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Monitoring and Predictive Modeling of Subdivision Impacts on Groundwater in Wisconsin

Final Report to the Wisconsin Department of Natural Resources

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PROJECT SUMMARY

Title: Monitoring and Predictive Modeling of Subdivision Impacts on Groundwater in Wisconsin

Project ID: 178

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Period of Contract: July 1, 2003 to June 30, 2005

Background/Need: Population growth and urban expansion near many Wisconsin cities has resulted in residential development on agricultural land in rural areas. While developments closer to urban centers typically use city water and sewer services, rural developments usually rely on private wells and septic systems. Septic tank and leach field treatment of wastewater can release contaminants including nitrate, bacteria, viruses, and household chemicals to the subsurface; at the same time, rural residents depend upon groundwater for a clean, reliable drinking-water source. In some areas, zoning ordinances discouraging unsewered residential development – or prohibiting them altogether – have been passed by local officials concerned about groundwater quality. However, there is currently little information available about the long-term impact of unsewered residential use on groundwater quality and quantity.

Objectives: The goals of this project were to collect groundwater quality and water-level data at a developing subdivision near Madison, WI, and use the data to draw preliminary conclusions and construct predictive groundwater flow models to assess subdivision impacts.

Methods: This project required collecting water samples, analyzing water-level fluctuations and rainfall/runoff relationships, and developing computer models in order to identify the effects of subdivision development on groundwater quantity and quality and predict future impacts under different development scenarios.

Results and Discussion:

Nitrate concentrations decreased in shallow groundwater beneath the subdivision site after agricultural loading sources were removed, and a mass balance model showed that nitrogen loading from septic systems may be similar, or even less than during previous agricultural land use. Acetaminophen, a caffeine metabolite, two hormones, and elevated concentrations of nitrate and chloride were detected in septic effluent, and nine of ten septic samples contained estrogenically-active compounds. No target organic compounds or estrogenicity have been detected in groundwater at the site, but the possibility of future detections cannot be ruled out at this point. Flow through the recently-installed drainage basins and a subsurface drain tile responds rapidly to storm events and to spring snow melt. We estimate a 10% decrease in annual recharge across the site due to the diversion of precipitation and snow melt as surface drainage.

Groundwater modeling shows that drinking-water wells may pump at least some water that was recharged within the subdivision boundaries. Well location, well depth, and the vertical conductivity of soil or bedrock at the site play a significant role in determining whether septic plumes will intersect the open borehole of a drinking-water well. Net transfer of water from deeper to shallow aquifers occurs when water pumped at depth is recharged as septic effluent to the vadose zone. Model simulations indicate that switching from shallow individual wells to deep community water-supply wells would have little impact on groundwater levels across the site. From a water-quality standpoint, municipal or community wells may be preferable because these wells can be drilled deeper than is economically feasible for most individual homeowners. Finally, we found that seasonal and year-to-year variations in recharge have much more impact on groundwater levels in this area than any changes caused by residential development.

Conclusions/Implications/ Recommendations:

For this study, we monitored water quality and quantity at a developing rural subdivision site in south-central Wisconsin and constructed groundwater flow models to simulate the effects of subdivision development on groundwater. Data collected thus far do not indicate that the subdivision will significantly affect regional groundwater quality or quantity relative to predevelopment conditions, but groundwater monitoring should continue at the site as more homes are built and septic systems come into use. On a more local scale, septic effluent plumes could eventually intersect the open borehole of nearby water-supply wells within the subdivision. Additional research focusing on processes affecting contaminant mobility – including denitrification, sorption, and microbial degradation – in the unsaturated and saturated zones beneath septic systems would be useful for improved assessment of drinking-water quality in unsewered areas.

Related Publications:

Wilcox, J.D., K.R. Bradbury, C.L. Thomas, and J.M. Bahr, 2005, Assessing background ground water chemistry beneath a new unsewered subdivision: Ground Water, vol. 43, no. 6, *(in press)*.

Key Words: Subdivisions, septic systems, nitrate, pharmaceuticals, hormones, groundwater modeling

