Hydrol. Earth Syst. Sci. Discuss., 8, 5701–5732, 2011
www.hydrol-earth-syst-sci-discuss.net/8/5701/2011/
doi:10.5194/hessd-8-5701-2011

Hydrology and
Earth System
Sciences
Discussions

Domestic wells have high probability of pumping septic tank leachate

J. E. Horn[1,*] **and T. Harter**[1]

[1]Department of Land, Air, and Water Resources, University of California,
Davis, CA 95616-8629, USA
[*]now at: Institute of Hydromechanics, Karlsruhe Institute of Technology (KIT),
Kaiserstr. 12, 76131 Karlsruhe, Germany

Received: 16 May 2011 – Accepted: 5 June 2011 – Published: 15 June 2011

Correspondence to: T. Harter (thharter@ucdavis.edu)

Published by Copernicus Publications on behalf of the European Geosciences Union.

Abstract

Onsite wastewater treatment systems such as septic systems are common in rural and semi-rural areas around the world; in the US, about 25–30 % of households are served by a septic system and a private drinking water well. Site-specific conditions and local groundwater flow are often ignored when installing septic systems and wells. Particularly in areas with small lots, thus a high septic system density, these typically shallow wells are prone to contamination by septic system leachate. Typically, mass balance approaches are used to determine a maximum septic system density that would prevent contamination of the aquifer. In this study, we estimate the probability of a well pumping partially septic system leachate. A detailed groundwater and transport model is used to calculate the capture zone of a typical drinking water well. A spatial probability analysis is performed to assess the probability that a capture zone overlaps with a septic system drainfield depending on aquifer properties, lot and drainfield size. We show that a high septic system density poses a high probability of pumping septic system leachate. The hydraulic conductivity of the aquifer has a strong influence on the intersection probability. We conclude that mass balances calculations applied on a regional scale underestimate the contamination risk of individual drinking water wells by septic systems. This is particularly relevant for contaminants released at high concentrations, for substances which experience limited attenuation, and those being harmful even in low concentrations.

1 Introduction

In rural and many suburban areas, septic systems (onsite wastewater treatment systems, OTWS) are the primary method for wastewater disposal. In the United States, about one in four households operate a septic system and almost one-third of new homes are constructed with an OWTS as their wastewater disposal system (US EPA, 2003a; US DC, 2008). Septic systems traditionally include a septic tank linked to a drainfield through which minimally treated wastewater is leached into groundwater (Kaplan, 1991; Woodson, 2003). Surveys indicate that at least 20 % of these systems are malfunctioning; over half of all US septic systems are over 30 yr old (US EPA, 2002b). Old and improperly maintained systems are prone to fail and provide inadequate conditions for the effluent treatment processes including physical filtration, surface adsorption, sedimentation, and inactivation of the contaminants in the soil (Canter, 1997; Charles et al., 2005). Leachate from septic systems has been identified as a major potential source of groundwater contamination from pathogens such as bacteria, viruses, helminths, and protozoa, nutrients (nitrogen, phosphorus), pharmaceutically active compounds (PhACs), endocrine active substances (EAS) and other household chemicals (Perkins, 1984; US EPA, 1998b, 2002a; Gerba and James, 2005; Carroll et al., 2006; Fong et al., 2007; Stanford et al., 2010).

Commonly, residences that are using septic systems also provide their own water from a domestic well located on the same property as the septic system leading to a potential risk of drinking water contamination (DeSimone, 2009; Katz et al., 2011). Water wells in close proximity to septic systems on soils with a very high sand fraction, shallow unconfined aquifers, in karst terrain, or on fractured crystalline rocks are especially vulnerable to contamination by pathogens (Scandura and Sobsey, 1997; DeBorde et al., 1998; Frazier et al., 2002; Miller and Ortiz, 2007; Harden et al., 2008; Humphrey Jr. et al., 2010). Yates et al. (1985) and Yates (1991) pointed out that the most common cause of waterborne disease outbreaks in the US is contamination of well water by septic systems. Over 168 000 viral illnesses and 34 000 bacterial illnesses occur each year due to consumption of improperly treated groundwater used for drinking water purposes, according to US Environmental Protection Agency estimates (US EPA, 2003a).

States and local governments increasingly regulate the design, installation, and maintenance of septic systems. Partially with nearby septic systems in mind, many local and state regulations also address the design of domestic wells including minimum screen depth and surface seal depth. Most regulatory development is recent and did not apply at the time most existing septic systems and domestic wells were constructed. Moreover, regulations are typically developed based on very simple conceptual perceptions of groundwater flow. Site-specific conditions are not accounted for. As Borchardt et al. (2011) demonstrated in a case study, poor understanding of groundwater flow can lead to drinking water contamination, even if newly established septic systems are strictly built according to the regulations. Few quantitative tools are available that would allow regulators, planners, or homeowners to assess the probability of domestic wells pumping septic leachate-impacted groundwater (Wilcox et al., 2010).

One approach to prevent excessive aquifer contamination is to determine a minimal required lot size or a critical maximum septic system density, which is the maximum number of septic systems per area that would not lead to overstraining the soil's purifying and the aquifer's dilution capacity. Many authors such as Schmidt (1972), Pitt (1974, 1975), Konikow et al. (1978), Katz et al. (1980), Duda and Cromartie (1982), Bicki et al. (1985), Yates (1985, 1991), Hantzsche and Finnemore (1992), Nizeyimana et al. (1996), Canter (1997), Whitehead and Geary (2000), Borchardt et al. (2011), and Standley et al. (2008) determined significant correlations of septic system density to contaminant concentrations and disease outbreaks. In US EPA (1998a), the agency specifies a septic system density exceeding 40 systems per square mile (1 system per 16 acres) as at risk of groundwater contamination and considers septic system density to be the most important control of contamination risk from septic systems. Dawes and Goonetilleke (2003) come to similar conclusions as well as Miller (1972), Bauman and Schäfer (1985) and also (Wright, 1975) after having measured and studied excessive nitrate levels in areas with high septic system density. In comparison, rural areas of the central and western United States are typically zoned to a minimum lot size of 0.5–1 acre (~2000–4000 m^2) in agricultural-residential areas and a lot size of 20 acres (~81 000 m^2) in exclusively agricultural areas. In some areas lot sizes are even smaller.

Several approaches have been used to determine a minimum lot size or maximum septic system density, respectively:

- Empirical and statistical field studies to determine the minimum lot size required that have historically not lead to groundwater contamination in a specific region.
- Mass balance computations based on the amount of groundwater needed to sufficiently dilute typical septic leachate loading to levels that meet drinking water requirements (e.g., for nitrate).
- Numerical transport modeling studies that allow for a fully two- or three-dimensional evaluation of subsurface flow and transport conditions.
- Site-specific evaluation and measurements.

Lowe et al. (2003), Bishop et al. (2007) and Lowe et al. (2011), for example, used a groundwater flow model coupled with a mass balance approach to evaluate the risk associated with septic system densities. Trela and Douglas (1978) established a dilution model to determine a maximum septic system density for sandy soils; the model has later been modified by Brown (1980) to calculate the required lot size to prevent nitrate concentrations exceeding $10\,\mathrm{mg\,N\,l^{-1}}$; the model of Viraraghavan (1988) was constructed for the same purpose. The flow and transport model of Konikow et al. (1978) was applied to study the effect of septic system densities.

To prevent contamination, regulators also use minimum requirements for horizontal setback distances between drinking water wells and drainfields, and minimum vertical separation distances between drainfields and the seasonally highest groundwater table. A common setback distance, for example, used by local regulators in California is 30.5 m (100 ft) with a minimum vertical separation distance of 1.5 m–3 m (5 ft–10 ft). Different approaches have been used to determine necessary setback distances including site-specific transport studies, mass balance calculations to ensure adequate dilution, geostatistics, and numerical modeling to determine well capture zones. However, setback distances required by local regulations in the US are sometimes inadequate for soils and aquifers with certain hydraulic properties (Yates and Yates, 1989; Postma et al., 1992; DeBorde et al., 1998; Corbett et al., 2002; Lipp et al., 2001). Kerfoot

(1987) raised the question whether well protection by setback distances is adequate without considering groundwater flow direction and demonstrated the need to consider groundwater velocity and direction in the determination of actual pollution risk. Harmsen et al. (1991b) and Harmsen et al. (1991a) used a three-dimensional groundwater flow and transport modeling approach to implement a sensitivity analysis of horizontal and vertical setback distances as a function of aquifer properties. It showed that the necessary distances are most sensitive to horizontal and vertical hydraulic conductivities and the hydraulic gradient. Horn and Harter (2009) showed that typical setback distances are not adequate for aquifers with certain hydraulic conductivities if the typical depth and gravel packs of domestic wells are taken into account.

