ESTIMATING GROUND-WATER QUALITY IMPACTS
FROM ON-SITE SEWAGE TREATMENT SYSTEMS

B.J. Bauman W.M. Schafer

The primary objective of on-site wastewater treatment systems is to provide acceptable treatment and disposal of domestic wastewater with minimal impact upon the quality of local water resources. The properties of the soil resource play a vital role in determining what these potential impacts may be, but it is important to also consider the properties of the ground-water system, which serves as the ultimate medium of disposal for the treated effluent, and its relation to local surface water bodies (streams and lakes). It is the integrated properties of these three components (soils, ground water, and surface water) which determine the potential of the natural system to assimilate man-made wastes while minimizing threats to public health and environmental quality. There are two principle issues of concern: the ability of the soil to accept and treat septic tank effluent; and the impact of the treated effluent upon the local surface and ground-water resources.

Historically, the determination of land suitability for septic systems has been a site specific process, with emphasis on the evaluation of soil properties for acceptance and treatment of the effluent produced from an individual septic system (Anderson et al., 1982), and little consideration given to the possible impacts on water quality from a single system or many systems. This in spite of the fact that a major problem of ground-water contamination from septic systems was documented 25 years ago, and many more have come to light in recent years (Fabryka, 1978). A recent study (Pye et al., 1983) stated that septic systems are both the most frequently reported source of ground-water contamination in the United States, and the single largest source (by volume) of wastewater discharged to ground water.

Even with recent increased concern for threats to the ground-water resource, it appears that in general, little distinction has been made between the isolated impact of individual septic systems, and the cumulative, long-term impact of hundreds or thousands of systems upon local or regional ground-water quality. This problem needs to be addressed in order to effectively manage and protect the quality of the ground-water resource. It is proposed that there is a tolerable "loading rate" of nitrates to an aquifer, which, if not exceeded, will prevent ground-water nitrate concentrations from reaching unacceptable levels.

Ground-water pollution from septic systems may develop in an insidious manner. The first 10, 100, or even 1000 septic systems in an area may not create any apparent degradation in water quality, as the ground-water system may have sufficient dilution capacity to mitigate the impact of the added nitrate loading. If another 1000 systems are added, some contamination may become noticeable, but still remain within acceptable limits. However, the addition of another 1000 septic systems might overload the system, exceeding its capacity to adequately dilute the incoming nitrates, resulting in an intolerable level of pollution. Again, the contribution of any one septic system is relatively innocuous, yet their cumulative impact may be substantial.

The authors are: B.J. Bauman, Graduate Research Assistant, W.M. Schafer, Extension Soils Specialist, Dept. of Plant & Soil Science, Montana State U.

There appears to be insufficient recognition that aquifers have a limited capacity to acceptably dilute a continuously increasing amount of wastewater nitrate. Perhaps this is because regulatory agencies either do not have the technical expertise to evaluate such potential problems, and/or are not used to making policies in terms of long-range impacts that are difficult to quantify. It is difficult to prove that such contamination may occur, and in many areas continued land development has resulted in increases in the density of and contaminant loading from septic systems. Given these concerns, the intent of this discussion is to: 1) examine the possible ground-water quality impacts of nitrate loading from septic systems, and the factors influencing such impacts; 2) propose the addition of aquifer assessment criteria to the site evaluation process.

MODEL DEVELOPMENT

There are a number of complex numerical models which have been developed to predict pollutant flow within and impact upon aquifers (Perkins, 1984), but in general they demand a high level of mathematical competence, computer access, and often require a more detailed knowledge of aquifer characteristics than is available (e.g. values for coefficient of storage, dispersivity). Given the complexity and diversity of the major components that influence such models (e.g. soils, geology, climate), it would seem unlikely that detailed models developed in a single area can be applied successfully to other areas. Further, local governing bodies, public health officials, sanitarians, planners, (the people who must make the decisions), typically do not have the resources that would allow them to use such models. They could benefit from a simplified model that would illustrate the mechanics of the natural system of interest, and help them estimate the potential impacts of septic systems on ground-water quality.

With these considerations, the model presented here makes it possible to: 1) obtain a first approximation of the potential impacts of septic system nitrates on ground-water quality; 2) evaluate which physical properties most greatly influence the capacity of an aquifer to adequately dilute septic system nitrates; 3) gain useful information for comparing the relative susceptiblity of various aquifers to nitrate contamination from septic systems. The goal of this model is not to accurately predict ground-water nitrate levels. Rather, through use of the model, local regulatory personnel can develop a better understanding of the physical and chemical processes that determine the ground-water quality impacts of suburban and rural use of septic systems.