Funding: Wisconsin Department of Natural Resources

INTRODUCTION

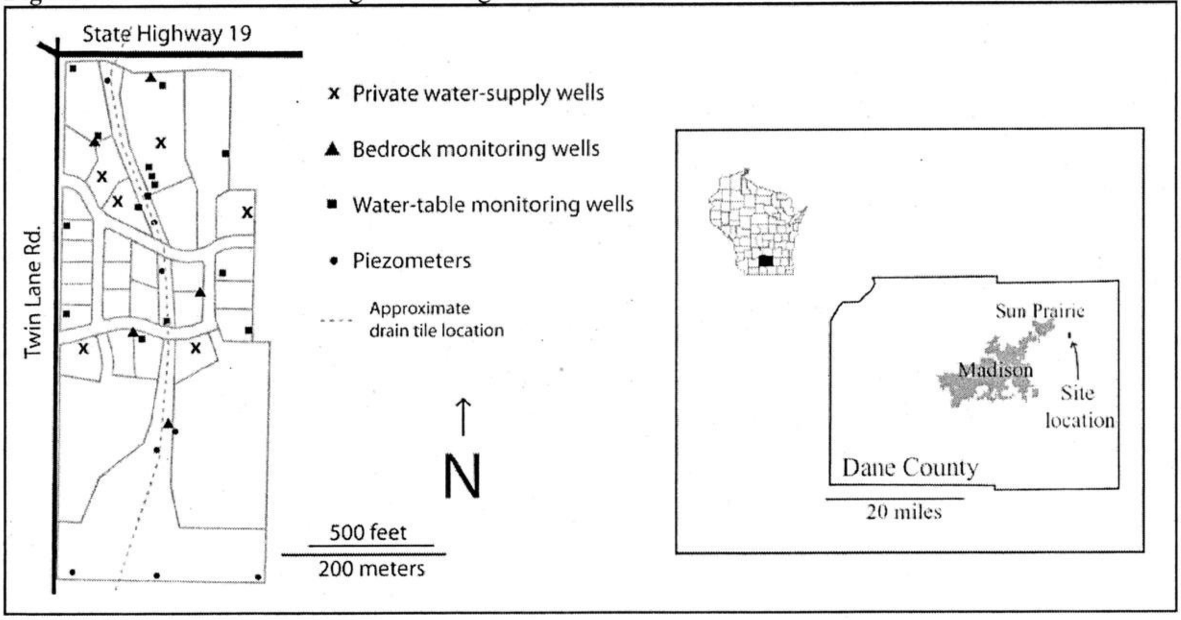
Motivation

Population growth and urban expansion near many Wisconsin cities have resulted in residential development on agricultural land in rural areas. While developments closer to urban centers typically use city water and sewer services, rural developments usually rely on private wells and septic systems. Septic tank and leach field treatment of wastewater can release contaminants including nitrate, bacteria, viruses, and household chemicals to the subsurface; at the same time, rural residents depend upon groundwater for a clean, reliable drinking-water source. In some areas, zoning ordinances discouraging unsewered residential development – or prohibiting them altogether – have been passed by local officials concerned about groundwater quality. However, there is currently little information available about the long-term impact of unsewered residential use on groundwater quality and quantity. The goals of this project were to collect groundwater quality and water-level data at a developing subdivision near Madison, WI, and use the data to draw preliminary conclusions and make predictions about the future impact of unsewered subdivisions on groundwater.

Background information

This project builds upon a previously-funded study, described in Wilcox (2003) and Wilcox et al., (2005), of a new subdivision located 4 miles east of Sun Prairie, WI (Figure 1). The site is characterized by a silt-loam soil overlying glacial till. The till ranges in thickness from 20 to 80 feet and is composed primarily of sand and gravel, although some clay and silt are also present. The till is underlain by the St. Peter sandstone along the western edge of the property. This unit is absent over the rest of the site, where the uppermost bedrock unit consists of interbedded sandy dolomite and dolomitic sandstone.

Figure 1. Site location showing monitoring well network.



The 30-lot, 78-acre subdivision, Savannah Valley, was proposed in summer 2001. When the previously-funded project began, it was anticipated that site improvement and home construction would begin that fall. Delays in site selection and development pushed back new home construction to spring 2003. In the meantime, we installed numerous monitoring wells on the site (locations shown in Figure 1), collected geophysical logs, measured groundwater levels, determined properties of groundwater flow, and collected a full year of water chemistry data prior to any development at the site. Predevelopment monitoring revealed that groundwater chemistry was extremely variable, both spatially and temporally (Wilcox, 2003). As of June 2005, there were only five completed homes with residents and two houses under construction, although several additional lots had been purchased and may be developed soon.

Objectives

Continued monitoring after the first septic system was installed in 2003 was necessary to assess the potential impacts on groundwater. Several previous studies have characterized traditional wastewater-quality parameters including nitrate, chloride, fecal coliform, and BOD from traditional on-site wastewater treatment systems (e.g. Alhajjar, 1990). However, there have been only limited studies of new-technology treatment systems, including aerobic treatment and sand filtration, that have been proposed for use at the Savannah Valley subdivision (Wilcox, 2003). There have also been few studies investigating the release of organic household contaminants such as hormones, pharmaceuticals and personal-care products (HPPCPs) from septic systems. Numerous studies have identified these compounds in municipal wastewater and in streams and lakes receiving wastewater treatment plant effluent (e.g., Halling-Sørenson et al., 1998), and these same contaminants are likely to be found in septic tank effluent. The Savannah Valley subdivision, therefore, provided an ideal field setting for expanded monitoring to include both alternative wastewater treatment systems and emerging organic contaminants.