Until recently, most studies concerning potential contamination of drinking water by septic systems generally estimated set back distances and maximum septic system densities with respect to coliform bacteria and nitrate. These studies generally did not consider other potential contaminants of septic system leachate. Some contaminants of particular concern, e.g., organic micro-pollutants and viruses are among the most mobile contaminants and are harmful at small concentrations (Gerba, 1984; Heberer, 2002; Osenbrück et al., 2007; Gray, 2008; Stanford et al., 2010). Thus, typical mass-balance approaches have a limited significance regarding these substances. Generally, Kaplan (1991) noted that even if some calculations show that a considered area is not contaminated as a whole this does not necessarily imply that individual wells are unaffected by septic effluent contaminants.

In this study, we use a three-dimensional groundwater flow and transport model (Horn and Harter, 2009) for a probabilistic spatial analysis. We evaluate the probability that domestic wells pump water that originates at least partially from septic leach fields (drainfields). Specifically, we determine the likelihood that the source area (capture zone) of a domestic well overlaps with one or more septic leach field as a function of aquifer hydraulic conductivity, conductivity anisotropy ratio, and septic system densities typical for (semi-)rural areas overlying unconsolidated sedimentary (alluvial) aquifers. We estimate the probability that particles from septic systems, which can be harmful even in low concentrations, contaminate drinking water. We consider a typical domestic well design derived from an extensive data base. The actual groundwater flow direction at the local scale in (semi-)rural areas is strongly influenced by nearby large production wells. It can consequently differ significantly from the regional groundwater flow direction and is therefore often unknown; this fact is taken into account within our probability analysis.

2 Conceptual framework

Domestic well capture zones are obtained using a highly resolved three-dimensional numerical groundwater model. By overlaying capture zones with various home and septic system layouts, we compute the probability of one or more septic drainfield systems intersecting with the capture zone of a domestic well. The well capture zone model is described in more detail in Horn and Harter (2009). Briefly, a high resolution finite difference groundwater flow model is constructed (McDonald and Harbaugh, 1988), centered on a typical domestic drinking water well with a gravel pack. The depth of the domestic well screen is from 48 m to 56 m below the water table with the gravel pack extending upward to 18 m below the water table. This value is typical of many domestic wells in alluvial aquifers. The specific depths correspond to the average screen depth obtained from over 3500 domestic well logs in the Central Valley, California (Burow et al., 2004). To obtain the capture zone, we assume steady-state flow with an average areal recharge (from precipitation, irrigation, lawn irrigation, septic system drainfield) of 0.669 $\mathrm{m\,a^{-1}}$ and a pumping rate of 1234 $\mathrm{m^3\,a^{-1}}$ (1 acre-foot per year, the typical annual consumption of a US single family household).

After computing the flow field for specific horizontal and vertical aquifer hydraulic conductivity, K_h and K_v, and for a specific gravel pack hydraulic conductivity, K_g, we define the capture zone through backward particle tracking from the domestic well (Pollock, 1994). The model does not account for the effects of dispersion or aquifer heterogeneity. The geometry of the capture zones is found to be divided into two parts:

one that represents water entering the well screen directly without vertical transport through the gravel pack, and a narrow, elongated part located towards the well that represents capture by the gravel pack above the screen but below the surface well seal (Horn and Harter, 2009). The size and location of a capture zones for specific K_h, K_v, and K_g are used as input to the following spatial analysis.

3 Probability of capture zone intersection with septic drainfield systems

The probability that a domestic well pumps water partially recharged by a septic system is controlled by the uncertainty of the spatial relationship between well capture zone and septic system drainfields. That uncertainty in turn arises from uncertainty about aquifer properties, well properties, aquifer heterogeneity, the patterns of local groundwater flow, and the lack of information about the location of septic drainfield systems (relative to a domestic well).

Aquifer properties vary widely, but are often approximated from geologic and well drilling information (Burow et al., 2004). Subsurface heterogeneity is difficult to assess and typically leads to increased dispersion. Thus, the capture zone is slightly larger than predicted when heterogeneity is ignored (Kunstmann and Kastens, 2006). For a basic assessment, the effects of smaller-scale heterogeneity can be neglected due to its relatively limited effect. However, we account for different aquifer properties by evaluating the probability that a domestic well capture zone overlaps with at least one drainfield as a function of horizontal hydraulic conductivity, vertical hydraulic conductivity, and well gravel pack hydraulic conductivity.

Regional groundwater flow direction can typically be retrieved from regional water level contour maps generated by state or federal agencies. However, the distance from a domestic well to its capture zone is often on the order a few tens to a few hundreds of meters. At that scale, groundwater flow directions may be highly variable due to the influence of large production wells used for irrigation or municipal water supply, due to the influence of local topography and hydrogeology (including large scale heterogeneity), and due to the local influence of nearby streams. Therefore, it is often difficult if not impossible to determine the exact configuration of the flow field near a domestic well. Lacking detailed information on the local flow field, we use a probabilistic approach that assumes a uniform probability of the flow direction being between 0° and 360° relative to the orientation of the property lots.

For simplicity, we assume that the geometric shape of all lots surrounding a domestic well of interest have the same size and are square-shaped. Each lot in such a hypothetical neighborhood is assumed to operate a septic system with a single drainfield system. Drainfields are assumed to occupy a squared area with a user-defined size. The sides of the drainfield area are considered to be parallel to the property boundary. The location of the drainfield within the property boundaries is often unknown. From a regional perspective, the location of the drainfield system relative to the domestic well varies in near random fashion. We model the variability of the location of the drainfield by randomly varying the location among the individual property lots. We assume that the center of the drainfield system is located at any one location within the property lot with equal probability. However, the center of the drainfield cannot be located within less than half its width from the property boundary because the drainfield boundaries must not extend beyond the property boundary. Further, we assume that the drainfields are separated by a distance of at least 3 m (10 ft) from the boundary. The domestic well is assumed to be in the center of the owners property, furthest away from any possible location of neighboring drainfields.

To determine the probability of capture zone intersection with the septic drainfield, we implement a probabilistic spatial analysis. The variable input parameters are the aquifer and well hydraulic properties, K_h, K_v, and K_g. We assume fixed values for the depth of the domestic well, its screen length, gravel pack extent, average pumping rate, and the average recharge rate to the aquifer. The direction of groundwater flow and the location of individual drainfields are random (uncertain, spatially variable) input parameters to the spatial analysis. The purpose of this analysis is to compute the average probability of at least one drainfield system being located within the capture zone of the well. The following explains the details of the spatial analysis, which we programmed into a MATLAB code (MATLAB® – The MathWorks, Inc.).

4 Probabilistic spatial analysis

We represented the capture zone by a polygon defined as the merged area of two rectangles, one representing the maximum width and length of the main capture zone and one representing the maximum width and length of the elongated portion of the capture zone due to the gravel pack extending upwards from the well screen (Horn and Harter, 2009).

The first step of the spatial analysis is to generate a square grid checkerboard, where each grid cell represents a property lot. The domestic well (in the center of a lot) is located at the origin of the coordinate system. The grid is oriented parallel to the x- and y-axis of the coordinate system. The grid contains the minimum number of lots in both the x- and y-direction such that the length of the grid in either direction is greater or equal to twice the length of the extended capture zone. Due to the axis-symmetric configuration around the well, only the central column and row of the grid and one of its quarters needs to be considered for the spatial analysis. Figure 1 shows the upper left quarter of such a grid.