The model employs both the approach of Mercado (1976), whose analysis of ground-water pollution in Israel reduced a geographically broad and complex system into a single cell, as well as the mass balance and the steady state concepts of Fried et al. (1976), as applied to the influence of the agricultural nitrogen budget on ground-water quality. Other simplifying assumptions are used to facilitate development of a model that can be utilized in the decision-making process. The initial step is to define a geographical area or unit of interest, i.e. determine the boundaries of the unit system. Then two mass balance equations must be developed, one for the water budget of this system, and one for the nitrogen budget. By determining the amount of nitrate entering the system (all nitrogen inputs are assumed to be in the form of nitrates), and the volume of water that is discharged at the system boundary, an estimate can be made of the average nitrate concentration of the ground water as it leaves the system. Use of the model is restricted to isotropic and homogeneous water table aquifers composed of unconsolidated sediments.

Water and Nitrogen Budgets

The model considers three main sources of water and nitrate to a ground-water system (Figure 1) which has reached a steady state condition with regards to these inputs (Fried et al., 1976). At the upgradient boundary, ground water (W_g) enters the system containing some level of nitrate-nitrogen (N_g). Both natural precipitation (hereafter referred to as natural recharge, W_r), and septic system effluent (W_e) percolate through the soil to the ground-water system, each contributing some nitrogen to the aquifer (N_r and N_e respectively). Complete mixing within a prescribed depth increment of the aquifer is assumed. No other inputs of water or nitrogen are considered. Impacts of such potential additional inputs from agricultural fertilizer, feedlots, etc. are not considered, as a large subdivision setting is assumed, and an intent of the analysis is to examine the ground-water pollution potential from septic systems alone. It is further assumed that the only output from the system is through the down-gradient boundary (i.e. there is no consumptive use of the water removed by domestic wells penetrating the aquifer—it is all returned via septic systems, and there are no other wells removing water from the system). With these preconditions, and evaluating the system over a time period of one year, values needed for mass balance considerations are simply calculated.

The ground-water volume entering the system at its upgradient booundary can be calculated using Darcy's law:

$$Q = W_g = Kb(dh/dl)w \qquad \text{Eq. 1}$$

where Q = volume of flow (m^3/yr), K = the hydraulic conductivity of the aquifer (m/yr), b = aquifer thickness (m), dh/dl = the hydraulic gradient (slope of the water table), and w = aquifer width (m). K can be estimated through knowledge of the particle size distribution of the geologic material comprising the aquifer or through field measurement techniques (Freeze and Cherry, 1978). Likewise, the hydraulic gradient can be established by field measurement of the piezometric surface at several locations, or can be estimated. The width (w) of the aquifer is defined by the areal extent of the system of interest (Figure 1).

Aquifer thickness can be set without knowledge of the actual depth of the aquifer using the rationale that limited mixing of influent nitrates occurs within the aquifer. This tendency for influent nitrate to remain stratified in the upper portions of the aquifers has been noted by both Hill (1982), and Spruill (1983). With this assumption, the depth of such mixing can be estimated to obtain a value for b, now defined as depth of mixing. If the actual aquifer thickness were used, greater dilution of nitrate would erroneously be calculated, as much of the aquifer may remain relatively isolated from the influent nitrate. For this model, b will take on the values 5, 10 and 20 m to evaluate the impact of different degrees of mixing. Complete mixing will be assumed within this prescribed thickness. Nitrate-N concentrations for ground water entering the system can be

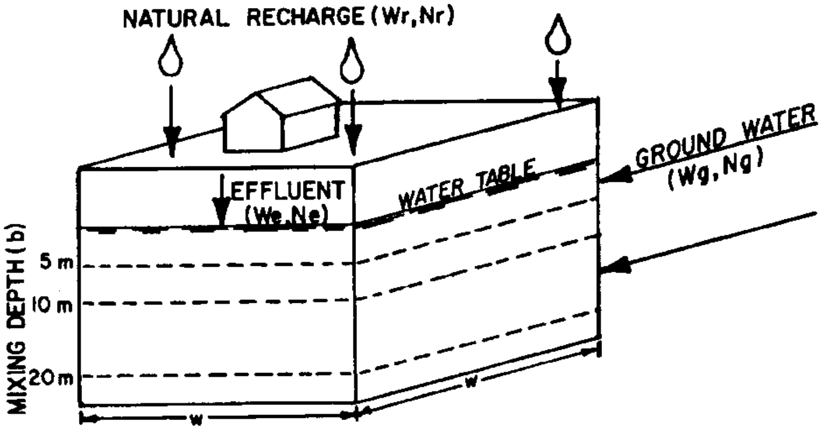

Figure 1. Schematic representation of model inputs.