A key objective of this project was to use computerized groundwater models to predict the impacts of subdivision developments on future groundwater quality and quantity at the Sun Prairie site and at other sites in Dane County and elsewhere in the Midwest. Our goal was to vary model parameters from the Sun Prairie site – including water use, recharge patterns, well design, site layout, housing density, and bedrock type –to assess possible impacts to groundwater under a variety of possible scenarios.

PROCEDURES AND METHODS

This project required collecting water samples, analyzing water-level fluctuations and rainfall/runoff relationships, and developing computer models in order to identify the effects of subdivision development on groundwater quantity and quality and predict future impacts under different development scenarios. The monitoring network at the Savannah Valley subdivision site was expanded during the course of this project to include four additional water-table wells and six soil lysimeters to monitor groundwater and soil pore water beneath two recently-installed Wisconsin mound distribution systems Of the six houses that have been completed as of summer 2005, these were the only two sites where homeowners gave us permission to drill additional wells and where the water table was shallow enough for drilling to be feasible. We collected samples of septic effluent, soil pore water, shallow water from water-table wells, and water from deeper bedrock wells. Major ion concentration, temperature, conductivity, pH, and alkalinity were measured regularly for all sample types, and selected samples were analyzed for HPPCPs. Sampling protocol is described in further detail in Wilcox (2003).

A significant accomplishment of this project was the development of an analytical technique that could be used for this project and for possible future studies in Wisconsin. We experimented with a technique using gas chromatography-mass spectrometry (GC/MS) in the summer of 2004, but were ultimately unsatisfied with the results. We began collaborating with the Wisconsin State Laboratory of Hygiene to develop the solid-phase extraction and liquid chromatography-mass spectrometry (LC-MS/MS) techniques described by Vanderford et al. (2003). Targeted pharmaceuticals, hormones, and other household products in this study were chosen based upon their widespread use or because they had been identified in previous wastewater or surface water studies. Appendix B lists the target compounds, their primary functions, limits of quantitation, and extraction recovery efficiencies. A subclass of hormones, pharmaceuticals, and personal-care products are those compounds considered to be "estrogenically active," meaning they can cause either additive or inhibitory effects when combined with natural estrogens. The e-screen was developed to measure the total estrogenicity of environmental water samples using human breast MCF-7 cells, which grow in response to estrogens or other estrogenically-active compounds (EACs) (Soto et al., 1992). The Wisconsin State Laboratory of Hygiene (Madison, WI) analyzed several groundwater, soil porewater, and septic effluent samples for this study.

When subdivision development began in late 2002, drainage basins were installed in the lowest parts of the landscape to capture stormwater runoff. Site engineers vented the underlying agricultural drain tile (location shown on Figure 1) up to the drainage basins; a direct conduit was created between the surface drainage basins and the subsurface drain tile. We installed a weir at the drain tile outlet and a pressure transducer to collect hourly water-level measurements just upstream from the weir to estimate the volume and timing of water drained from the site.

We developed a computerized groundwater model of the Savannah Valley site to predict the future effects of the subdivision on groundwater quality and quantity, and we varied several parameters to investigate the impacts of different natural conditions or alternative development scenarios. A subdivision-scale model was created from the regional-scale Dane County Hydrologic Model (Krohelski et al., 2000). The refined model domain was chosen so that it included the Savannah Valley subdivision, a southern portion of Deansville Marsh, and several private homes in the area (where well construction reports might provide some information about water levels and subsurface stratigraphy). A variably-spaced grid was used, with row- and column-spacing of 10 feet within the subdivision property boundaries. Layers 1 and 2 in the regional model, representing the unconfined glacial aquifer and the upper Paleozoic bedrock aquifer, respectively, were subdivided into 37 layers to more accurately simulate different bedrock units and variable till thicknesses. Calibration was achieved using the trial and error procedure outlined by Anderson and Woessner (1992) using water level measurements and drain tile flow as calibration targets. The program MODPATH (Pollock, 1989) was used to track particles released from septic systems and pumped by monitoring or drinking water wells. Since particle tracking does not take into account processes such as sorption or degradation of contaminants, this method simulated a "worst-case scenario" transport of wastewater contaminants.