In the second step of the spatial analysis, the capture zone polygons are extended and the lots are reduced at all sides. By extending the capture zones and reducing the lots graphically, we are able to represent the drainfield as a point (drainfield center) instead of a square (Figs. 1 and 2). To prevent overlap of the drainfield from one lot to a neighbor, we reduce the lot area, A_L, by the distance, d_L, at each side. This distance equals half the square root, l_D, of the drainfield area plus a distance, s_L, of 10 ft (3.05 m) assuming the drainfield is build at least 10 ft (3.05 m) away from a property line:

$$d_L = s_L + l_D \tag{1}$$

The reduced lot area A_{RL} equals:

$$A_{\mathrm{RL}} = (\sqrt{A_{\mathrm{L}}} - d_{\mathrm{L}})^2 \tag{2}$$

The capture zones are extended such that intersection between the original capture zone and a drainfield can be measured as intersection of the extended capture zone with the center point of the drainfield system. Figure 2 illustrates this for the flow direction parallel to the y-axis; in this case, the extension, d_{D}, equals l_{D}. The direction of flow is defined as the angle, α_R, between the flow direction and the y-axis. If α_R is other than 0° and 90°, d_{D} depends on α_R and can be geometrically derived by

$$d_{\mathrm{D}} = \sin(45^\circ + \alpha_R) \cdot r_{\mathrm{D}} \tag{3}$$

where r_{D} equals half the length of the diagonal of the drainfield.

The stochastic spatial analysis is implemented by numerical approximation: of all possible flow directions in this quarter-grid (Fig. 1), we consider a finite number of different directions in user-defined steps of x° from $\alpha_R = 0^\circ$ to $\alpha_R = 90^\circ$. For our analysis, we chose 1° increments (90 directions per quarter of the lot grid). The positions of the vertices of the capture zone polygon are calculated by means of a rotation matrix, D_α:

$$D_\alpha = \begin{pmatrix} \cos(\alpha_R) & -\sin(\alpha_R) \\ \sin(\alpha_R) & \cos(\alpha_R) \end{pmatrix} \tag{4}$$

The transformation is repeated for every polygon vertex at each rotation step. At at each rotation step, a Boolean operation determines the intersection of the capture zone with any one of the individual lots of reduced size A_{RL} (Figs. 1, 2). In MATLAB, this analysis is implemented using the polygon operation bundle "polybool". If an intersection exists, the function provides the vertex coordinates of the intersection area. From these, we compute the size of the intersection area, $A_{\mathrm{I}}(i)$, of the capture zone with the area of lot grid cell i. Given $A_{\mathrm{I}}(i)$, the probability, p_i, that the center of the drainfield system area in lot i is located within the intersection area, is:

$$p_i = \frac{A_{\mathrm{I}}(i)}{A_{\mathrm{RL}}} \tag{5}$$

For all non-overlapping grid cells, $p_i = 0$. We further define the complementary probability, $\bar{p}_i$, as the probability that the capture zone does not intersect with the drainfield system center in that lot:

$$\bar{p}_i = 1 - p_i \tag{6}$$

The probability, $p(\alpha_R)$, that the extended capture zone intersects with at least one drainfield center at a given flow direction, α_R, is the complementary of the probability that the extended capture zone intersects none of the drainfield centers:

$$p(\alpha_R) = 1 - \prod_{i_1}^{n_{\mathrm{L}}} \bar{p}_i \tag{7}$$

where the index i is running from one to the number of lots in the grid, n_{L}. Note that for all non-overlapping lots, $\bar{p}_i = 1$.

Having determined the intersection probability $p(\alpha_R)$ for each discrete flow direction step, and each flow direction being equiprobable (in a regional sense), we determine the total probability of intersection, p_{T}, by taking the expected value of the individual probabilities:

$$p_{\mathrm{T}} = \frac{1}{n_R} \cdot \sum_{r=1}^{n_R} p(\alpha_r) \tag{8}$$

where n_R is the total number of discrete, equiprobable flow directions considered.

For our analysis, we consider a range of lot sizes from 2023 m^2 (1 acre), which is the minimum lot size typically required for properties to have septic systems, to nearly 353 300 m^2, which represents an approximate average density of domestic wells in our exemplary study area (Burow et al., 2004). The following lot sizes were chosen:

- 2023 m^2 ("L1"). Lots, which were established before the current regulations regarding minimum required lot sizes of 1 acre (see below) were introduced, are sometimes smaller than 1 acre; here we assume a hypothetical neighborhood of 0.5 acre lots.
- 4047 m^2 ("L2"). This size equals the minimum allowable lot size of 1 acre for "rural-residential areas" in the study area.
- 35 330 m^2 ("L3"). This area is determined by dividing the example study area by 6510 wells (estimated number of domestic wells) and multiplying the result by 10 to represent areas with an increased population density; this size is equal to 8.7 acre.
- 53 000 m^2 ("L4"). The study area is divided by 4340 wells (number of domestic wells with well logs) and multiplied with 10 to represent areas with an increased population density. The resulting lot size is 13.1 acre.
- 80 937 m^2 ("L5"). This area corresponds to an area of 20 acres and was chosen since many rural regulations use this area as minimum required lot size in areas zoned as "agricultural".
- 353 300 m^2 ("L6"). This size corresponds to the estimated current average density of domestic wells in the example study area and presumably of the drainfield density (one system per well).

For the drainfield area, we use three values, 40 m^2 ("D1"), 70 m^2 ("D2") and 100 m^2 ("D3"). The three values cover a typical range required for the drainfield of a household septic system. Equation (6) is evaluated for all possible combinations of the five lot sizes, the three drainfield areas, several K_h ranging from 1 m d^{-1} to 300 m d^{-1}, hydraulic conductivity anisotropy ratios, K_v:K_h, of 1:2 and 1:5, and K_g values ranging from 50 m d^{-1} to 1000 m d^{-1} and at least as large as K_h.

5 Results

The intersection probability, p_T, varies from 0.6 % for large lots (low septic system densities) and low hydraulic conductivities, K_h, to near 100 % for small lot sizes and high hydraulic conductivities. Figure 3 shows the results for all analyzed lot-drainfield-aquifer-configurations.

Under conditions typical of alluvial aquifers, the intersection probability, p_T, that a domestic well pumps water partially recharged from a septic drainfield system is primarily a function of the lot size and, hence, drainfield system density. The smaller the lot size, the higher the risk of a domestic well partially pumping septic leachate independent of aquifer conditions. For the half acre lots (L1), p_T is well over 50 %, regardless of K_h, anisotropy ratio, K_g, or drainfield system size. At very high aquifer and gravel pack hydraulic conductivities, the intersection probability for L1-lots is nearly 100 %.

For a given lot and drainfield set-up, the highest p_T-variation stems from aquifer hydraulic conductivity, K_h, as Fig. 3 reveals. An increase of K_h causes an enlargement of the capture zone which results in an eightfold increase in p_T. The smallest increase due to K_h is observed for the smallest lots, for which p_T is already very high at small K_h; the largest lot and drainfield sizes are most sensitive to K_h. For all lot sizes, the intersection probability is relatively insensitive to K_h at values less than 10 m d^{-1}. This insensitivity is due to the fact that the capture zone in low permeable aquifers is relatively short and close to the production well, i.e., dominated by the amount of recharge. Because of the proximity to the well, few if any neighboring lots are intersected by the capture zone and p_T is controlled primarily by the ratio of drainfield system size to lot size.

Although the gravel pack hydraulic conductivity, K_g, has less influence on p_T than K_h, higher gravel pack hydraulic conductivities are associated with a higher risk of intersection. Figure 4a shows an example for a small lot (L2) and a small drainfield (D1). In this example, the K_g varies from 0.5 % (low K_g) to over 8 % (high K_g). Considering all configurations, K_g modulates p_T up to 13 % . The highest sensitivity occurs for the smallest lot size (L1), largest drainfield size (D3), and at an anisotropy ratio of 1:5. Generally, the larger the lots are, the lower the sensitivity to the gravel pack hydraulic conductivity, i.e., absolute variations of the intersection probability due to K_g decrease with increasing lot size. For the largest lots (L6) and the smallest drainfield (D1), the probability increase between lowest and highest K_g is only 0.1 %. Here, the variation of the size of the capture zone has only a marginal effect on the intersection probability. Figure 4a is representative for all scenarios: p_T is most sensitive to K_g in the middle of the investigated K_h-range (10 and 30 m d^{-1}), while there is no significant change in p_T for any of the K_g values if the aquifer hydraulic conductivity is either very low (less than 5 m d^{-1}) or very high (100 m d^{-1} or more).