obtained from existing data, or estimated. Precipitation contributing to recharge of the water table can be estimated using the method of Dunne and Leopold (1978, p. 253). Several values are used in this analysis (Table 1) representing typical values for differing climates, arid to humid. Nitrate-N concentration of this recharge water must be estimated (this model assumes 3 mg/L). The volume of effluent passing to the water table can be calculated by assuming a water use of 170 l/day/capita, and 3 people per family (from 1980 U.S. Bureau of Census figures for occupancy of rural housing units) for an average yearly water use of 186,510 L per family (186 m^3). The nitrate-N content of this effluent is taken as 62 $\pm$ 21 mg/L, the average value of 20 published studies representing more than 500 samples (Bauman, 1985). It is assumed that there are no losses of nitrogen after it leaves the septic tank, and that upon exiting the drainfield trench all nitrogen is converted to nitrate-N.

SENSITIVITY ANALYSIS

Sensitivity analysis can be used to examine the relative importance of the variables in the model. It is performed by selecting a parameter and varying its input value while keeping others constant. Parameters that produce large changes in the results can be considered sensitive, and are of greater interest than those of low sensitivity. Estimates of hydraulic conductivity and hydraulic gradient (Table 1) represent a range of values commonly encountered in aquifers serving as sources of domestic water supply. Values for other variables are also found in Table 1.

Table 1. Parameters and values used in the sensitivity analysis.

W_g: Ground water entering upgradient boundary
  Hydraulic conductivity (K) — 3154 & 315.4 m/yr
  Hydraulic gradient (dh/dl) — 0.01 & 0.001
  Aquifer mixing thickness (b) — 5, 10, and 20 m
  Aquifer width (w) — 402 m (one side of a square 16 ha (40 acre) parcel)
N_g: Background nitrate-N in ground water — 1, 3, 5, 7 mg/L
W_r: Natural recharge — 0, 10, 20, 30, 40 cm/yr
N_r: nitrate-N in recharge — 3 mg/L
W_e: Septic system effluent — 186,510 l/family/year
N_e: nitrate-N in effluent — 60, 50, 40, 30 mg/L
Housing density — lot sizes of 5, 4, 3, 2, 1, 0.5 acre

Sample Calculation (for a 40 acre development, 402 m X 402 m)

Water Budget

1) Ground water entering (W_g): K=3154 m/yr; dh/dl=0.01; w=402 m; b=10 m;

$$(3154\ m/yr)(0.01)(402\ m)(10\ m)(1000\ l/m^3) = 126 \times 10^3\ m^3$$

2) Natural recharge (W_r): assume 0.10 m/yr (2 inches)

$$(402\ m)(402\ m)(0.10\ m)(1000\ l/m^3) = 16.2 \times 10^3\ m^3$$

3) Septic system effluent recharge (W_e): (assuming one family per acre):

$$40(186{,}510\ l) = 7.46 \times 10^3\ m^3$$

TOTAL WATER INPUT (W_t): $W_g + W_r + W_e$

$$(126 \times 10^3\ l) + (16.2 \times 10^3\ m^3) + (7.46 \times 10^3\ m^3) = 149.7 \times 10^3\ m^3/year.$$

Nitrogen Budget

1) Ground water (N_g): (background concentraton of 1 mg/L nitrate-N):

$$(126 \times 10^6\ l) \times (1\ mg/L) = 126 \times 10^3\ g$$

2) Natural recharge (N_r): (assume 3 mg/L nitrate-N in recharge water):

$$(16.2 \times 10^6\ l) \times (3\ mg/L) = 48.6 \times 10^3\ g$$

3) Septic system effluent (N_e): (one family/acre, 60 mg/L nitrate-N):

$$(7.46 \times 10^6 \ l) \times (60 \ mg/L) = 447.6 \times 10^3 \ g$$

TOTAL NITROGEN INPUT (N_t): $N_g + N_r + N_e$

$$(126 \times 10^3 \ g) + (48.6 \times 10^3 \ g) + (447.6 \times 10^3 \ g) = 622.2 \ g \text{ nitrate-N}$$