RESULTS AND DISCUSSION

Nitrate and chloride

Nitrate concentrations decreased in shallow groundwater beneath the subdivision site after agricultural loading sources were removed, and chloride concentrations remained high in several wells due to the nearby application of road salt. Prior to subdivision construction, nitrate and chloride concentrations were elevated and varied spatially and temporally over three orders of magnitude due to previous land use (Wilcox, 2003; Wilcox et al., 2005). Figure 2a illustrates a significant decrease in both nitrate concentrations and variability since subdivision construction began – and agricultural activities ceased – in late fall, 2002. Overall, concentrations appear to have stabilized across the site since early 2004. Of the 11 water-table wells drilled initially at the site in December 2001, nitrate concentrations above 10 mg/L-N were detected at least once in 8 wells (73%) and in 32 of 106 (30%) samples. In the six sampling rounds since that time, however, only 2 of the 11 wells (18%) and 3 of 65 samples (5%) have been above the drinking water standard. This indicates that shallow groundwater quality responded rapidly, and improved significantly, following the removal of agricultural inputs at the surface.

Chloride concentrations remained high and variable for wells located adjacent to Twin Lane Rd. and Highway 19 (Figure 2b), probably due to the continued seasonal application of road salt. Concentrations in the remaining wells were consistently below 50 mg/L, and there were no significant increases over time due to septic system effluent or road salt application on the new roads within the subdivision.

Septic effluent samples were collected at three sites within the Savannah Valley subdivision and showed elevated concentrations of nitrogen and chloride. Nitrogen output from septic systems was consistently around 50 mg/L as N, although the form of nitrogen – ammonia or nitrate – depended upon the use of aerobic treatment (Table 1). The nitrogen concentrations found in this study are similar to those found in previous surveys of septic effluent in Wisconsin. In 47 single-pass sand filters, Converse (1999) found median total nitrogen concentrations of 55 mg/L-N before and 38 mg/L-N after passing through a sand filter. Blasing (2003) found median concentrations of 46.8 mg/L-N in septic effluent that had no additional pretreatment, 31.9 mg/L-N in systems using aerobic treatment, and 38.3 mg/L-N in systems using a media filter.

Chloride concentrations were very consistent in each septic system between sampling rounds; however, the significant variation from one system to another (Table 1) is likely due to differences in water softeners or other household activities. Overall, the nitrogen and chloride concentrations in septic effluent are significantly higher than the background chloride or the now-stabilized nitrate concentrations in the shallow aquifer.

Figure 2. Nitrate (a) and chloride (b) concentrations from selected monitoring wells from December 2001 through June 2005. Wells MW02 and MW11 had the most variable nitrate concentrations over the first two years of the project, but more consistent concentrations following subdivision development in late 2002. Wells MW05, MW06, and MW08 have had consistently elevated chloride concentrations, presumably due to road salt from nearby Twin Lane Rd. Both nitrate and chloride concentrations decreased dramatically in the spring of 2002 due to dilution by recharge. This "dilution effect" has been limited since that time, probably due to a dry winter and spring in 2003 and a redistribution of recharge following subdivision development (discussed later).

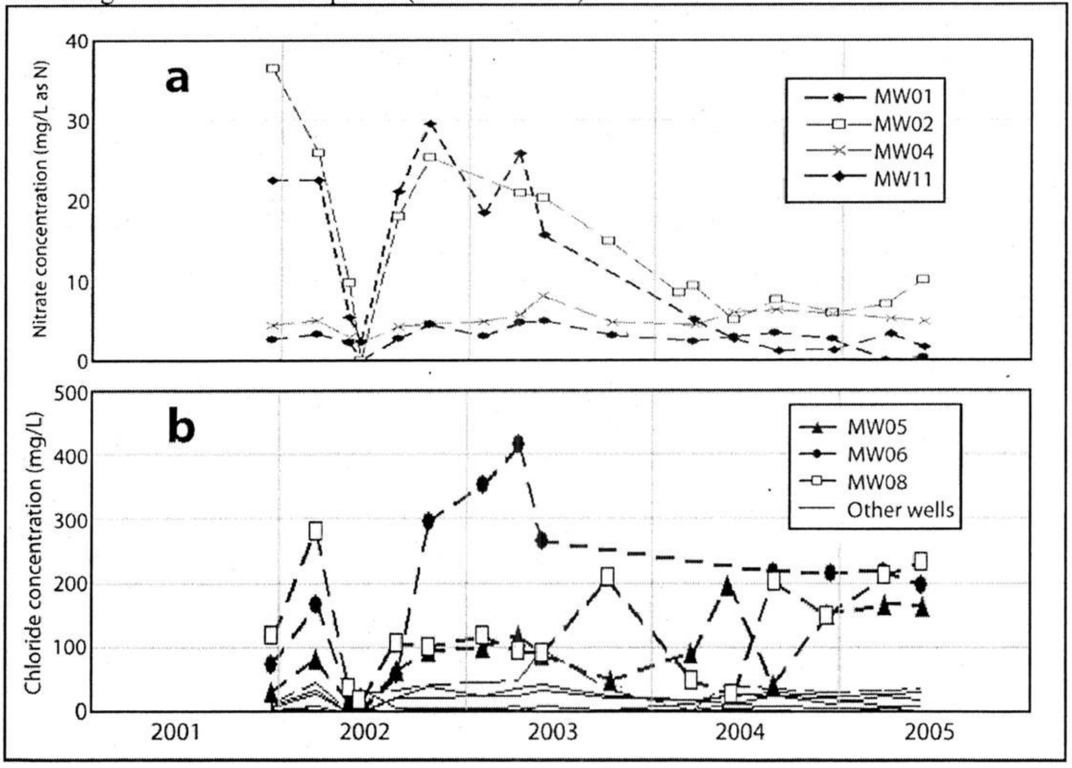


Table 1. Nitrogen and chloride data from septic system effluent.