The anisotropy ratio of the aquifer hydraulic conductivity has a relatively small influence on p_T. Figure 4b compares the probabilities resulting from the scenarios with an anisotropy ratio of 1:2 to those with an anisotropy ratio of 1:5; the different K_h-values are indicated. It is obvious that in scenarios resulting in either a low or high p_T, there is practically no effect by the anisotropy ratio. For scenarios resulting in p_T-values between 35 % and 75 % (L1–L3), however, the anisotropy ratio of 1:5 often causes a slight increase in the intersection probability, but not exceeding 5 %. The higher K_g relative to K_h, the larger is this difference. In these cases of a large K_g and a K_h not higher than 30 m d^{-1}, the higher anisotropy ratio (1:5) leads to a particularly larger elongation part of the capture zone increasing the risk of intersection with a drainfield. For smaller differences between K_g and K_h, the effect by the anisotropy ratio is less pronounced or even reverses and leads to marginally lower probabilities for the scenarios with an anisotropy ratio of 1:5. K_h-values of 1 m d^{-1} principally result in slightly lower probabilities for an anisotropy ratio of 1:5. For this very small hydraulic conductivity, the elongation part of the capture zone, which is mainly affected by the anisotropy ratio, does not exist.

Drainfield system size has a greater influence on p_T than the anisotropy ratio (within the parameter ranges chosen): larger drainfield system size results in an increased intersection probability. For 100 m^2 drainfield systems, the intersection probability is up

Title Page
Abstract Introduction
Conclusions References
Tables Figures
◀| |▶
◀ ▶
Back Close
Full Screen / Esc
Printer-friendly Version
Interactive Discussion

to 15 % higher than for 40 m^2 systems. Drainfield system size is most influential when the lot size is small. In case of the largest lots (much larger than 100 000 m^2), however, the drainfield system size is so small compared to the lot that is has negligible influence on the intersection probability resulting in probability variations of less than 1 %.

6 Discussion and conclusions

The probabilistic spatial analysis demonstrates that domestic wells have a wide range of intersection probabilities, controlled primarily by lot density and aquifer hydraulic conductivity. The results can be grouped into three general categories driven by lot size: lot size and spatial septic system densities, respectively, of approximately one septic system per 0.5–5 acre (~2000–20 000 m^2) yield a very high probability of intersection with at least 3 in 10 domestic wells pumping water that is partially recharged from septic systems. For septic system densities on the order of one system per 5–20 acres (~40 000–100 000 m^2), p_T is on the order of 5–10 % for medium hydraulic conductivities. When septic system density is very low (~one per 400 000 m^2), the probability of intersection decreases to approximately 1 one in 100 domestic wells pumping some septic system leachate (p_T is on the order of 1 %). Hydraulic conductivity weighs into this classification to some degree, increasing the probability of intersection for highly permeable aquifers and gravel packs around the well, while in aquifers with relatively low hydraulic conductivity, less wells than the above order-of-magnitude estimates tend to be affected. Anisotropy and gravel pack hydraulic conductivity have only limited influence.

It is useful to compare the risk obtained from the statistical spatial analysis against some simple, yet important mass balance based risk indicators: the fraction of areal recharge across all lots that becomes domestic well water, Q_P/Q_R, is defined as the ratio of total domestic pumpage, Q_P, (1234 m^3 lot^{-1}) to total recharge flux, Q_R, from residential lots (varies as a function of lot size) in a (hypothetical) infinite domain.

The ratio Q_P/Q_R is 91.1 %, 45.6 %, 5.2 %, 3.5 %, 2.3 %, and 0.5 % for L1 through L6 lot sizes. Absent any information about septic leach fields and conservatively assuming that all lot recharge poses a high risk of septic contamination, this ratio can be interpreted statistically as a (regional) probability that a well intercepts a septic leachate plume. Despite the very conservative assumption, these ratios indeed provide a good order-of-magnitude approximation of the risk estimates obtained from the probabilistic spatial analysis (compare to Fig. 3). Our spatial analysis demonstrates that simple mass balance ratios are insufficient to capture actual risk: at intermediate and high hydraulic conductivity in the aquifer, which results in long and narrow recharge source areas for each well, risk drastically increases the probability of interception with a septic leach field plume. This increase stems from the fact that Q_P/Q_R does not account for the source area of a well intersecting multiple properties, each with its own septic leach field.

For intermediate and large lots, Q_P/Q_R only provides a lower bound estimate, although it may seem to be an overly conservative estimate. Only for L1, and to a much lesser degree also for L2, the risk computed from the statistical spatial analysis is lower than the value computed from Q_P/Q_R, except at high K_h. In small lots, the geometric shape of the recharge source area significantly lowers the potential risk compared to the simple Q_P/Q_R model.

Some researchers have used mixing model to assess the risk from septic systems. Consider the fraction of septic recharge in the total recharge, Q_S/Q_R, where Q_S is the septic drainfield recharge. For D1, D2, and D3, the recharge from the area immediately underlying the drainfield is 2.2 %, 3.8 %, 5.4 % of domestic water use, respectively. Typical sewage return flow, however, is at least 25 % of total domestic water use (the remainder of the domestic water use ultimately transpires from lawns). A 25 % fraction of total pumpage recharged from the septic leach field (308 $\text{m}^3\,\text{a}^{-1}$) represents 22.8 %, 11.4 %, 1.3 %, 0.9 %, 0.6 %, and 0.1 % of the total lot recharge for L1 through L6 lot sizes. Multiplying these latter fractions, Q_S/Q_R, with the respective lot-size dependent fraction of domestic pumping Q_P/Q_R, yields the (instantaneously) mixed fraction of septic leachate in the domestic well water: 20.8 %, 5.2 %, 0.07 %, 0.03 %, 0.01 %, and 0.001 % for L1 through L6, respectively. Using these mixing model based ratios to evaluate the risk of domestic wells intercepting septic leachate leads to substantial, order-of-magnitude underestimation of the actual risk.

The analysis has several limitations that must be considered. As stated above, the uniform recharge assumption means that recharge from D1, D2, and D3 leach field sizes are only 10 % to 25 % of realistically expected drain field recharge. In particular for L1 and L2, our estimates must therefore be considered to be low. For larger lot sizes, this simplification is less significant. Furthermore, our recharge is based on typical recharge from lawn irrigation, typical drainfield discharge, and recharge from surrounding irrigated agricultural land in relatively surface water-rich, semi-arid climates (Burow et al., 2004) or in humid regions with high precipitation. Elsewhere, recharge rates in areas surrounding a septic drainfield are significantly smaller than $0.7\,\mathrm{m\,a^{-1}}$. Where recharge rates from septic drainfields are nearly an order of magnitude higher than in surrounding areas due to low precipitation, lack of irrigation, or large areas of surface sealing from pavement and buildings (i.e. particularly where lot density is high), p_T-values are likely much higher than estimated above due to the focused contribution of the septic drainfield recharge.

Our analysis also does not consider the effects of aquifer heterogeneity or macrodispersion on the source area. This is a reasonable simplification, since aquifer heterogeneity adds uncertainty, but does not change the recharge contribution from septic leach fields and hence does not change the overall size of the source area, only its actual shape and location (Kunstmann and Kastens, 2006). Since our analysis is already probabilistic with respect to the exact shape and location relative to the lot shape, aquifer heterogeneity would not add to the reported uncertainty. However, spatial variability among lot sizes and shapes, and in septic system densities, not considered here, do affect actual contamination risk.

From a risk management perspective, our results raise significant concern about allowing septic systems to be build on lots smaller than 20 acres (8 ha). Under most aquifer conditions, an assembly of lots of that small size (sub-rural or sub-urban subdivisions, ranchettes) is associated with a potentially significant risk for impacting domestic well water quality in one or several domestic wells. From a public policy perspective, the results imply that regulators and public health agencies may be able to reduce costs of inspection and risk of epidemic by targeting regions with relatively high lot and septic system densities. We caution, however, that the above intersection probabilities are not equal to the risk for domestic well contamination. The contamination risk is determined not only by the intersection probability, but also by contaminant concentration in the septic leachate (source strength), the amount of pumped water that originates from the septic system relative to the total amount of water pumped (dilution), and by the degree to which contaminants released from the drainfield system are inactivated (pathogens) or degraded (chemicals) along their travel path to the well (attenuation).