NITRATE-N IN GROUND WATER EXITING SYSTEM:

$$\frac{N_t}{W_t} = \frac{622.2 \times 10^3 \ g}{149.7 \times 10^3 \ m^3} = 4.15 \ g/m^3 \ (4.15 \ mg/L) \text{ nitrate-N}$$

RESULTS

Figures 2-6 illustrate the relationship between predicted ground-water nitrate-N and housing density for various combinations of the above factors. Unless otherwise noted, the mixing thickness used is 10 m, recharge is 5 cm/yr, K = 3154 m/yr, and dh/dl = 0.01). The nitrate-N concentration of concern is 10 mg/L, the maximum allowable for drinking water (National Academy of Sciences, 1978). In each of these figures the model predicts that at higher housing densities (approaching 1 and 2 acres/lot) this standard is likely to be exceeded.

Figure 2 displays the influence of different levels of hydraulic conductivity and gradient on resultant ground-water nitrate-N concentration. Higher conductivities and gradients result in higher ground-water velocities, providing a greater diluting capacity (lower nitrate-N concentation) for the system. This effect was noted in a field study by Pitt et al. (1975) where septic system densities of up to 0.25 acres/lot caused no significant elevation of ground-water nitrates. Lower gradient and conductivity ground-water systems have little dilution capacity, and may be considered more susceptible to appreciable contamination.

Figure 3 shows the influence of mixing depth on dilution of effluent nitrates for systems with high and low ground-water velocities. In the low velocity system, mixing depth has little impact on nitrate-N concentration. A more pronounced dilution effect is seen for the high velocity system at higher housing densities, but even with small lot sizes (0.5 acre), increasing mixing depth from 5 to 20 m only decreases nitrate-N from 11 to 4 mg/L. Another parameter that has little impact in this analysis is the background nitrate-N concentration of incoming ground water (Figure 4). The upper curves representing a low velocity system are so close together they appear as one, indicating that the contribution of nitrogen from incoming ground water plays a relatively minor role in the nitrate loading of the system, The effect is more pronounced for the higher velocity system, however, most values still remain below the 10 mg/L standard. Geographical areas with higher precipitation and infiltration will be better able to dilute septic system nitrogen (Figure 5). In this model, this effect is especially noticeable when natural recharge is less than 20 cm/yr. Thus arid and semi-arid parts of the country may be at greater risk from such contamination.

The effect of lower effluent-N concentrations and/or denitrification are shown in Figure 6. The previous examples have all assumed that 60 mg/L total nitrogen is exiting the drainfield, is all converted to nitrate, all passes to the water table, and out the system's boundary. If 50% denitrification of the nitrate contributed by the effluent is assumed (equivalent to a decrease in effluent nitrate-N to 30 mg/L), it is seen that for the low velocity system, at a density of 1 home/acre, ground-water nitrate-N decreases from 28 mg/L to 15 mg/L. The high velocity system shows little impact from such a reduction in effluent nitrogen, the result of its high dilution capacity. As shown in Figure 6, if the amount of septic system nitrogen reaching the water table is reduced (for example by denitrification in the unsaturated zone), nitrate-N content of ground water will decrease, especially in low velocity systems. Denitrification can also

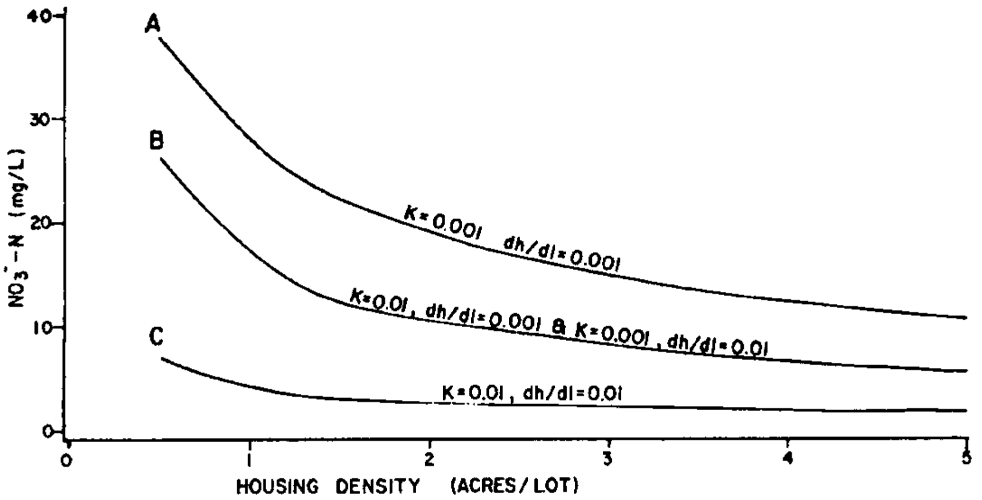

Figure 2. Influence of hydraulic conductivity and gradient on ground-water nitrate-N.

