Evaluating Groundwater Nitrates from On-Lot Septic Systems, a Guidance Model for Land Planning in Pennsylvania

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Abstract

As land development in Pennsylvania continues to expand into rural areas, land owners, planners and regulators struggle to balance development with the need to provide sustainable drinking water quality. In many areas throughout the state, residential subdivisions will use conventional on-lot septic systems and water wells. When lots are clustered into high density subdivisions, nitrates from septic systems, lawn fertilizers or historic land use can concentrate in the shallow groundwater at levels exceeding safe drinking water standards. Once in the groundwater, nitrates are mobile and fairly recalcitrant and tend to persist for long periods of time, sometimes migrating great distances down-gradient from the source. Dilution from infiltrating groundwater recharge is the primary attenuation process for nitrates; so that providing sufficient lot or open space acreage to allow offsetting groundwater recharge for each septic system is essential to maintain water quality. A mass balance model is presented in this paper as a planning tool to estimate the lot size needed to provide sufficient recharge to maintain nitrate concentrations below a desired water quality goal. The model adapts similar methods presented by others with multiple elements of the nitrate-groundwater system in a unique manner that incorporates common input variables, Pennsylvania specific default values, or more complex site specific information so that is can be used by a variety of user abilities.

Introduction

Nitrates in groundwater are becoming a ubiquitous problem, particularly in rural and suburban areas where domestic water supplies are obtained from individual on-lot water supply wells. Nitrates in groundwater can come from natural sources such as soil, bedrock and organic material; however, the overwhelming loading of nitrates originates from anthropogenic sources, particularly agricultural practices and residential septic systems. As residential subdivisions expand into previously undeveloped or agricultural areas, homeowners, developers, planners and township regulators are challenged to balance sustained growth with a safe supply of drinking water. Prevalent concerns are with high density developments or subdivided lots in close proximity to one another that utilize conventional on-lot septic systems and water supply wells.

Of the estimated 11.8 million residents of Pennsylvania (2000 U.S. census), more than one third use on-lot septic systems (PSATS, 1998) and groundwater as their primary source of drinking water (Hamlet, 1995). Drinking water

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containing elevated nitrates has been attributed to several adverse health effects and can be particularly severe or fatal to small infants. The U.S. Environmental Protection Agency (USEPA) currently sets a limit or maximum contaminant level (MCL) under the Safe Drinking Water Act for nitrates in public drinking water supplies of 10 milligrams per liter (mg/L) reported as nitrate-nitrogen (NO_3-N). In order to prevent drinking water from becoming impacted by the various chemicals and pathogens present in septic system effluent, most states, including Pennsylvania, regulate the placement of septic systems and the separation distance between septic systems and drinking water wells. However, nitrates in groundwater are principally reduced through dilution from the natural recharge of infiltrating precipitation, and when multiple sources of nitrates are located in close proximity to one another, nitrate concentrations in groundwater can concentrate to unacceptable levels. Median nitrate concentrations above 10 mg/L NO_3-N have been reported in groundwater beneath unsewered residential subdivisions, with levels found in excess of 130 mg/L NO_3-N (MPCA, 1999; Yates, 1985). Once in groundwater, nitrates attenuate very slowly and can persist for years or decades, and improving the water quality becomes expensive or even physically impossible (Nolan, 1996). Without sufficient dilution, a nitrate groundwater plume can move considerable distances down-gradient from the source. Distances of 300 feet and more have been reported in the literature (MPCA, 1999); far exceeding the typical separation distance between well and septic system. Therefore, when nitrates are going to be introduced into the groundwater, it is best to prevent their adverse impact in the first place.

One approach to ensure that sufficient nitrate dilution occurs on a proposed development is to promote groundwater recharge and dilution using with optimal lot sizes or open space. Most municipalities require minimum lot sizes, which typically range from ½ to 1 acre (Yates, 1985). While these lot sizes are generally adopted in an effort to maintain separation between neighboring septic systems and water wells, they may not provide sufficient acreage to dilute nitrates in groundwater. Smaller lot sizes may be adequate for low density housing areas, but insufficient when development increases or previously installed septic systems age and begin to fail. Several studies sampled groundwater nitrates beneath unsewered residential communities and suggest lot sizes larger than 1 acre may be needed (Brown, 1987; Yates, 1985). The Pennsylvania Department of Environmental Protection (PADEP) reports a lot size of 1.4 acres (PADEP, 1997) is needed for each septic system. These suggested lot sizes are based on empirical studies or statewide generalizations, and actual lot sizes can vary significantly due to a number of local factors.

This paper presents a mass balance model that can be used to predict the land size per septic system needed to achieve a groundwater nitrate concentration goal. The model expands on the mass balance approach of others by incorporating multiple variables and utilizing input values that are specific to Pennsylvania and local areas within the state. The model is designed to provide the flexibility to accept common default values or more complex site specific information. This allows users with a wide range of abilities to apply the model as a planning tool. Several state and local agencies have formally adopted mass balance methods to evaluate future development and septic system density including New Jersey, Massachusetts and Wyoming.