Nitrate-N is usually considered one of the principal indicators of pollution from septic systems. Hence, most detailed evaluations for maximum septic system densities have been based on groundwater flow model coupled with a mass balance approach for nitrate-N. Generally, the contamination risk is significantly lower for nitrate-N than shown in Fig. 1. The regulatory limit for nitrate-N ($10\,\mathrm{mg\,N\,l^{-1}}$) is approximately one order of magnitude lower than typical septic leachate concentrations. While not attenuated in many unconfined alluvial aquifers (Robertson et al., 1991; Robertson and Cherry, 1992; Pang et al., 2006), dilution of the septic leachate with water from non-septic recharge within the domestic well itself will often reduce nitrate-N levels to below the MCL. Where the lot density is high, or where the ratio Q_P/Q_R is high due to low recharge outside the septic drainfield, the dilution effect is insufficient, as shown by field surveys (Arnade, 1999; Whitehead and Geary, 2000; Drake and Bauder, 2005; Verstraeten et al., 2005). Lowe et al. (2003, 2011) and Bishop et al. (2007), for example, recommended a maximum density of one system per 2.5 to 15 acre that would prevent groundwater contamination with regard to nitrate-N in (semi-)rural areas on unconsolidated basin-fill of fluvial and lacustrine deposits and on alluvial fans. Our study indicates that the lower end of this lot density range is not sufficient to minimize the probability of direct intersection of a well capture zone with a drainfield. In areas predominantly under intensive agricultural use, septic system leachate of course is a very small fraction of the total nitrogen load to groundwater.

The highest risk exists for contaminants frequently released from septic drainfield systems at a high dose relative to drinking water standards and subject to very limited attenuation. Where coarse-textured or fractured soils overly sandy or gravelly aquifer, this includes some pathogenic viruses and bacteria (Scandura and Sobsey, 1997). The MCLG-value (Maximum Contaminant Level Goal) of the US EPA for total coliforms and microbial contaminants like Cryptosporidium is less than one per liter (US EPA, 2003b). Particularly in improperly operated drainfields, pathogens can be released in high concentrations (e.g. Alhajjar et al., 1988; Nicosia et al., 2001; Lowe et al., 2003; Ahmed et al., 2005). Even if only a small fraction of the domestic well water originates from septic leachate, such capture is almost certain to carry pathogens, pathogen indicators or low concentration of organic micropollutants such as PhaCs, personal care products, and endocrine-disrupting chemicals some of which can be transported over long distances (Carrara et al., 2008; Swartz et al., 2006; Godfrey et al., 2007; Rudel et al., 1998; Osenbrück et al., 2007). Effects of long-term exposure and synergistic effects on the human health of various pollutants in low concentrations are not always known; research therefore progressively focuses on these wastewater compounds (Musolff, 2009; Stanford et al., 2010). With respect to these substances, the above values of p_T may be considered a close approximations of the contamination risk (on a per well basis) or of the degree of contamination in a group of wells, e.g., on a county-wide basis.

This study indicates the need for a dual perspective on septic leachate contamination of groundwater. One view is regional, based on the concern that an aquifer may be contaminated by septic leachate from many drainfields. The other perspective is much more local, and concerns the direct contamination of a well with drainfield leachate as investigated in this study. Lot densities that protect regional aquifers (as assessed by the simple mixing model shown above) may not be effective at preventing the second concern, that of direct contamination, as indicated by the risk of well source recharge areas intersecting septic drainfields.

Acknowledgements. We are grateful for the careful and constructive review of this manuscript by Aaron King, University of California, Davis. This work was partially funded by State Water Resources Control Board, California, Agreement # 04-184-555-0. Further funding was provided by the German Academic Exchange Service (DAAD).

References

Ahmed, W., Neller, R., and Katouli, M.: Evidence of septic system failure determined by a bacterial biochemical fingerprinting method, J. Appl. Microbiol., 98, 910–920, doi:10.1111/j.1365-2672.2004.02522.x, 2005. 5720

Alhajjar, B., Stramer, S., Cliver, D., and Harkin, J.: Transport modelling of biological tracers from septic systems, Water Res., 22, 907–915, doi:10.1016/0043-1354(88)90028-0, 1988. 5720

Arnade, L.: Seasonal correlation of well contamination and septic tank distance, Ground Water, 37, 920–923, doi:10.1111/j.1745-6584.1999.tb01191.x, 1999. 5719

Bauman, B. and Schäfer, W.: Estimating ground-water quality impacts from on-site sewage treatment systems, in: Proceedings of the Fourth National Home Sewage Treatment Symposium, American Society of Agricultural Engineers, New Orleans, Louisiana, 285–294 1985. 5704

Bicki, T., Brown, R., Collins, M., Mansell, R., and Rothwell, D.: Impact of on-site sewage disposal systems on surface and ground water quality – Summary of a report to the Florida Department of Health and Rehabilitative Services, Notes in Soil Science No. 17, 1985. 5704

Bishop, C., Wallace, J., and Lowe, M.: Recommended septic tank soil-absorption system densities for the principal valley-fill aquifer, Sanpete Valley, Sanpete County, Utah, Report of Investigation 259, Utah Geological Survey, Salt Lake City, Utah, 2007. 5705, 5719

Borchardt, M., Bradbury, K., Alexander Jr., E., Kolberg, R., Alexander, S., Archer, J., Braatz, L., Forest, B., Green, J., and Spencer, S.: Norovirus outbreak caused by a new septic system in a dolomite aquifer, Ground Water, 49, 85–97, doi:10.1111/j.1745-6584.2010.00686.x, 2011. 5704

Brown, K. W.: An assessment of the impact of septic leachfields, home lawn fertilization, and agricultural activities on groundwater quality, Report to the new jersey pinelands commission, K.W. Drown Assoc., College Station, Texas, 1980. 5705

Burow, K., Shelton, J., Hevesi, J., and Weissmann, G.: Hydrogeologic characterization of the Modesto area, Scientific Investigations Report 2004-5232, US Department of the Interior, US Geological Survey, Reston, Virginia, USA, 2004. 5707, 5708, 5712, 5718

Canter, L.: Nitrates in groundwater, CRC, Chelsea, Michigan, USA, 1997. 5703, 5704

Carrara, C., Ptacek, C., Robertson, W., Blowes, D., Moncur, M., Sverko, E., and Backus, S.: Fate of pharmaceutical and trace organic compounds in three septic system plumes, Ontario, Canada, Environ. Sci. Technol., 42, 2805–2811, doi:10.1021/es070344q, 2008. 5720

Carroll, S., Goonetilleke, A., Thomas, E., Hargreaves, M., Frost, R., and Dawes, L.: Integrated risk framework for onsite wastewater treatment systems, Environmental Manage., 38, 286–303, doi:10.1007/s00267-005-0280-5, 2006. 5703

Charles, K., Ashbolt, N., Roser, D., McGuinness, R., and Deere, D.: Effluent quality from 200 on-site sewage systems: design values for guidelines, Water Sci. Technol., 51, 163–169, 2005. 5703

Corbett, R., Dillon, K., Burnett, W., and Schaefer, G.: The spatial variability of nitrogen and phosphorus concentration in a sand aquifer influenced by onsite sewage treatment and disposal systems: a case study on St. George Island, Florida, Environ. Pollut., 117, 337–345, doi:10.1016/S0269-7491(01)00168-3, 2002. 5705

Dawes, L. and Goonetilleke, A.: An investigation into the role of site and soil characteristics in onsite sewage treatment, Environ. Geol., 44, 467–477, doi:10.1007/s00254-003-0781-6, 2003. 5704

DeBorde, D., Woessner, W., Lauerman, B., and Ball, P.: Virus occurrence and transport in a school septic system and unconfined aquifer, Ground Water, 36, 825–834, doi:10.1111/j.1745-6584.1998.tb02201.x, 1998. 5703, 5705