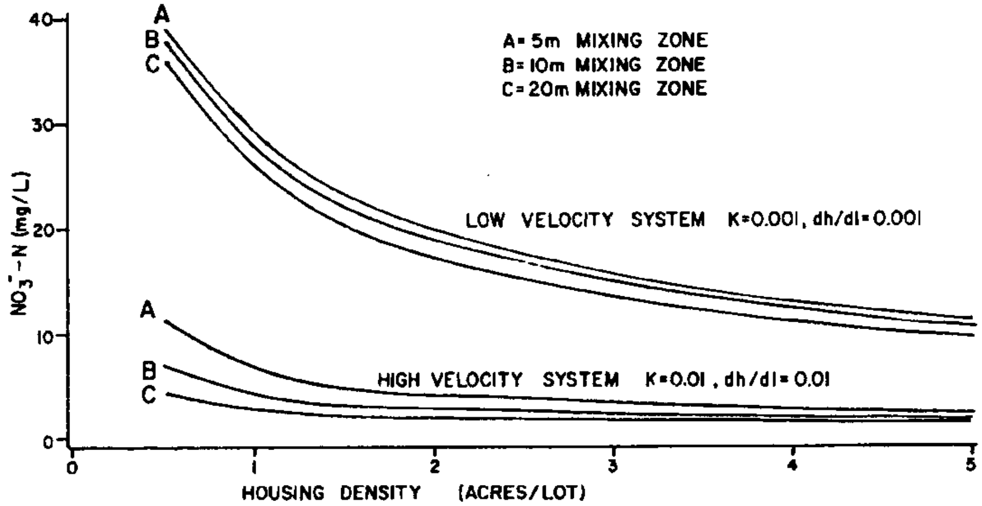

Figure 3. Influence of mixing depth on nitrate-N in ground water for a high and low velocity system.

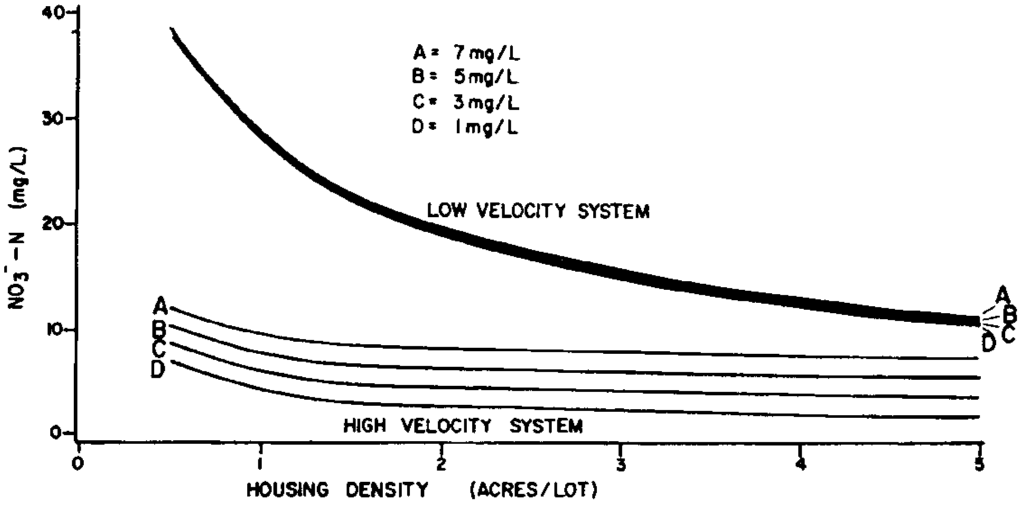

Figure 4. Influence of nitrate-N concentration of incoming ground water on resultant ground-water nitrate-N.