System name and treatment	Ammonia*	Nitrate-N*	Chloride*
System 1 – with aerobic treatment	0.95 (1)	47.6 ± 1.7 (3)	1109 ± 138 (3)
System 2 – no aerobic treatment	53.5 (1)	0.1 ± 0.17 (3)	458 ± 50 (3)
System 3 – no aerobic treatment	-	< 0.1	45.5 (1)

*( ) indicate the number of samples analyzed, and all concentrations are listed as mg/L

Samples collected from bedrock wells at the Savannah Valley site – including private wells of completed homes – had nitrate concentrations ranging from < 0.1 mg/L-N to 14.2 mg/L-N. The latter value was obtained from a well that was poorly constructed and is not currently connected to any homes (Wilcox, 2003). Excluding this well, mean nitrate values were 5.3 ± 2.2 mg/L-N in 19 samples from 7 wells. Additional samples were collected from the kitchen sinks of two homes utilizing reverse-osmosis treatment systems. In both cases, drinking-water nitrate was below the detection limit of 0.1 mg/L-N, illustrating the effectiveness of in-home water treatement systems.

A survey of 21 private wells in the nearby Drovers Woods subdivision was conducted in September 2004. One well, cased over 200 feet below the ground surface, had a nitrate concentration of 0.7 mg/L-N. The remaining 20 wells, cased 40-60 feet below ground surface, had an average nitrate concentration of 9.2 ± 0.7 mg/L-N. This not only illustrates the importance of casing depth (Wilcox, 2003), but also that nitrate concentrations are laterally consistent in the bedrock aquifer beneath the subdivision.

In an attempt to estimate the nitrogen input from a completed Savannah Valley subdivision to the shallow aquifer, we used a mass balance approach similar to the one described by Tinker (1991). Using estimates of water usage (250 L/person/day), household size (3 people), and nitrogen concentrations measured in septic effluent (50 mg/L-N), the total nitrogen load from septic systems in a completed 30-home subdivision would be 410.6 kg/year. Assuming an average annual recharge rate of 6.5 inches, the average nitrate concentration of water recharging beneath the subdivision would be 7.9 mg/L-N. This calculation assumes uniform distribution of effluent released beneath the site, a valid model only if considering the subdivision as a point source in the context of regional groundwater quality. Lawn fertilizer was not included in the calculation because, to our knowledge, it has not been used at the subdivision site. Actual nitrogen loading rates would be affected by future residents who may elect to fertilize their lawns, advanced treatment systems that reduce septic effluent concentrations below 50 mg/L-N, denitrification, and any changes in precipitation and recharge that affect dilution potential. Despite these uncertainties, the mass balance calculation and data collected from the mature Drovers Woods subdivision suggest that 8-10 mg/L-N is a reasonable estimate of average nitrate concentrations recharging beneath the completed subdivision. Drinking-water concentrations may be higher or lower depending on upgradient agricultural activity and other factors to be discussed below.

Hormones, pharmaceutical, and personal-care products (HPPCPs)

Septic effluent, soil pore water, and groundwater from the two instrumented treatment systems in the subdivision were initially sampled and screened for HPPCPs in summer 2004. Using the GC/MS method, ibuprofen and the hormones estrone and β-estradiol were detected in septic effluent, although quantification was difficult due to matrix effects.

In addition to the sample types described above from the two Savannah Valley systems, septic effluent was collected from 8 other treatment systems in southeast Wisconsin in April 2005. These samples were analyzed using the LC-MS/MS method for HPPCPs and e-screens for estrogenic activity described earlier, with the septic effluent results shown in Table 2. Estrogenic activity was detected in all but one sample. In general the samples with no pretreatment (SF63A, SE5, and SE8) had higher estrogenic activity than those utilizing aerobic systems or sand filters. There was great variability among the samples, with reported estrogenicities ranging from 0.07 to 192.52 ng/L as β-estradiol. The owners of MF02 reported recent trouble with their system; this may or may not explain the anomalously estrogenicity in the effluent. A significant finding is the apparent effectiveness of sand filtration for removing estrogenically-active compounds from wastewater effluent. Sample SF20B was the only septic sample with no detectable estrogenicity, and sample SF63B showed a one hundred-fold decrease in estrogenicity from the inflow concentrations. Analysis of additional samples would help confirm this finding. Only 4 of the target compounds were detected any of the septic effluent samples using the LC-MS/MS method. These included acetaminophen (tylenol), paraxanthine (a metabolite of caffeine), and the hormones estrone and β-estradiol.