DeSimone, L.: Quality of Water from Domestic Wells in Principal Aquifers of the United States, 1991–2004, Scientific Investigations Report 2008-5227, US Geological Survey, Reston, Virginia, USA, 2009. 5703

Drake, V. and Bauder, J.: Ground water nitrate-nitrogen trends in relation to urban development, Helena, Montana, 1971–2003, Ground Water Monit. R., 25, 118–130, doi:10.1111/j.1745-6592.2005.0017.x, 2005. 5719

Duda, A. and Cromartie, K.: Coastal pollution from septic tank drainfields, J. Env. Eng. Div.-ASCE, 108, 1265–1279, 1982. 5704

Fong, T., Mansfield, L., Wilson, D., Schwab, D., Molloy, S., and Rose, J.: Massive microbiological groundwater contamination associated with a waterborne outbreak in Lake Erie, South Bass Island, Ohio, Environ. Health Persp., 115, 856, doi:10.1289/ehp.9430, 2007. 5703

Frazier, C., Graham, R., Shouse, P., Yates, M., and Anderson, M.: A field study of water flow and virus transport in weathered granitic bedrock, Vadose Zone J., 1, 113, doi:10.2113/1.1.113, 2002. 5703

Gerba, C.: Applied and theoretical aspects of virus adsorption to surfaces, Adv. Appl. Microbiol., 30, 133–168, doi:10.1016/S0065-2164(08)70054-6, 1984. 5706

Gerba, C. and James, E.: Sources of pathogenic microorganisms and their fate during land application of wastes, J. Environ. Qual., 34, 42–48, 2005. 5703

Godfrey, E., Woessner, W., and Benotti, M.: Pharmaceuticals in On-Site Sewage Effluent and Ground Water, Western Montana, Ground Water, 45, 263–271, doi:10.1111/j.1745-6584.2006.00288.x, 2007. 5720

Gray, N.: Drinking Water Quality: Problems and Solutions, Cambridge University Press, Cambridge, U.K., 2008. 5706

Hantzsche, N. and Finnemore, E.: Predicting Ground-Water Nitrate-Nitrogen Impacts, Ground Water, 30, 490–499, doi:10.1111/j.1745-6584.1992.tb01524.x, 1992. 5704

Harden, H., Roeder, E., Hooks, M., and Chanton, J.: Evaluation of onsite sewage treatment and disposal systems in shallow karst terrain, Water Res., 42, 2585–2597, doi:10.1016/j.watres.2008.01.008, 2008. 5703

Harmsen, E., Converse, J., and Anderson, M.: Application of the Monte Carlo simulation procedure to estimate water-supply well/septic tank-drainfield separation distances in the Central Wisconsin sand plain, J. Contam. Hydrol., 8, 91–109, doi:10.1016/0169-7722(91)90010-X, 1991a. 5706

Harmsen, E., Converse, J., Anderson, M., and Hoopes, J.: A model for evaluating the three-dimensional groundwater dividing pathline between a contaminant source and a partially penetrating water-supply well, J. Contam. Hydrol., 8, 71–90, doi:10.1016/0169-7722(91)90009-P, 1991b. 5706

Heberer, T.: Tracking persistent pharmaceutical residues from municipal sewage to drinking water, J. Hydrol., 266, 175–189, doi:10.1016/S0022-1694(02)00165-8, 2002. 5706

Horn, J. and Harter, T.: Domestic well capture zone and influence of the gravel pack length, Ground Water, 47, 277–286, doi:10.1111/j.1745-6584.2008.00521.x, 2009. 5706, 5708, 5710

Humphrey Jr., C., O'Driscoll, M., and Zarate, M.: Controls on groundwater nitrogen contributions from on-site wastewater systems in coastal North Carolina., Water Sci. Technol., 62, 1448, doi:10.2166/wst.2010.417, 2010. 5703

Kaplan, O.: Septic Systems Handbook, Lewis Publishers, Chelsea, Michigan, USA, 1991. 5703, 5706

Katz, B., Lindner, J., and Ragone, S.: A Comparison of Nitrogen in Shallow Ground Water from Sewered and Unsewered Areas, Nassau County, New York, from 1952 through 1976a, Ground Water, 18, 607–616, doi:10.1111/j.1745-6584.1980.tb03655.x, 1980. 5704

Katz, B., Eberts, S., and Kauffman, L.: Using Cl/Br ratios and other indicators to assess potential impacts on groundwater quality from septic systems: A review and examples from principal aquifers in the United States, J. Hydrol., 397, 151–166, doi:10.1016/j.jhydrol.2010.11.017, 2011. 5703

Kerfoot, W.: Is private well protection adequate when ground water flow is ignored, in: Focus on Eastern Ground Water Issues, National Water Well Association, Burlington, Vermont, 1987. 5705

Konikow, L., Bredehoeft, J., and of the Interior, U. S. D.: Computer model of two-dimensional solute transport and dispersion in ground water, US Govt. Print. Off., Reston, Virginia, 1978. 5704, 5705

Kunstmann, H. and Kastens, M.: Determination of stochastic well head protection zones by direct propagation of uncertainties of particle tracks, J. Hydrol., 323, 215–229, doi:10.1016/j.jhydrol.2005.09.003, 2006. 5708, 5718

Lipp, E., Farrah, S., and Rose, J.: Assessment and impact of microbial fecal pollution and human enteric pathogens in a coastal community, Mar. Pollut. Bull., 42, 286–293, doi:10.1016/S0025-326X(00)00152-1, 2001. 5705

Lowe, M., Wallace, J., and Bishop, C.: Ground-water quality classification and recommended septic tank soil-absorption-system density maps, Cache Valley, Cache County, Utah, Special Study 101, Utah Geological Survey, Salt Lake City, Utah, 2003. 5705, 5719, 5720

Lowe, M., Sabbah, W., and Wallace, J.: Ground-water model based septic-tank density recommendations using a mass balance approach to protect ground-water quality, Cedar Valley, Iron County, Utah, in: Rocky Mountain (63rd Annual) and Cordilleran (107th Annual) Joint

Meeting (18–20 May 2011), vol. 43 of Rocky Mountain (63rd Annual) and Cordilleran (107th Annual) Joint Meeting (18–20 May 2011), p. 74, Geological Society of America, 2011. 5705, 5719

McDonald, M. and Harbaugh, A.: A modular three-dimensional finite-difference ground-water flow model. Techniques of water-resources investigation of the United States Geological Survey, USGS Open-File Report 83-875, US Geological Survey, 1988. 5707

Miller, J.: Nitrate contamination of the water-table aquifer in Delaware, Report of Investigation 20, Delaware Geological Survey, 1972. 5704

Miller, L. and Ortiz, R.: Ground-Water Quality and Potential Effects of Individual Sewage Disposal System Effluent on Ground-Water Quality in Park County, Colorado, 2001–2004, Scientific Investigations Report 2007-5220, US Geological Survey, 2007. 5703

Musolff, A.: Micropollutants: challenges in hydrogeology, Hydrogeol. J., 17, 763–766, doi:10.1007/s10040-009-0438-y, 2009. 5720

Nicosia, L. A., Rose, J. B., Stark, L., and Stewart, M. T.: A field study of virus removal in septic tank drainfields, J. Environ. Qual., 30, 1933, doi:10.2134/jeq2001.1933, 2001. 5720

Nizeyimana, E., Petersen, G., Anderson, M., Evans, B., Hamlett, J., and Baumer, G.: Statewide GIS/census data assessment of nitrogen loadings from septic systems in Pennsylvania, J. Environ. Qual., 25, 346–354, doi:10.2134/jeq1996.00472425002500020020x, 1996. 5704

Osenbrück, K., Gläser, H., Knöller, K., Weise, S., Möder, M., Wennrich, R., Schirmer, M., Reinstorf, F., Busch, W., and Strauch, G.: Sources and transport of selected organic micropollutants in urban groundwater underlying the city of Halle (Saale), Germany, Water Res., 41, 3259–3270, doi:10.1016/j.watres.2007.05.014, 2007. 5706, 5720

Pang, L., Nokes, C., Simunek, J., Kikkert, H., and Hector, R.: Modeling the impact of clustered septic tank systems on groundwater quality, Vadose Zone J., 5, 599, doi:10.2136/vzj2005.0108, 2006. 5719