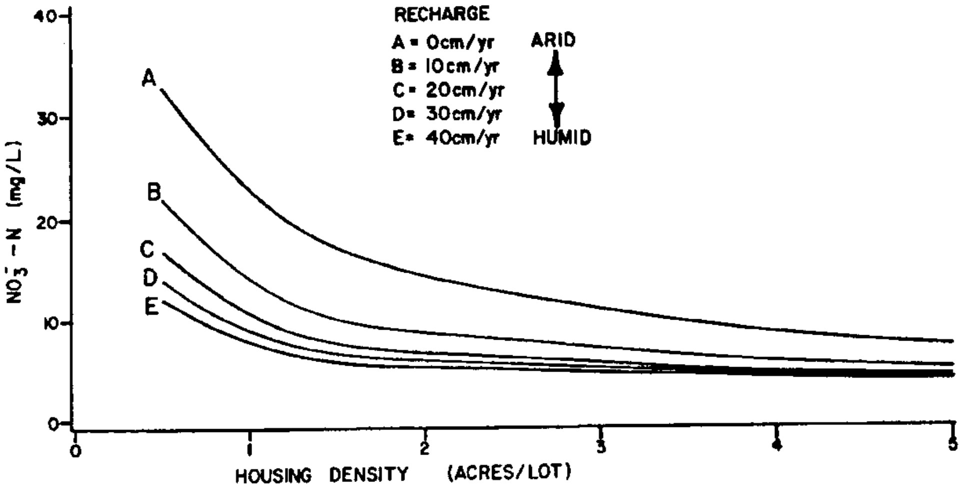

Figure 5. Influence of natural recharge on ground-water nitrate-N.

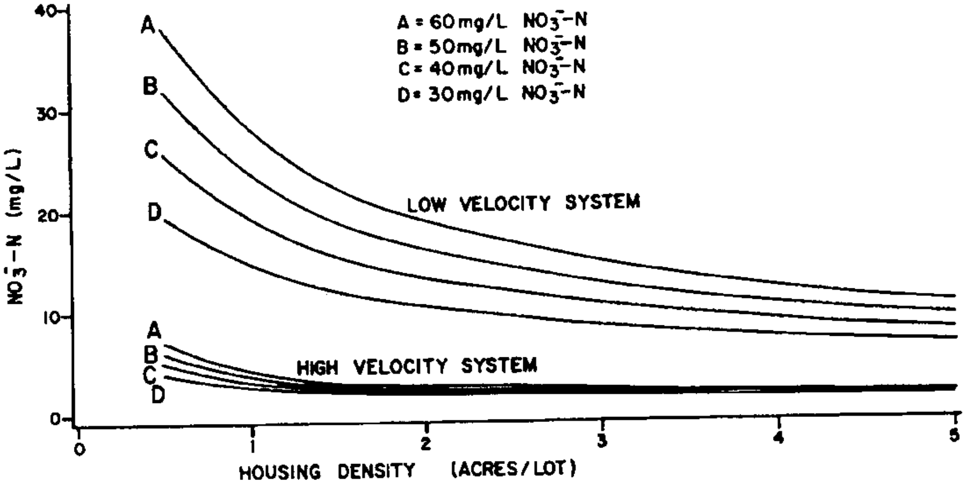

Figure 6. Influence of effluent nitrate-N concentration on ground-water nitrate-N.

occur after effluent N reaches the water table (Reneau, 1977). Factors influencing the potential for denitrification include:

1) Presence of restrictive layers between the water table and septic system. Such layers will provide local zones of saturation where anaerobic conditions may develop, promoting denitrification.

2) Temperature. Higher temperatures mean increased metabolic activity for bacteria, increasing the rate of denitrification.

3) Residence time of effluent as it travels to the water table, and as it then travels to the system boundary. Longer residence times provide a greater opportunity (i.e. more time) for denitrification processes to occur. While Figure 2 suggests systems with low ground-water velocities have higher nitrate concentrations, longer residence times will be longer, increasing the time during which denitrification may occur.

4) Dissolved organic carbon (DOC) content of ground water. Higher concentrations stimulate bacterial activity, increasing the potential for both anaerobic conditions and denitrification. In most aquifers, DOC can be considered the limiting factor for denitrification, as approximately 1.3 g of carbon are required to convert 1 g of nitrate-nitrogen to nitrogen gas (Lance, 1972). While effluent contains a 1:1 C:N ratio, most of the carbon is utilized by heterotrophic bacteria under aerobic conditions, leaving insufficient carbon for denitrification when the effluent reaches an

anaerobic environment. Champ et al. (1979) reported a median DOC content of 0.7 mg/L in a survey of American ground waters. This value would suggest denitrification potential is limited in most aquifers. However, it would seem that shallow aquifers should have higher DOC levels, as organic carbon is leached from organic matter in surface soils. Also, because of the short travel time to the water table, perhaps less of the DOC in effluent is removed. Further, at least during part of the year, water temperature will be higher. This suggests that placing septic systems in areas with shallow water tables may be desirable, if suitable protection from contamination by pathogenic micro-organisms is insured. These factors could contribute to considerable denitrification, as found by Reneau (1977).