Nitrogen and Septic Systems

Nitrogen is an essential element in the environment and exists in many forms (Canter, 1997). The interactions of nitrogen between the atmosphere, soil and groundwater can be represented by the nitrogen cycle shown on Figure 1. Plants assimilate nitrogen from the atmosphere and animals consume the plants. Animal waste, fixation by certain plant species and decay of plant and animal material can all contribute to nitrogen in the soil. Human releases of nitrogen into the environment occur through industrial process, agricultural practices, fertilizer applications, and sewage treatment systems such as on-lot septic systems.

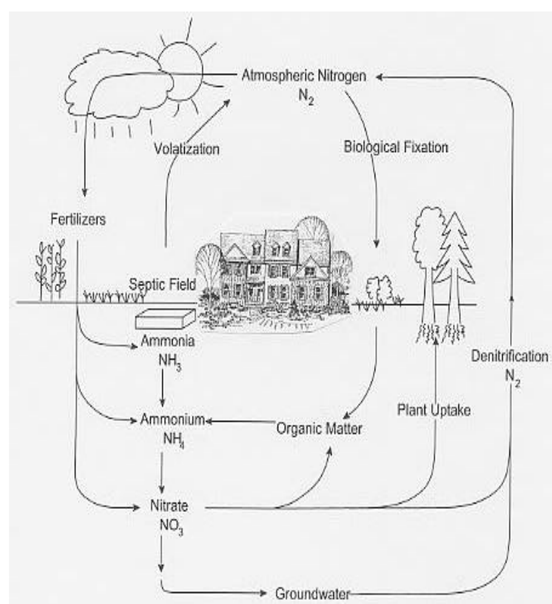

Figure 1. Nitrogen cycle (adapted from Alliance for the Chesapeake Bay fact sheet)

Human waste contains urea and organic nitrogen which is converted to ammonia and ammonium in the septic tank. Conventional septic tank systems are a cost effective means of treating the solids, organic pollutants and microorganisms in waste water, however they are not specifically designed to remove nitrogen (Mooers, 1996; MPCA, 1999) and it tends to pass directly into the disposal field with the liquid effluent. In the disposal field, the ammonia nitrogen in the waste water is quickly converted sequentially to nitrite (NO_2^-) and then nitrate (NO_3^-) through biological aerobic nitrification processes. Nitrate is highly soluble and is a negatively charged ion, and since soil particles are negatively charged there is very little adsorption and nitrate is easily transported through the soil horizon by the infiltrating waste water and rain water. While initially certain soils such as clays may have some capacity to retain nitrates, studies have found that this adsorptive capacity is lost in as little as two years after a septic system is installed (Rogers 1988). Depending on the subsurface materials and separation distances, it may take several years for nitrates to migrate from the septic field or land surface to the water table. Therefore, increases in nitrate concentrations may not be observed in groundwater or well water for many years after the septic system is installed (Rogers, 1988). Once in groundwater, nitrates continue to move unimpeded, generally migrating concurrently in the direction and velocity of the groundwater itself.

The removal of nitrates through denitrification occurs under anaerobic conditions when biological processes convert nitrate to nitrogen gas (N_2). The nitrogen gas is released back to the atmosphere. Denitrification also requires a source of available organic carbon. These conditions are not generally present in conventional septic systems where unsaturated sandy soils are needed for proper filtration and percolation requirements. The role of denitrification in groundwater is variable and not fully understood. Some studies report that very little denitrification occurs in groundwater while other studies indicate that denitrification may play a significant role (Canter, 1997; Hantzsche, 1992; Lindsey, 1997; MacLeod, 1995; Mooers 1996; Nolan, 2003; NJDEP 2001; Puckett, 2002; Taylor, 1996; Trojan, 1999). Groundwater that is rich in oxygen and lacking in carbon is not likely to have any significant denitrification occurring. On the other hand, deeper anoxic groundwater zones or shallow groundwater entering organic rich riparian buffer zones may have substantial denitrification.

Background Nitrates in Groundwater

Nitrates in the subsurface environment originate from a variety of sources including atmospheric deposition, bedrock, decaying organic material, industrial discharges, agricultural practices and septic systems. A study by the U.S. Geological Survey (Nolan, 2003) identified natural background nitrate concentrations (groundwater free of anthropogenic sources) in the groundwater across the United States to be on the order of 0.1 mg/L. Concluding that most groundwater has been influenced to some degree by human interaction, the "relative background" was found to be around 1 mg/L. Their findings were consistent with other generally considered background nitrate (as NO_3-N) concentrations of 3 mg/L or less (MacLeod, 1995; McFarland, 1996; Tesoriero, 1997).