Perkins, R.: Septic tanks, lot size and pollution of water table aquifers, J. Environ. Health, 46, 298–304, 1984. 5703

Pitt, W.: Effects of septic tank effluent on groundwater quality, Dade County, Florida – An interim report, Open-File Report 74-010, US Geological Survey, Reston, Virginia, USA, 1974. 5704

Pitt, W.: Effects of septic tank effluent on groundwater quality, Dade County, Florida, Open-File Report 75-607, US Geological Survey, Reston, Virginia, USA, 1975. 5704

Pollock, D.: User's guide for MODPATH/MODPATHPLOT, Version 3: A particle tracking post-processing package for MODFLOW, the US Geological Survey finite-difference ground-water flow model, USGS Open-File Report 94-464, US Geological Survey, Reston, Virginia, USA, 1994. 5707

Postma, F., Gold, A., and Loomis, G.: Nutrient and microbial movement from seasonally-used septic systems, J. Environ. Health, 55, 5–5, 1992. 5705

Robertson, W. and Cherry, J.: Hydrogeology of an unconfined sand aquifer and its effect on the behavior of nitrogen from a large-flux septic system, Hydrogeol. J., 0, 32–44, doi:10.1007/PL00010960, 1992. 5719

Robertson, W., Cherry, J., and Sudicky, E.: Ground-Water Contamination from Two Small Septic Systems on Sand Aquifers, Ground Water, 29, 82–92, doi:10.1111/j.1745-6584.1991.tb00500.x, 1991. 5719

Rudel, R., Melly, S., Geno, P., Sun, G., and Brody, J.: Identification of alkylphenols and other estrogenic phenolic compounds in wastewater, septage, and groundwater on Cape Cod, Massachusetts, Environ. Sci. Technol., 32, 861–869, doi:10.1021/es970723r, 1998. 5720

Scandura, J. and Sobsey, M.: Viral and bacterial contamination of groundwater from onsite sewage treatment systems, Water Sci. Technol., 35, 141–146, doi:10.1016/S0273-1223(97)00249-7, 1997. 5703, 5720

Schmidt, K.: Nitrate in Ground Water of the Fresno-Clovis Metropolitan Area, Californiaa, Ground Water, 10, 50–64, doi:10.1111/j.1745-6584.1972.tb02900.x, 1972. 5704

Standley, L., Rudel, R., Swartz, C., Attfield, K., Christian, J., Erickson, M., and Brody, J.: Wastewater-contaminated groundwater as a source of endogenous hormones and pharmaceuticals to surface water ecosystems, Environ. Toxicol. Chem., 27, 2457–2468, doi:10.1897/07-604.1, 2008. 5704

Stanford, B., Amoozegar, A., and Weinberg, H.: The impact of co-contaminants and septic system effluent quality on the transport of estrogens and nonylphenols through soil, Water Res., 44, 1598–1606, doi:10.1016/j.watres.2009.11.011, 2010. 5703, 5706, 5720

Swartz, C., Reddy, S., Benotti, M., Yin, H., Barber, L., Brownawell, B., and Rudel, R.: Steroid estrogens, nonylphenol ethoxylate metabolites, and other wastewater contaminants in groundwater affected by a residential septic system on Cape Cod, MA, Environ. Sci. Technol., 40, 4894–4902, doi:10.1021/es052595+, 2006. 5720

Trela, J. and Douglas, L.: Soils septic systems and cyrrying capacity in the pine barrens, in: Proc. and Papers of First Res. Conf., N.J., Pine Barrens, 37–58, 1978. 5705

US DC: American Housing Survey for the United States: 2007. H150/07, available at: http://www.census.gov/prod/2008pubs/h150-07.pdf, 2008. 5702

US EPA: The Report to Congress: Waste Disposal Practices and Their Effects on Groundwater. EPA 570/9-77-002, 1998a. 5704

US EPA: National Water Quality Inventory: 1996 Report to Congress. EPA 841-R-97-008., 1998b. 5703

US EPA: National Water Quality Inventory: National Water Quality Inventory: 2000 Report. EPA-841-R-02-001, 2002a. 5703

US EPA: Decentralized Wastewater Treatment Systems: A Program Strategy. EPA 832-R-05-002, 2002b. 5703

US EPA: Voluntary national guidelines for management of onsite and clustered (decentralized) wastewater treatment systems. EPA 832-B-03-001., 2003a. 5702, 5703

US EPA: National primary drinking water standards. EPA 816-F-03-016, 2003b. 5720

Verstraeten, I., Fetterman, G., Meyer, M., Bullen, T., and Sebree, S.: Use of tracers and isotopes to evaluate vulnerability of water in domestic wells to septic waste, Ground Water Monit. R., 25, 107–117, doi:10.1111/j.1745-6592.2005.0015.x, 2005. 5719

Viraraghavan, T.: Nitrogen budget and septic tank systems- an appraisal, Water Supply, 6, 89–91, 1988. 5705

Whitehead, J. and Geary, P.: Geotechnical aspects of domestic on-site effluent management systems, Aust. J. Earth Sci., 47, 75–82, doi:10.1046/j.1440-0952.2000.00769.x, 2000. 5704, 5719

Wilcox, J., Gotkowitz, M., Bradbury, K., and Bahr, J.: Using Groundwater Models to Evaluate Strategies for Drinking-Water Protection in Rural Subdivisions, J. Am. Plann. Assoc., 76, 295–304, doi:10.1080/01944361003742403, 2010. 5704

Woodson, R. D.: Water wells and septic systems handbook, Irwin/McGraw Hill, New York, NY, USA, 2003. 5703

Wright, J.: Water quality and solid waste problems in rural new mexico and some solutions, in: Water pollution control in low density areas, edited by: Jewell, W. and Swan, R., University Press of New England, Hanover, New Hampshire, 1975. 5704

Yates, M.: Septic Tank Density and Ground-Water Contamination, Ground Water, 23, 586–591, doi:10.1111/j.1745-6584.1985.tb01506.x, 1985. 5704

Yates, M.: Microorganisms: The drinking water contaminats of the '90's?, Cooperative Extension University of California Environmental Toxicology Newsletter 11(1), University of California, 1991. 5703, 5704

Yates, M. and Yates, S.: Septic Tank Setback Distances: A Way to Minimize Virus Contamination of Drinking Watera, Ground Water, 27, 202–208, doi:10.1111/j.1745-6584.1989.tb00441.x, 1989. 5705

Yates, M., Gerba, C., and Kelley, L.: Virus persistence in groundwater., Appl. Environ. Microb., 49, 778–781, 1985. 5703