This sensitivity analysis suggests that the following variables are most responsible for large changes in predicted ground-water nitrate-nitrogen: hydraulic conductivity and gradient, natural recharge, housing unit density, and concentration of effluent nitrate-nitrogen reaching the water table. The first three parameters determine the recharge characteristics of the aquifer, and the second two influence the nitrogen budget (loading rate) of a given system. The hydraulic conductivity of geologic materials can range from 10^2 to 10^{-7} cm/sec, and hydraulic gradient from 10^{-1} to 10^{-4}. Such order of magnitude changes imply that different ground-water systems may vary greatly in their ability to dilute incoming pollutants. Ground-water systems with high conductivities and gradients (high velocities) can tolerate larger loading rates of nitrogen, i.e. will have a greater diluting capacity. Density of septic systems, amount of natural recharge, and amount of effluent-N reaching the water table can be important parameters, especially in systems with low velocities.

A model similar to this one was applied to a field study in Illinois (Wehrmann, 1983) and provided reasonably close estimates of observed ground-water nitrate values. However, given the simplifying assumptions of the model, and its lack of testing, care should be exercised in its application. Rather than serving as a predictor of ground-water nitrate concentrations, it can be used as an informative tool for evaluating the relative contribution of various inputs to a "real world" system, and to compare the relative potential for contamination of different sites. Sites with low, medium, and high predicted nitrogen values could be grouped as having low to high susceptibility for significant ground-water pollution. Such information would indicate those aquifers more sensitive to contamination and deserving of more careful study.

OTHER CONSIDERATIONS FOR AQUIFER ASSESSMENT

Present assessment of the ground-water system during the site evaluation process is generally limited to determining depth to the water table. The information presented in the preceeding discussion suggests that if protection of ground-water quality is to be genuinely addressed, it is important to consider the aquifer characteristics discussed in the model. Other variables, whose quantification is difficult, and that were not evaluated in the model, may also play important roles in the net impact of septic system effluent on ground-water quality. These variables are included in Table 2. Internal factors define the ability of the ground-water system to dilute the incoming contaminants to acceptable levels. External factors determine the relative significance of the resultant level of contamination to both public health and environmental quality.

Table 2 suggests that evaluation of potential ground-water impacts should include more than physical measurements or estimates of aquifer characteristics. In particular, it may be appropriate to appraise the relative importance of the receiving aquifer. To illustrate this point, two hypothetical cases will be considered. The first is a proposed high-density development (1 acre lots) that is underlain by a shallow (2 m deep), perched aquifer that has a saturated thickness of only 3 m, and which discharges to a local stream or wetlands area several kilometers

Table 2. Factors to consider when evaluating the impact of septic systems on ground-water quality.

INTERNAL FACTORS

Depth to the water table
Hydraulic conductivity of the aquifer
Ground-water gradient and direction of flow
Aquifer thickness and effective mixing thickness
Background nitrate-nitrogen concentration of ground water
Aquifer geology (e.g. unconsolidated sands, fractured bedrock, etc.)
Potential for denitrification

EXTERNAL FACTORS

Density of septic systems
Present and potential use of aquifer
Impact on down-gradient users
Leakage to (i.e. contamination of) other aquifers
Nutrient status of surface waters in discharge areas
Future land use
Number of pumping wells in the aquifer and volume of water pumped
Length of flow path to surface discharge area or downgradient users

away. It is unlikely that such an aquifer would ever serve as a drinking water source. Because the wetlands represent a preferred sink for the nutrients in the discharging ground water, a greater degree of degradation of this shallow aquifer may be acceptable (or even desirable if it decreases the potential for development that may affect other, more highly used aquifers). In such an example, it is possible that ground-water nitrate-N concentrations of even 20 mg/L or more would be unlikely to create health or water quality problems. Likewise, if discharge was to a stream, its dynamic environment might assimilate the added nitrogen load with little adverse effect.

By contrast, consider a similar development proposed above a deeper (10 m), thicker (50 m) aquifer that is heavily used for domestic water by an existing development down-gradient from the proposed one, or that discharges to a nearby lake. The potential for increased risk of drinking water contamination or nutrient enrichment of the lake is substantial. Accordingly, care should be exercised in determining an acceptable housing density, and/or alternative methods of sewage treatment should be investigated. Integrating the above information into the land-use planning process would result in a decision that development in the former area would be more desirable, other factors being equal.