Factors that may affect an aquifer's susceptibility to nitrates and the concentration of nitrates in groundwater include land-use, climate, topography, groundwater flow, infiltration rates, subsurface biogeochemical conditions, bedrock types, and soil characteristics (Lindsey, 1997; Nolan, 2003). For example, researchers at the University of California (Bailey, 1998) found higher concentrations of nitrates in watersheds dominated by bedrock such as phyllite, slate, and biotite schist, while water in contact with igneous rock did not have unusually high nitrate concentrations. Coarse grained soils can readily transport nitrates to the groundwater, while fine grained soils limit vertical migration. In forested areas, the source of nitrogen is mostly from atmospheric deposition and decomposing vegetation (Lindsey, 1997) and will typically approach background levels. Agricultural areas add nitrates from fertilizer and animal waste, and studies suggest groundwater nitrate concentrations have increased as a result of the increased fertilizer use since the 1960's (Canter, 1997; Puckett, 2002; Macleod 1995; McFarland, 1996). Even if land use is changed from agricultural to residential, residual nitrogen can be left in the soil and take many years to attenuate. Continued residential fertilization may contribute higher concentrations of nitrates per area than previous agricultural practices.

Two USGS studies looked at groundwater nitrate concentrations in the Lower Susquehanna River Basin of Pennsylvania (Lindsey, 1997), and the Piedmont and Coastal Plain Provinces of Maryland (McFarland, 1996), which has similar geology to Pennsylvania. In the Susquehanna River Basin study, highest groundwater nitrate concentrations were found in agricultural areas underlain by carbonate bedrock, followed by crystalline bedrock. Groundwater nitrate-nitrogen concentrations were found to seldom exceed 10 mg/L in urban areas underlain by carbonates, and in forested and agricultural areas underlain by sandstone and shale. In the Piedmont, McFarland (1996) found nitrate concentrations were higher in alluvium and shallow saprolite than in deeper saprolite and less still in schist bedrock. Nitrate concentrations in the Piedmont schist were consistently an order of magnitude or more lower than nitrates found in the overburden sites. Dissolved oxygen was found to be high in the saprolite and low in the schist, and was thought to have possibly been consumed by reactions with iron and manganese. It was concluded that the lower nitrate concentrations in the schist resulted from denitrification processes that could occur with the absence of oxygen.

Temporal variations of groundwater nitrates have also been observed. Lindsey (1997) found nitrate concentrations were highest in the winter and lowest in the summer, with some increases noted in the spring. The explanation for the seasonal pattern included plant growth uptake and other nitrogen losses during the summer growing season and an increase in soil leaching when there was more precipitation in the fall. Wehrmann (1984) found increases in groundwater nitrates related to heavy rainfall events when nitrate nitrogen increases of up to 6 mg/L were reported within a one week period. Taylor (1996) also found nitrate concentrations in well water to increase during wet periods and decrease during dry or drought conditions.

Nitrate Mass Balance Approach

Mathematical modeling is a common tool used to predict future conditions based on known or assumed site conditions. Models range from complex to simple and the results and accuracy can vary greatly. Complex models generally require a greater knowledge of mathematics, computers and site-specific conditions. Simple models typically make underlying assumptions that may result in order of magnitude results. A benefit of more simplistic models is that they can be applied by a wider range of users. Mass balance models have been used increasingly to assess the potential impact to groundwater from septic systems. While being relatively simple, their application has shown good correlation between measured and predicted nitrate concentrations in groundwater (Mooers, 1996).

The general mass balance approach assumes through conservation of mass, that the mass of nitrate entering the groundwater system ($MASS_{Ni}$) equals the mass of nitrate leaving the groundwater system ($MASS_{No}$). Although there may be many other factors, the underlying assumption in the approach used for estimating lot sizes and septic system density is that the nitrates entering the system are primarily reduced by dilution. The comprehensive mass balance would consider all sources of nitrates and water entering and exiting the groundwater system, such as infiltrating rainwater, septic system, background groundwater, fertilizer, animal waste, soil, bedrock, stormwater runoff, groundwater flow, water well withdrawal and denitrification. To simplify the mass balance approach the following assumptions can be made regarding the complex interactions:

- All nitrogen leaving the septic system is completely converted to nitrates and reaches the groundwater.
- Dilution primarily accounts for the reduction in nitrate concentration. The model does not consider dispersion, diffusion or adsorption, and denitrification is generally assumed to be absent.
- Rainwater infiltrating across the entire lot provides groundwater recharge to dilute nitrates from the septic system.
- There is uniform and complete mixing of the septic system waste water with the infiltrating rainwater and up-gradient groundwater.
- Complete mixing between waste water and recharge water occurs within a shallow depth of the water table.

Mass balance models for estimating lot sizes have been developed by Trela (1978), Whermann (1984), Bauman (1985), Tinker (1986), Frimpter (1990), Hantzsche (1993) and others. Variations to the different methods presented by these other researchers consist of how the models consider the source and volume of water entering the system and the source and mass of nitrates entering the system. Of these other models, the Trela and Hantzsche methods are the most simplified by assuming the only source of nitrate comes from the septic system, and that the reduction in nitrate only comes from recharge associated with the septic system and rainwater infiltration. The Frimpter method also assumes recharge from the septic system and infiltration and expands the nitrate loading to include lawn fertilizers and stormwater runoff. The Bauman and Whermann methods consider lateral groundwater flow into the subject property from up-gradient areas. Tinker combined the Whermann method with a nitrogen mass-balance model developed by Cornell University called the BURBS model, in order to consider nitrates from lawn fertilizers. The Hantzsche method incorporates denitrification of the septic effluent while the other methods do not consider denitrification.