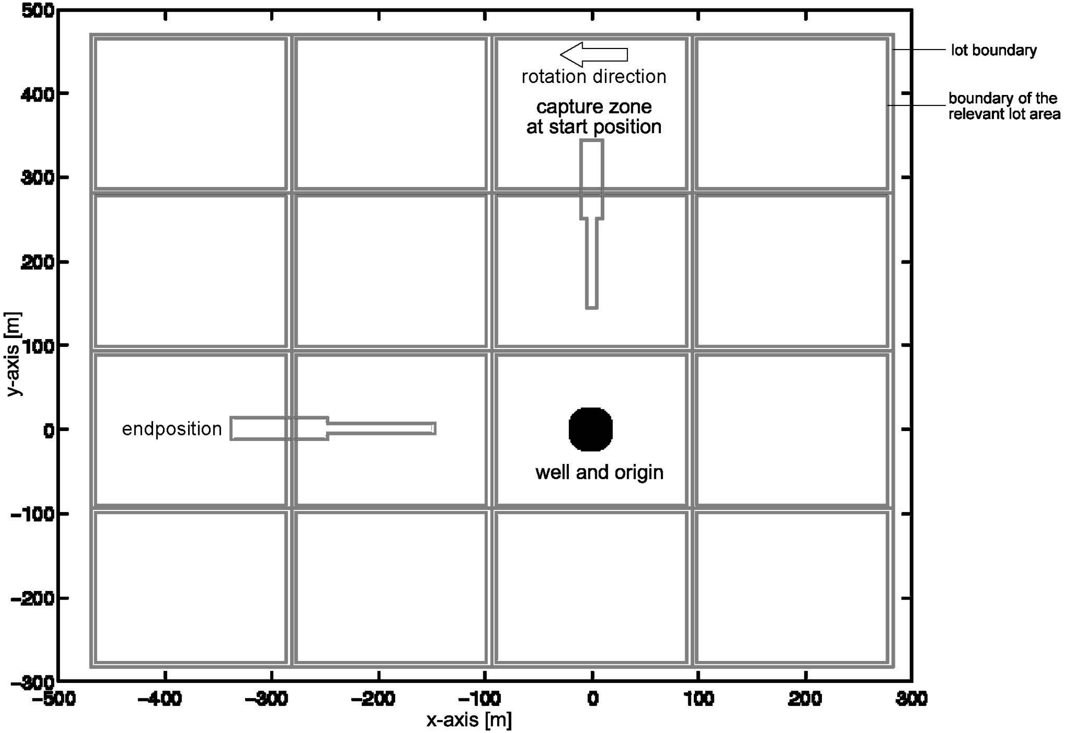

Fig. 1. Upper left part of the lot grid (here for a lot size of 35 330 m^2) with the well as center of the grid and as rotation center for the capture zone. The start position is at $\alpha_R = 0°$, the end position at $\alpha = 90°$. Only a quarter of the lot is modeled for symmetry reasons. The probability of each position of the capture zone is uniform. The capture zone results from a K_h-value of 10 m d^{-1}, an anisotropy ratio, K_v:K_h, of 1:2, and a K_g-value of 1000 m d^{-1}. The "relevant lot area" depends on the size of the drainfield (see text below for further details.)

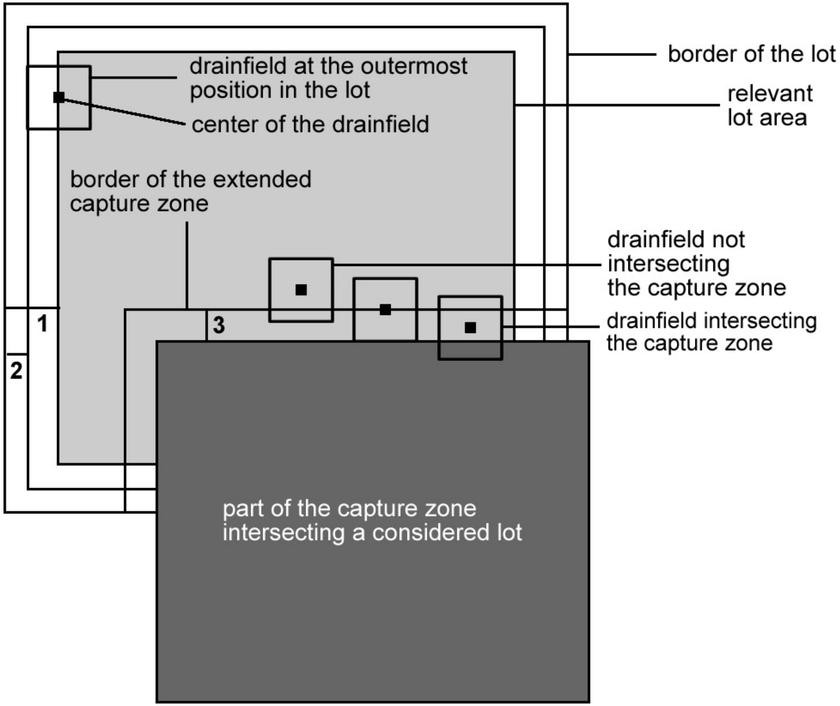

Fig. 2. Capture zone (dark grey) overlapping a lot (grey) with drainfields at different positions. To be able to consider the drainfield as a point, the lot area is reduced by d_L (line 1, Eq. 1) being the sum of half the length of the drainfield, l_D, and a separation distance s_L (line 2); the capture zone is enhanced for the same reason by d_D (line 3, Eq. 3).

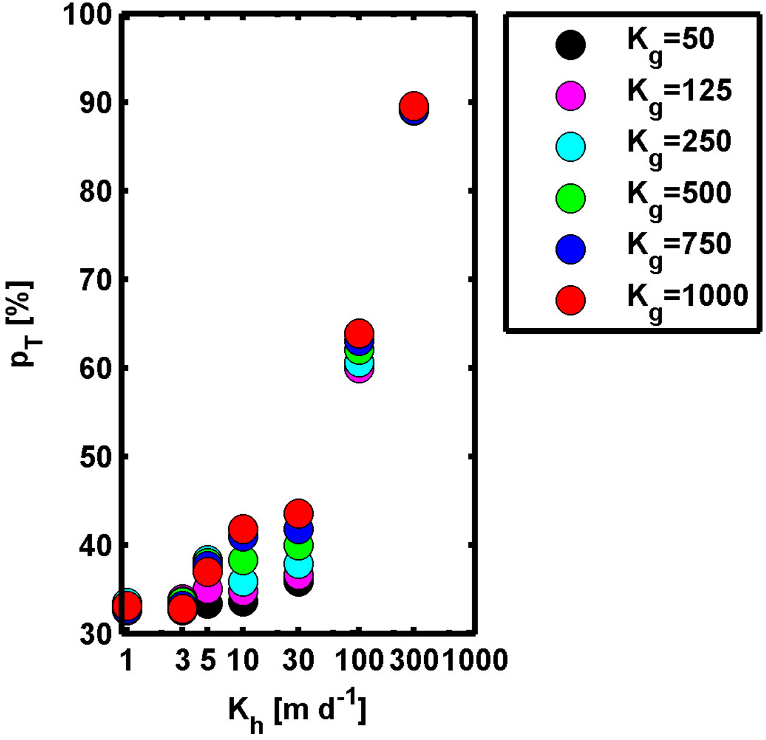

Fig. 3. Resulting intersection probabilities for all analyzed aquifer and septic system configurations. The color of the markers indicates the various lot sizes (L1–L6; see text for further specifications) and septic system densities, respectively. The form of the markers indicates the three drainfield sizes. The range of p_T for a given lot and drainfield size stems from the variation of K_g (see Fig. 4a).

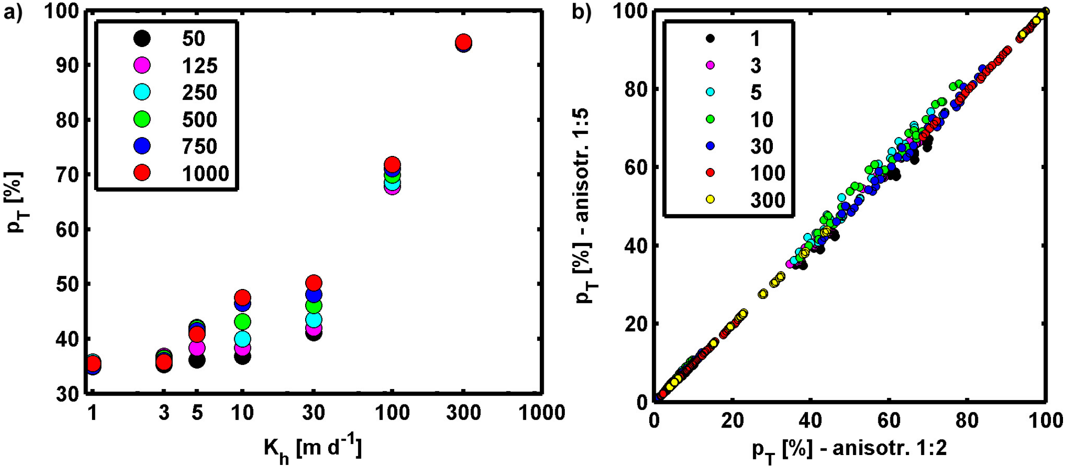

Fig. 4. **(a)** The influence of the hydraulic conductivity of the gravel pack (K_g) on the intersection probability depending on the hydraulic conductivity, K_h. The colors indicate different values of K_g [m d^{-1}]. Example for a lot size of 1 acre (L2), a drainfield size of 40 m^2 (D1), and an anisotropy ratio of 1:5. **(b)** The intersection probabilities for aquifers with an anisotropy ratio of 1:5 vs. those with an anisotropy ratio of 1:2. The colors indicate different values for K_h [m d^{-1}].