Table 2. E-screen and LC-MS/MS results for 10 septic effluent samples. Organic compounds are abbreviated as follows: acetaminophen (Ace), paraxanthine (Par), estrone (Est), and β-estradiol (β-est).

Sample ID	Pretreatment	E-screen (ng/L as E2)	LC-MS/MS screen
BM10	Biomicrobics aerobic system	0.77 ± 0.05	not analyzed
MF02	Multi-Flo aerobic system	192.52 ± 13.10	Ace (48.1 μg/L*), Par (17.8 μg/L*), Est (2.2 ng/L)
MF51	Multi-Flo aerobic system	0.91 ± 0.04	no detections
MF52	Multi-Flo aerobic system	0.19 ± 0.01	Ace (118 ng/L), Par (219 ng/L)
MF60	Multi-Flo aerobic system	0.07 ± 0.01	not analyzed
MF75	Multi-Flo aerobic system	6.06 ± 0.21	not analyzed
SF20B	Sand filter (after treatment)	< LOD	no detections
SF63A	**Sand filter (before treatment)**	**17.60 ± 0.26**	**Ace (15.2 μg/L*), Par (14.3 μg/L*) Est (4.5 ng/L), β-est (2.3 ng/L)**
SF63B	**Sand filter (after treatment)**	**0.27 ± 0.02**	**not analyzed**
SE5	No pretreatment	25.50 ± 2.10	Ace (81.6 μg/L*), Par (32.5 μg/L*)
SE8	No pretreatment	19.57 ± 1.12	Ace (34.9 μg/L*), Par (8.18 μg/L*), Est (3.3 ng/L)

* Indicates concentration above dynamic range of calibration curve. These samples have been diluted and will be re-analyzed.

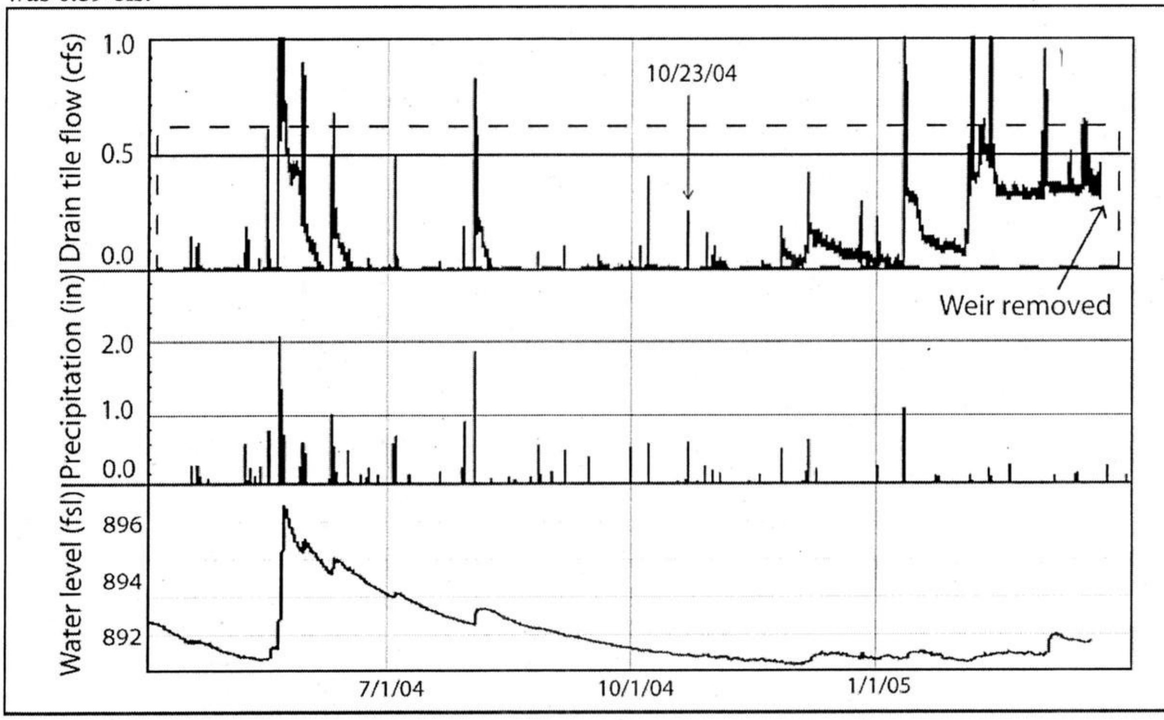
To date, no HPPCPs nor estrogenic activity have been detected in groundwater or soil pore water collected at the Savannah Valley subdivision site. It is possible the septic effluent travels through preferential flow zones in the subsurface that are not being sampled by our monitoring network. Based on the limited data collected from sand filter treatment systems, however, it is more likely that the target organic compounds are being attenuated by the soil.