The mass balance approach is not intended to accurately predict nitrate concentrations at a particular location down-gradient of the septic system, but provide an average approximation of the long term steady-state conditions. The accuracy and applicability of the model becomes greater when used to predict conditions on a large scale housing development as opposed to an individual lot.

A Nitrate Mass Balance Model for Pennsylvania

For the model presented in this paper, nitrates entering the groundwater system are considered to come from the septic system, lawn fertilizer and background groundwater quality. Nitrates from other sources such as soil, bedrock, rainwater are very small in comparison to the loadings from wastewater and fertilizers, and are assumed to be negligible. In the model, background nitrates entering groundwater through infiltration are represented by nitrates added through lawn fertilizer. The mass of nitrates entering the system, expressed as the volume of each source of nitrate multiplied by the concentration of nitrate in the volume, can then be written as follows:

$$MASS_{Ni} = V_sC_s + V_rC_f + V_gC_g \tag{1}$$

Where
$MASS_{Ni}$ = mass of nitrate-nitrogen entering the system
V_s = volume of septic system effluent (gpd)
C_s = concentration of nitrate-nitrogen in the septic system effluent (mg/L)
V_r = volume of groundwater recharge/infiltration across the site (gpd)
C_f = concentration of nitrate-nitrogen in the fertilizer applied to the site that reaches the groundwater (mg/L)
V_g = volume of the groundwater entering the up-gradient side of the site (gpd)
C_g = concentration of nitrate-nitrogen in groundwater entering up-gradient side of site (mg/L)

Nitrate dilution occurs from water added through the septic system, rainwater infiltration and groundwater entering the site from up-gradient. Diluted nitrates leave the system with groundwater through the down-gradient boundary. Groundwater removed by an on-lot well, that penetrates the aquifer mixing zone, is assumed to be returned to the system via the septic system, although in actuality there will be some loss through consumptive use. The mass balance approach also assumes that there is complete mixing at the water table surface across the entire site and the groundwater entering the water well is presumed to already be diluted and equal to the concentration of nitrate in the groundwater leaving the site. Therefore, it is assumed that there is no nitrate mass removed from the system and no loss in dilution recharge via the on-lot water well. Under specific conditions denitrification may be a factor, and the model assumes it occurs in the groundwater uniformly over the project area. The mass of nitrate leaving the system, therefore, equals the total volume of groundwater multiplied by the concentration of nitrate in the groundwater leaving the site, plus any mass removed through biological denitrification (N_d).

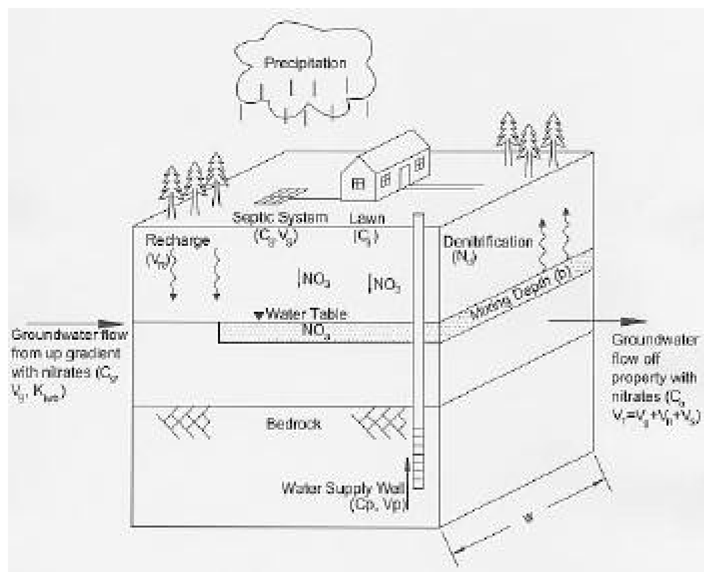

Figure 2. Conceptual mass balance model for nitrates beneath a residential subdivision using an on-lot septic system and water well (adapted from Bauman, 1985).

$$MASS_{No} = (V_s + V_r + V_g)C_o + (V_s + V_r + V_g)C_d \tag{2}$$

Where:
$MASS_{No}$ = mass of nitrate-nitrogen leaving the system
C_o = concentration of nitrate-nitrogen in the groundwater leaving the site (mg/L)
C_d = concentration of nitrate-nitrogen lost through denitrification (mg/L)

The resulting mass balance equation is as follows and conceptualized on Figure 2:

$$V_sC_s + V_rC_f + V_gC_g = (V_s + V_r + V_g)C_o + (V_s + V_r + V_g)C_d \tag{3}$$

Expanding the base model equation to incorporate parameters typically associated with subdivision planning allows for a more user-friendly equation, as provided below.