Consideration of the criteria outlined in Table 2 represents a substantially more sophisticated approach to the site evaluation process than commonly employed. However, given the increased concern for ground-water quality, such sophistication is necessary and beneficial to insure that the potential adverse impacts of septic systems are minimized. A detailed field assessment of these factors will not be necessary for every situation, but even in those cases it is helpful to review these factors to obtain a better understanding of the potential impacts. Information of this nature will assist public officials in calculating the risks associated with a site (i.e. weighing the probability of contamination versus the health and environmental consequences of such contamination).

CONCLUSION

Given the limitations in quantitative data available for most aquifers, such assessments are as much art as science, and their quality will reflect the acumen of the appraiser. The foregoing discussion is intended to improve both the reasoning power and scientific skills of the decision-maker. Recognizing the inherent difficulty in attempting to accurately quantify the parameters influencing ground-water quality, a simple model is proposed to assist local authorities in the evaluation process. Further, this discussion indicates that evaluating the ground-water quality impacts of septic systems is a four-step process. 1) The diluting capacity of the aquifer should be evaluated. 2) An estimate of nitrogen loading to the aquifer is needed. 3) The potential for denitrification within the system should be considered. 4) An assessment is needed of the relative importance of the receiving surface and ground waters.

Preservation of ground-water quality is an issue whose time has come. In the coming years, the loading rate of septic system nitrates to shallow aquifers can only increase, and with it, the potential for serious ground-water contamination. As with other types of environmental issues, the key to effective management is the early recognition of potential problems, and the development of appropriate strategies to mitigate or eliminate them.

REFERENCES

1. Anderson, J. L., R. B. Grossman, K. A. Healy, and W. Skaggs. 1977. Site testing for the design of septic systems. Proc. of the Second National Home Sewage Treatment Symposium. ASAE Publ. 5-77.
2. Bauman, B.J. 1985. Ground-water impacts of septic systems and their regulation in Montana. Ph.D. thesis, Montana State Univ., Bozeman, MT.
3. Champ, D.R., J. Gullens, and R.E. Jackson. 1979. Oxidation-reduction sequences in ground water flow systems. Can. J. Earth Sci. 16:12-23.
4. Dunn, T., and L. B. Leopold. 1978. Water in environmental planning. W.H. Freeman and Co., San Francisco, CA. 818 pp.
5. Fabryka, J. 1978. Septic systems and groundwater quality: A literature survey. New Castle Co. areawide waste treatment management program, Newark, Delaware. 85 pp.
6. Freeze, R.A., and Cherry, J.A. 1979. Groundwater. Prentice Hall, Englewood Cliffs, N.J. 604 pp.
7. Fried, M.K., K.K. Tanji, and R.M. Van De Pol. 1976. Simplified long-term concept for evaluating leaching of nitrogen from agricultural land. J. Environ. Qual. 5:197-200.
8. Hill, A. R. 1982. Nitrate distribution in the groundwater of the Alliston Region of Ontario, Canada. Ground Water 20:696-702.
9. Lance, J. C. 1972. Nitrogen removal by soil mechanisms. J. Water Poll. Cont. Fed. 44:1352-1361.
10. Mercado, A. 1976. Nitrate and chloride pollution of aquifers: A regional study with the aid of a single-cell model. Water Resour. Res. 12:731-747.
11. National Research Council, Environmental Studies Board. 1978. Nitrates: An environmental assessment. National Academy of Science Publ. Washington, D.C.
12. Perkins, R.J. 1984. Septic tanks, lot size, and pollution of water table aquifers. J. Env. Health 46:298-304
13. Pitt, W.A.J., Jr., H.C. Mattraw, and H. Klein. 1975. Groundwater quality in selected areas serviced by septic tanks, Dade Co., FL. U.S.G.S. Open File Report 75-607. Tallahasee, FL. 82 pp.
14. Pye, V. I., R. Patrick, and J. F. Quarles. 1983. Groundwater contamination in the United States. Univ. of Penn. Press.
15. Reneau, R. B. Jr. 1977. Changes in inorganic nitrogenous compounds from septic tank effluent in a soil with a fluctuating water table. J. Env. Qual. 6:173-178.
16. Spruill, T. R. 1983. Relationship between nitrate concentrations to distance of well screen casing below casing water levels. Water Res. Bull. 19:977-981.
17. Wehrmann, A. 1983. Potential nitrate contamination of groundwater in the Roscoe area, Winnebago County, IL. Illinois State Water Survey. Champaign, IL.