hydraulic conductivity of the aquifer in vertical and horizontal directions. The resulting values agreed with data obtained from direct permeability measurements at the seven observation wells in the project (Bouwer, 1970).

With known directional permeability components of the aquifer, the theoretical shape of the ground-water mound was evaluated by electric analog. The Dupuit-Forchheimer theory was then applied to this mound to obtain the effective transmissivity of the aquifer for the recharge flow system, which was only 11 percent of the total transmissivity (Bouwer, 1970). This effective transmissivity, corrected for the width of the infiltration strip, was used in analog analyses of flow systems for the prototype system (Figure 2) to predict the shape of the water table and to construct a network of streamlines and equipotentials (Bouwer, 1970). When a certain porosity of the aquifer material was assumed, the macroscopic velocities of the water from one equipotential to the next could be determined for each stream tube, which in turn yielded

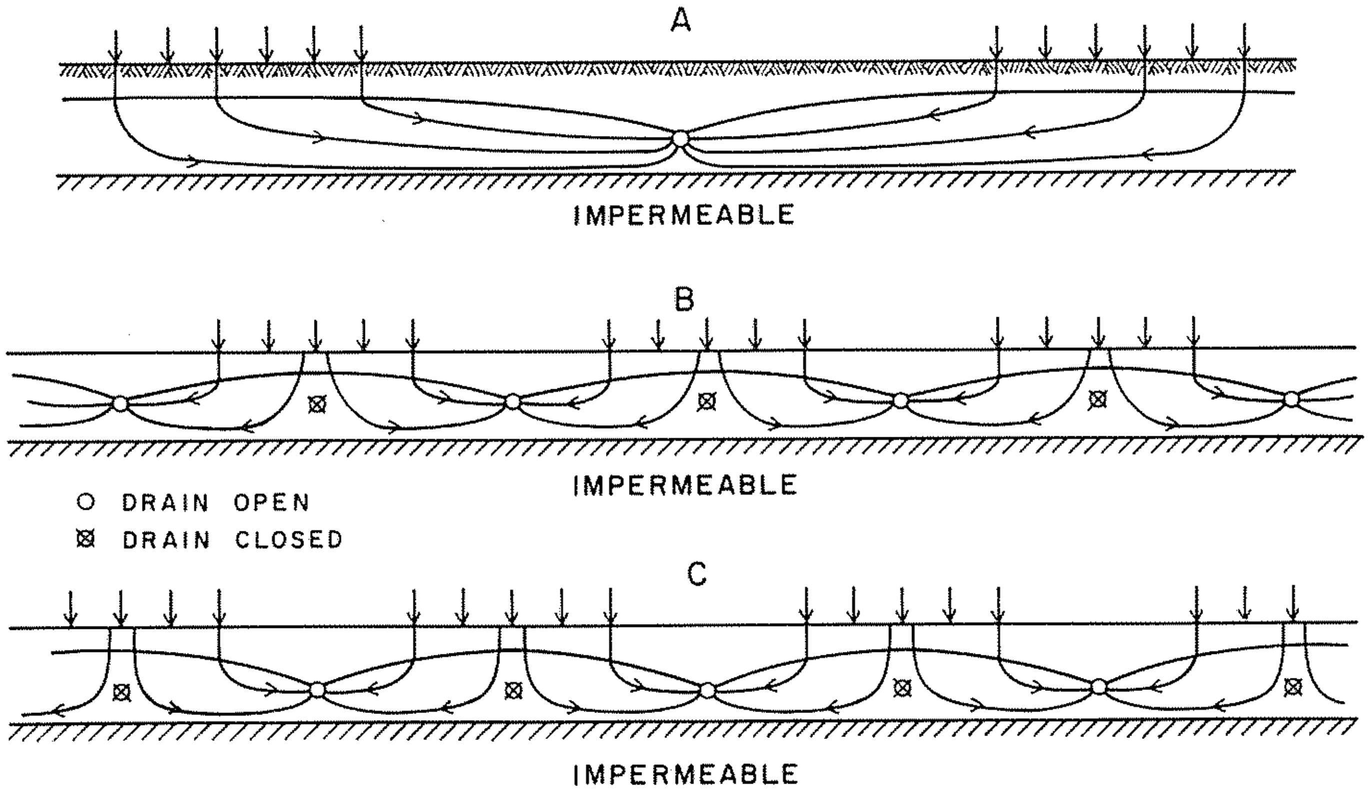

Fig. 4. Two parallel infiltration strips with drain midway between strips (A) and continuous system of infiltration strips and drains with alternate infiltration and drying (B and C).

estimates of the total underground travel time of the renovated water (Bouwer, 1970). The procedure was applied to various designs so that the optimum layout of infiltration areas and wells could be selected. Similar procedures can be applied to the design of other high-rate, closed, wastewater renovation systems.