The volume of septic effluent (Vs) is equal to the number of persons (P) in a household times the discharge (Q) in gallons per day per person ($V_s = PQ$).

Available recharge to the groundwater system (V_r) is the amount of pervious land times the estimated recharge rate.

$$V_r = 74.39 A_L L_p R_i \tag{4}$$

Where:
A_L = minimum lot size (acres)
L_p = pervious surface per lot expressed as fraction
R_i = estimated annual groundwater recharge (inches/year)
74.39 = conversion factor to convert the infiltration from inches per year to gpd/acre

The concentration of nitrate-nitrogen in the groundwater recharge is represented by the amount of nitrate in fertilizer placed on the lawn portion of the lot that reaches the groundwater table with the infiltrating precipitation (C_f). The equation to represent the nitrate-nitrogen concentration in the groundwater recharge is then:

$$C_f = \frac{192.69 M_f F_{nw} L_f}{R_i} \tag{5}$$

Where:
M_f = mass of nitrogen in fertilizer applied to lawn (pounds per 1000 ft^2 per year)
F_{nw} = fraction of nitrogen from fertilizer reaching the water table as nitrate-nitrogen
L_f = fraction of lawn area to total lot size
192.69 = conversion factor to convert lb/1000ft^2/in/yr to mg/L)

Groundwater entering the system at the up-gradient boundary is estimated using the Darcy equation for groundwater flow through a cross-sectional area:

$$V_g = Q = Kiwb \tag{6}$$

Where:
Q = volume of groundwater flowing into the system (gpd)
K = hydraulic conductivity (gpd/ft^2)
i = hydraulic gradient (unitless)
w = width of the mixing zone along the up-gradient aquifer boundary (feet)
b = depth of the nitrate mixing zone within the aquifer (feet)

The model can be used in different variations depending on the needs of the user. The concentration of nitrate-nitrogen leaving the site can be estimated using the equation:

$$C_o = \frac{(V_s C_s + V_r C_f + V_g C_g) - (V_s + V_r + V_g) C_d}{(V_s + V_r + V_g)} \tag{7}$$

To identify the size of a lot (A_L) that will adequately prevent nitrate-nitrogen from exceeding a pre-determined maximum concentration (C_o) in the groundwater; the resulting equation becomes:

$$A_L = \frac{PQ(C_s - C_o - C_d) + Kiwb(C_g - C_o - C_d)}{74.39 L_p R_i (C_o + C_d - C_f)} \tag{8}$$

The mass balance equation presented above is designed to provide a wide range of users' flexibility in its application. For planners who desire a quick evaluation, or where specific information is not known, default input parameters or assumptions can be used to greatly simplify the model. In its simplest form, several basic assumptions can be made such as no denitrification will occur, there will be no nitrate contribution from fertilizer, or there will be no dilution and nitrate contribution from up-gradient groundwater. For those that wish a more detailed evaluation, site specific information can be entered in place of the default input parameters.

Input Parameters for Model

Input parameters for the model can come from commonly accepted values obtained from the literature and Pennsylvania regulatory guidance manuals, such as those proposed as the default values in this paper, or can be determined through site-specific investigations and testing. The input parameters for the model are discussed below. When appropriate, ranges of common input values are provided for comparison and sensitivity analysis.

The maximum concentration of nitrate-nitrogen in the groundwater leaving the site (C_o) dictates the minimum lot size allowed for a proposed development. The USEPA MCL level for nitrates in drinking water is 10 mg/L as NO_3-N. While this should be considered the upper limit for C_o, other states and organizations have adopted more stringent nitrate concentrations, ranging from 1 to 8 mg/L, in order to provide a factor of safety for temporal nitrate fluctuations and also prevent nitrate degradation to down-gradient receptors. The allowable nitrates leaving the site may also vary based on current and proposed land use and should ultimately be decided at a state or local level. A target concentration should be based on the specific goals of the user and regulators and may vary depending on the down-gradient receptor. A value of 6 mg/L is used as the default value in this model.

The PADEP requires a wastewater nitrate-nitrogen effluent concentration (C_s) of 45 mg/L (PADEP, 1997). This value should always be the default value unless it is changed by the PADEP or extensive site-specific data demonstrates otherwise. Effluent concentrations have been reported in the literature ranging from 25 to 100 mg/L NO_3-N and generally average around 35 to 45 mg/L. Effluent concentrations are likely to be the single area that can be controlled in order to meet smaller lot size demands. Alternative treatment systems that consistently produce lower nitrate concentrations leaching to the groundwater could significantly reduce the lot size needed for dilution. Reliable on-lot denitrification systems are not currently available and further product development in this area is needed.