Recharge and runoff

Manual flow measurements made at the drain tile outlet ranged from 0.0024 to 0.59 cfs. A linear calibration curve (R=0.9778) was used to convert recorded head measurements from the transducer into units of flow. The tile was draining at full capacity when the 0.59 cfs measurement was made; it is possible that higher flow measurements are indicative of water flowing up the transducer access pipe, at which point the calibration curve would lose its linearity. Therefore, we have defined 0.59 cfs as the upper limit for quantifying flow.

A flow hydrograph from April 2004 through April 2005 is shown in Figure 3, alongside precipitation data collected from an on-site rain gauge. Baseflow through the tile was approximately 0.023 cfs (2000 cfd), although the tile was periodically dry during the summer and early fall of 2004. Flow through the tile responded rapidly to storm events and to snow melt in the spring of 2005. Analysis of a single storm showed that 0.73" of rain fell between 6:00 p.m on October 22, 2004 and 8:00 am the following morning, with the greatest hourly intensity of 0.22" between 4:00 and 5:00 am on October 23. The drain tile, which had been dry, began flowing by 6:00 am on October 23, and was dry again by 11pm. During that time, 2928 ft^3 were drained from the tile, or about 0.01" if averaged over the entire 78-acre subdivision site. This indicates that less than 2% of precipitation was diverted from the site via the drainage basins and drain tile during this storm. Monitoring-well hydrographs (bottom of Figure 3) showed no evidence of recharge in the shallow aquifer. Since the storm occurred in October, when antecedent soil moisture was low, much of the remaining 98% of precipitation likely infiltrated to the vadose zone or was lost to evapotranspiration.

Figure 3. Drain tile flow, daily precipitation, and well MW-08 hydrograph. The dotted box in the top graph denotes the zone of quantification; the largest flow measured for the calibration curve was 0.59 cfs.



Previous work has shown that most of the annual recharge at the site occurs during the spring months following snow melt and ground thaw (Wilcox, 2003; Wilcox et al., 2005). The soil reaches infiltration capacity in the spring, resulting in increased runoff and ponding. In 2005, about 0.35 cfs of water was flowing through the tile from February until the end of the record. By late March, several feet of standing water accumulated in drainage basins in the southern half of the subdivision property. We removed the weir at this time so the tile could drain at its full capacity. It is difficult to determine how much of the runoff draining through the tile would have otherwise recharged the groundwater system if the drainage basins and tile were not directly connected; we do not have a long enough record of water levels before or after drainage systems were engineered to account for variations in precipitation, temperature, and the timing of snowmelt. Making numerous assumptions, however, we calculated that drainage through the basins and drain tile during the spring months alone may reduce annual recharge across the site by 10%.

Groundwater Modeling

The final calibrated model of the Savannah Valley subdivision site produced modeled heads in 12 wells that were all within 6 feet of measured heads and modeled drain tile flow that was consistent with estimated baseflow measurements. The calibrated model indicated groundwater flowing regionally from east to west, but shallow groundwater converging toward the drain tile and outlet within the subdivision; this is consistent with observations made by Wilcox (2003). Groundwater conditions following completion of the subdivision were simulated by adding water-supply wells and septic systems to the calibrated model. Twenty-two single-lot wells and four shared wells were added to the model domain. Each well was given an open pumping interval from 60-160 feet below ground surface – similar well construction to other rural wells – and set to pump 200 gallons per household per day. We accounted for septic system discharge by distributing 160 gallons per household per day of additional recharge over three model cells within each lot. The locations for wells and septic recharge cells on sites that have not yet been developed were chosen based on setback requirements and the most likely house location and orientation.