Shallow Aquifers

If the water table and the impermeable layer are relatively close to field surface, wells may not be effective and the renovated water can be collected better by open or closed drains. The system can consist of two parallel strips where the wastewater is applied to the soil with a drain midway between the strips (Figure 4A), or of a series of infiltration strips and drains (Figure 4B). Since infiltration periods are usually rotated with drying periods, short underground travel distances and detention times can be avoided in the system of Figure 4B by closing the drains below the strips receiving wastewater and collecting the renovated water with the drains below the drying strips. These drains are closed and the other drains opened when infiltration and drying periods are rotated (Figure 4C).

The water table in the systems of Figure 4 should not rise so high that it reaches the soil surface in the infiltration areas and reduces the infiltration rates. The shape of the water table in these systems can be calculated with drainage theory (Bouwer and van Schilfgaarde, 1963). By using the Dupuit-Forchheimer assumption of horizontal flow and by assuming a uniform infiltration rate, the following equation can be derived for the system below the infiltration area (Figure 5).

$$IX = -K \frac{H_c + H_e}{2} \frac{dH}{dX} \qquad (2)$$

where

I = infiltration rate (length/time),

X = horizontal distance,

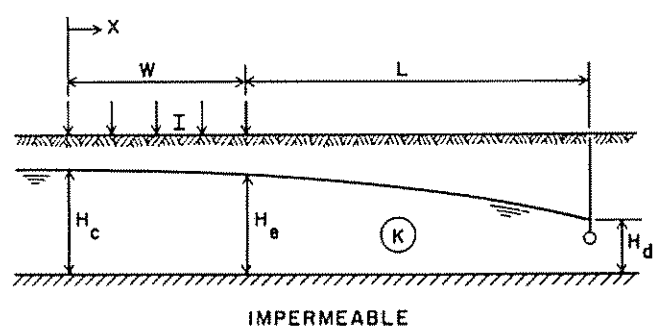

Fig. 5. Geometry and symbols for parallel infiltration strip and drain.

H = vertical distance between water table and impermeable layer, and

K = hydraulic conductivity of soil (length/time).

The term H_c refers to H at the outer edge of the infiltration strip for Figure 4A and at the center of the strip for Figure 4B, while H_e refers to the water table beneath the edge of the infiltration area near the drains. The horizontal distance, X, is measured from the outer edge of the infiltration strip for the system of Figure 4A, and from the center of the infiltration strip for the system of Figure 4B. Integrating equation (1) between X = 0 and X = W (Figure 5) yields

$$H_e^2 = H_c^2 - \frac{IW^2}{K} \tag{3}$$

The flow from the edge of the infiltration strip to the drain can be described by the equation

$$IW = \frac{K}{L}\ \frac{(H_e + H_d)}{2}\ (H_e - H_d) \tag{4}$$

where H_d is H at the drain and L is the distance between the drain and the edge of the infiltration strip (Figure 5). Equation (4) can be rearranged to yield

$$H_d^2 = H_e^2 - \frac{2ILW}{K} \tag{5}$$

which, after combining with equation (3), gives

$$H_c^2 = H_d^2 + \frac{IW}{K}\ (W + 2L) \tag{6}$$

The term W refers to the longest horizontal distance of travel for the water beneath the infiltration strip. Thus, W is the entire width of the infiltration strip for the system in Figure 4A, and one-half the width of the strip for the system in Figure 4B.

If the drain is running free, H_d will be equal to the height of the center of the drain above the impermeable layer. However, if a back-pressure is maintained in the drain (as is sometimes done to exclude air and to avoid deposits of iron or manganese oxides in the drain), H_d is the height of the drain above the impermeable layer, plus the back-pressure head.

When H_d, I, and K are known, the value of H_c can be calculated for various combinations of W and L. Thus, the optimum combination of W and L whereby H_c does not exceed a preselected value can be evaluated. If the wastewater is applied to the soil in infiltration basins and the ground-water table is so high that it coincides with the water surface in the basins, equation (6) can be used to calculate the average infiltration rate in the basin.

Equation (6) applies to relatively shallow systems. Where the impermeable layer is at sufficient depth to render the horizontal-flow theory invalid, equivalent depths of the impermeable layer should be used, as is done in the design of agricultural drainage systems (Bouwer and van Schilfgaarde, 1963).

PILOT PROJECTS

Since the performance of a land treatment system depends so much on the local conditions of soil, climate, and hydrogeology, as well as on the characteristics of the wastewater itself, local experimentation and pilot projects are usually needed if local experience is not available. Such projects can also be used to evaluate hydraulic properties of the soil and aquifers necessary for the design of the full-scale project. Once a large-scale project is in operation, good management and monitoring of the system so that undesirable performance can be corrected before too much damage is done, are essential.

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