It is assumed that each lot will contain a three-bedroom house with 3.5 persons per household (P). The variable can be changed to reflect any occupancy. For instance, the 2000 U.S. census for Pennsylvania reports an average of 2.5 persons per household.

The default waste water discharge rate (Q) is based on the commonly accepted per capita water usage of 75 gpd per person. This provides a wastewater flow rate of 262.5 gpd/house (PQ). The PADEP's peak design criterion for an on-lot septic system is a 3 bedroom house at 400 gpd plus 100 gpd for each additional bedroom (PA Code Chapter 73, 1997).

Groundwater recharge (R_i) can be estimated from available literature pertaining to counties, watersheds, soil types or local geological regions. Local sub-units would provide more accurate estimates than regional values. Sub-units can be delineated by geologic formation in Physiographic Provinces as shown on Figure 3. Table 1 provides an example of groundwater recharge rates for a sampling of hydrogeological sub-units in central Pennsylvania garnered from Pennsylvania Geological Survey water resource reports (Cwienk, 2003; Taylor, 1984).

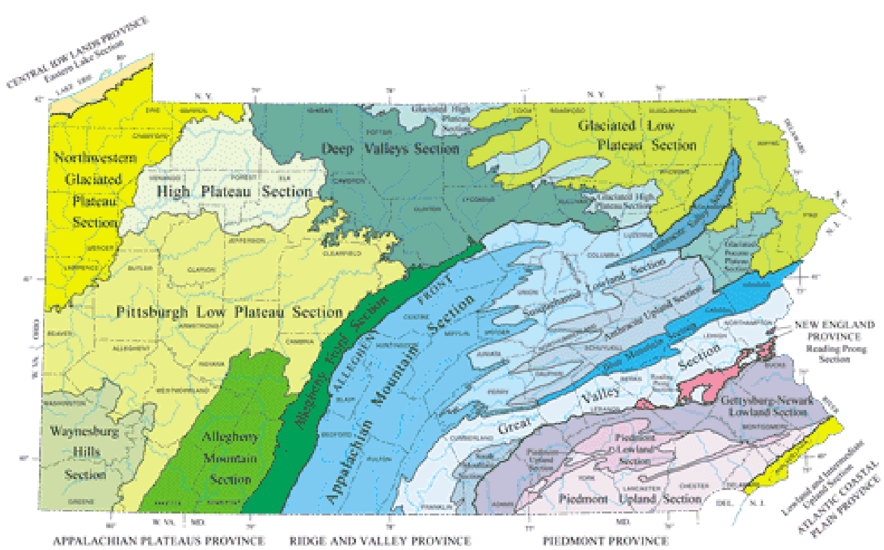

Figure 3. Physiographic Provinces of Pennsylvania provide sub-units to garner groundwater recharge rate data. Pennsylvania Geological Survey, Map 13.

The term for groundwater flow from the up-gradient side of the property (Kiwb) can be zero unless site-specific information is known or approximated for the aquifer. With low hydraulic gradient and conductivity, vertical recharge from waste water and rainfall will tend to accumulate and remain in a layer at the water table, largely unaffected by lateral groundwater flow due to the slow vertical mixing that occurs in horizontal groundwater flow (Hantzsche, 1992). With greater groundwater flow velocities, there will be a large mixing capacity and dilution will be significant. However, when up-gradient background nitrate concentrations are higher than the concentration goal, mixing will increase groundwater nitrate concentration beneath the subdivision and larger lot sizes will be projected. Sensitivity analysis of the above model, not discussed in this paper, and similar findings reported by Bauman (1985), indicate that dilution and nitrate loading from the up-gradient groundwater are not a significant factor in the model unless the aquifer is conductive (K>1 gpd/ft^2) or has a steep gradient (i>0.01). Typical values for K and i are readily available in the literature and can range between 10^{-5} and 10^5 gpd/ft^2 and 0.0001 and 0.1, respectively. The mixing zone thickness (b) should be approximated based on shallow aquifer conditions (e.g., 30 feet for a saprolite water table aquifer (McFarland, 1996)).

Background nitrate concentration in groundwater (C_g) can be estimated from representative well sampling.

The default input parameter for denitrification (C_d) is zero since shallow groundwater conditions are not always favorable for denitrification, rates are not easily measured, and estimates vary by many orders of magnitude ranging from 10^{-4} to 48 mg/L/day (Puckett, 2002). Denitrification rates would likely need to be based on site specific characterization or more extensive research presented in the literature.

Typically the maximum amount of pervious coverage and impervious coverage is regulated by individual townships or is estimated during the preliminary civil engineering analysis. Hoffman (2001) presented an equation to relate impervious coverage to lot size based on information obtained from the U.S. Department of Agriculture. The amount of permeable land (L_p) is the total lot size (A) multiplied by the percent of pervious area represented by a power series; so that $L_p = A(1-0.179A^{-0.5708})$. Resulting lot sizes predicted by the mass balance model will typically be greater than 1.5 acres so that the percent of impervious coverage will typically be in the range of 10 to 15 percent.

If developments propose stormwater management infiltration systems, the pervious coverage (L_p) term may represent the fraction of infiltration decreased or increased between pre- to post-development conditions. Nitrogen loading from stormwater runoff over impervious surfaces that subsequently infiltrates through pervious areas such as stormwater management basins was found to be more significant than natural loading (Eichner, 1992).

The amount of total nitrogen applied to lawns in fertilizer is reported in the literature ranging from approximately 1 to 5 pounds per 1000ft^2 (Carleton, 1996; Eichner, 1992; Hantzsche, 1993). The default value for the mass of nitrogen applied to a lawn (M_f) is 3 lbs/1000 ft^2/year, which is consistent with other reports.

The fraction of nitrogen from fertilizer that reaches the water table (F_{nw}) as nitrate-nitrogen is reported in the literature ranging from 0.05 to 0.80 (Carleton, 1996; Eichner, 1992; Nelson, (1988; Tinker, 1991). The default value is 0.20, which is consistent with other reports.

Table 1. Groundwater recharge rates for some hydrogeological sub-units in central Pennsylvania. Source for groundwater recharge rates: Taylor & Werkheiser. 1984.

Hydrogeological Sub-Unit	Groundwater Recharge (in/yr)
Conestoga Valley - Metamorphic Rocks	6.52
Conestoga Valley (Eastern) - Carbonate Rocks	14.71
Conestoga Valley (Northern) – Shale	11.14
Conestoga Valley (Western) - Carbonate Rocks	10.72
Great Valley (Eastern) - Carbonate Rocks	15.76
Great Valley (Eastern) - Shale with Graywacke	11.14
Great Valley (Eastern) - Shale without Graywacke	9.25
Great Valley (Western) - Carbonate Rocks	13.45
Great Valley (Western) – Shale	11.14
Piedmont Upland - Metamorphic Rocks	9.88
Triassic Lowland (Eastern) - Sedimentary Rocks	10.72
Triassic Lowland (Western) - Sedimentary Rocks	7.15

Example Use of the Pennsylvania Model

The initial step in evaluating the proposed land use with the mass balance model approach is to identify the local watersheds within the proposed project area. This provides an approximation for the area of land contributing recharge. An example is shown on Figure 4. The recharge area to the aquifer is estimated using the project area only so that the model does not rely on current or future land use of contiguous lands. The groundwater recharge rate is estimated for the hydrogeologic sub-unit in which the project is located. For preliminary evaluation purposes groundwater flow in the shallow aquifer is assumed to reflect topography (see Figure 4). A detailed hydrogeologic characterization, including the installation of shallow groundwater monitoring wells, may be prudent when the preliminary evaluation suggests complicated hydrogeologic conditions or when the output results are ambiguous.

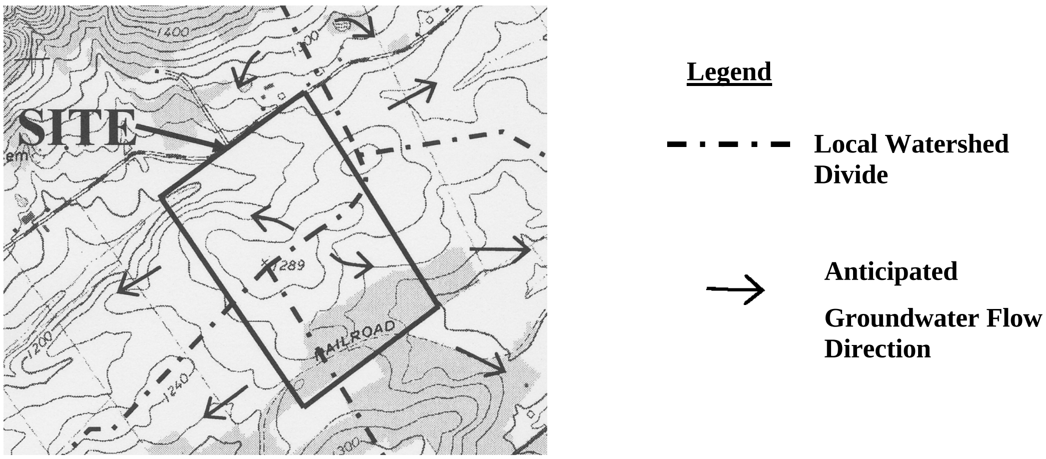

Figure 4. Delineation of local watersheds and anticipated shallow groundwater flow directions on proposed subdivision property (shown on USGS 7.5' topographic quadrangle map).

Finally, the appropriate parameters assembled from the site plans, literature and regulatory guidelines are input into the mass balance model. A computer spreadsheet program makes calculations and sensitivity analysis easier. An example spreadsheet is shown on Figure 5.

The output (A_L) identifies the lot size in acres that should be maintained to prevent nitrate-nitrogen in the groundwater from exceeding the water quality criteria goal. The total acres within the distinct watershed divided by the computed lot size provides the maximum number of lots with septic systems and water supply wells that can be managed within that watershed portion of the subdivision.

Clustering of lots should be considered as an option in order to reduce sprawl and maintain open space areas. However; clustering of lots requires special consideration in order to protect individual homes from potential nitrate impact from surrounding lots.

When existing groundwater nitrate concentrations on the proposed subdivision already exceed the water quality goal, such as 6 or 10 mg/L, or findings from the model preclude the proposed development, alternative waste treatment systems or drinking water sources should be considered.

The example calculations shown on Figure 5 evaluate the number of lots that can be subdivided from the 60 acres within the northern watershed portion of the 130 acre property shown on Figure 3. The output from the model identifies a lot size of approximately 3 acres is needed to prevent groundwater nitrate-nitrogen concentrations from exceeding 6 mg/L on the down-gradient side of the property. Therefore the 60 acres can be expected to handle a maximum of 20 subdivision lots containing conventional on-lot septic systems. Three additional scenarios are offered; the first does not consider nitrogen loading from lawn fertilizers, the second considers dilution from groundwater entering the up-gradient side of the property, and the third considers denitrification. The examples suggest a smaller lot size could be acceptable; however, it may be necessary to obtain site specific data to validate the assumptions.

Discussion

The mass balance model presented in this paper can be used to evaluate the expected nitrate concentrations that will accumulate in the shallow groundwater as the result of residential developments using individual on-lot septic systems. The model is a tool for considering the average, long-term conditions and not specifically for predicting groundwater nitrate concentrations at a single point. Basic assumptions can be used with common default input parameters to provide conservative land use predictions, or site specific data can be implemented to refine the model; thereby allowing a variety of users with different skill levels to adapt the model to the planning process.

Factors sensitive to modeling groundwater nitrates include groundwater recharge rates, septic effluent nitrate concentrations, the leaching potential of fertilizer, and background nitrate conditions from changes in land use. Since a model is only as good as the input, the planning process would benefit from additional research specific to Pennsylvania including:

	A	B	C	D	E	F	G
1	Mass Balance Calculation						
2							
3	=(D7*D8*(D9-D10-D16)+D12*D13*D14*D15*(D11-D10-D16))/(74.39*D17*D18*(D10+D16-D22))						
4							
5		Input Parameter	Units	Without Upgradient Contribution	Without Fertilizer Contribution	With Both Upgradient & Fertilizer Contributions	With Denitrification & Without Upgradient Contribution
6		AL	acres	3.16	2.55	1.95	2.55
7		P	persons	3.5	3.5	3.5	3.5
8		Q	gpd/person	75	75	75	75
9		Cs	mg/L	45	45	45	45
10		Co	mg/L	6	6	6	6
11		Cg	mg/L	3	3	3	3
12		K	gpd/ft2	0	0	10	0
13		i	unitless	0.01	0.01	0.01	0.01
14		w	feet	1300	1300	1300	1300
15		b	feet	10	10	10	10
16		Nd	mg/L/day	0	0	0	1
17		Lp	fraction	0.9	0.9	0.9	0.9
18		Ri	inches	10	10	10	10
19		Mf	lbs/1000ft2/yr	3	0	3	3
20		Fnw	fraction	0.2	0.2	0.2	0.2
21		Lf	fraction	0.1	0.1	0.1	0.1
22		Cf	mg/L	1.16	0.00	1.16	1.16
23							
24					=192.69*D19*D20*D21/D18		
25							

Figure 5. Mass balance model spreadsheet calculations for example property shown on Figure 4.

- o Refined groundwater recharge rates identified for specific geologic sub-units or individual soil series.
- o Amounts of nitrate-nitrogen in fertilizer reaching the groundwater within a specific sub-unit.
- o Denitrification rates for sub-units and methods to estimate denitrification rates from field sampling data.
- o Much of the current development in the state is occurring on previous farmlands. As a result, shallow groundwater in these developing areas is already impacted with elevated groundwater nitrates above acceptable drinking water levels. As the land use changes from agricultural to residential there will be a new equilibrium reached in groundwater nitrate concentration. Understanding the projected rate of change in nitrate concentrations over time would assist in long term planning for the site's water supply.
- o Studies that calibrate and validate the model with existing data, and compare the model's predictions with observed groundwater nitrate concentrations in the field. Findings from these studies may help prioritize what additional research is important in refining the model.

Along with minimum lot sizes, planning should also consider safe separation distances between septic systems and water wells, and proper well construction. A mass balance model, similar to the one presented in this paper, may be adapted to identify suitable wellhead protection zones around the water supply well. The model would identify a safe separation distance between septic systems and water wells through adequate recharge areas around the water well and the travel time from nitrate sources to a water well. Well construction and depth may be the largest single factor affecting nitrate concentrations in the water supply. Deeper well depths and longer, grouted well casing may decrease nitrate concentrations in the water supply by isolating shallow groundwater that is typically more susceptible to nitrate impact, and by obtaining groundwater from deeper, anoxic or multiple water-bearing zones.

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