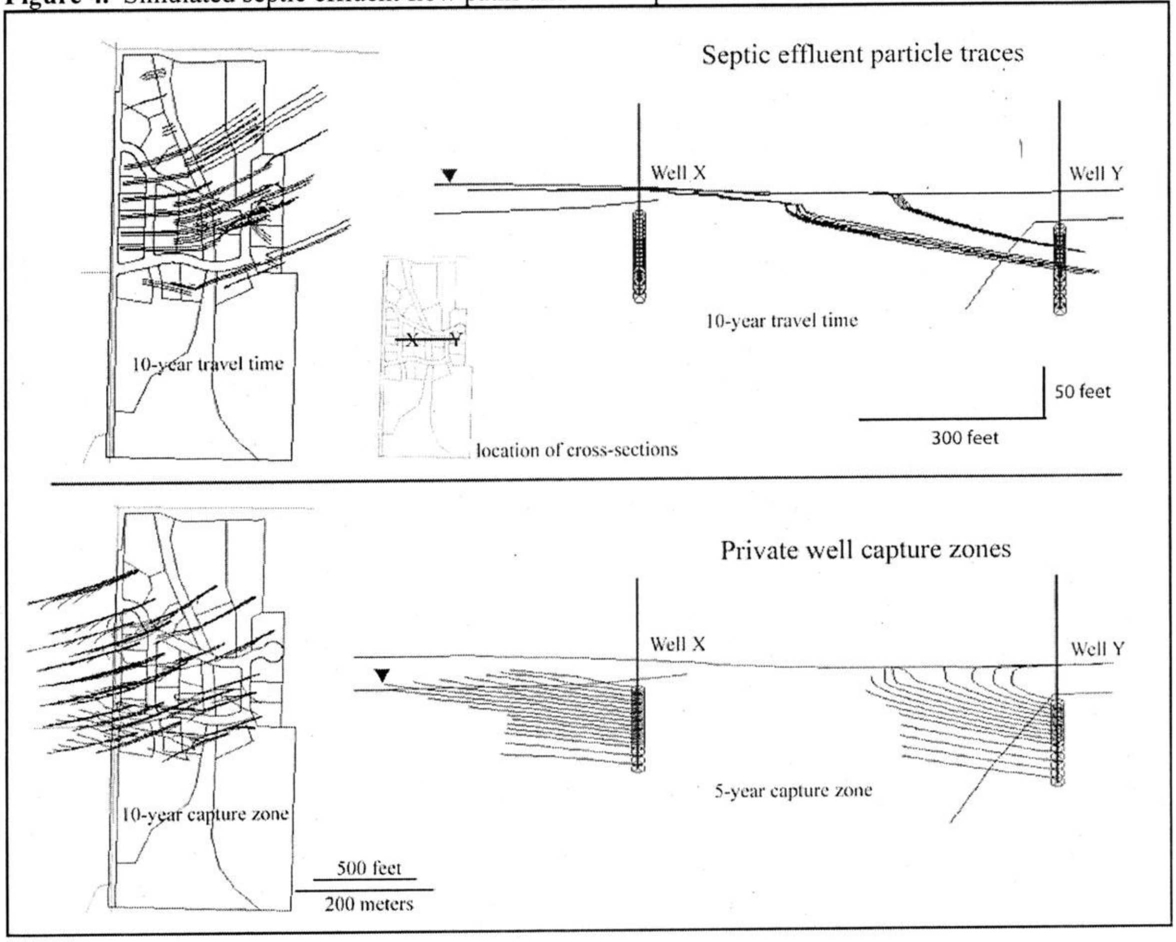
We used particle tracking software – MODPATH (Pollock, 1989) – to identify well capture zones and delineate likely wastewater flow paths from septic systems. Capture zone analysis showed that at least some wells will pump water that recharged within the subdivision boundaries (bottom of Figure 4). Particle travel time from recharge to the pumping well can be as short as two years for water being pumped from 60 feet below land surface (Well Y in Figure 4). Increasing well casing depth to 80 or 120 feet increases the travel time to 5 years and over 10 years, respectively. Sampling data have shown that this deeper, older water has significantly lower nitrate and chloride concentrations, probably due to dilution or degradation processes occurring over a longer period of time. In Figure 4, well X captures much older water than well Y, even though both wells are cased 60 feet below ground surface. Well X is open below the St. Peter Sandstone, which is absent over the eastern part of the site where well Y is located. In order to match the simulated water-table gradient at the site to field conditions during model calibration, it was necessary to assign an extremely low vertical hydraulic conductivity to the St. Peter. There is a known shale layer at the base of this formation, which would impede effluent from reaching the open interval in Well X; there is a much greater vertical component of contaminant flow where the St. Peter is absent (near Well Y).

Particle traces representing wastewater constituents (top of Figure 4) show the impact of well location, well depth, and soil or bedrock type in determining the influence of septic systems on drinking-water quality. Wells on the upgradient side of the site will not be influenced by septic systems; water quality in these wells will depend on land use west of Twin Lane Rd. Well Y in Figure 4 illustrates that septic plumes are more likely to intersect the open intervals of wells on the eastern margin of the site. In these downgradient wells, well depth is a particularly important factor in determining drinking-water quality.

We used groundwater models to investigate the effects on groundwater quantity of using private wells and septic systems versus municipal water and sewer services. We also considered pumping from municipal wells or smaller community wells, but disposing of wastewater through septic systems. Neither the use of individual wells nor a community well significantly impacted water levels across the site, even when we simulated housing densities up to 4 times greater than the approved subdivision. Water level changes in the various simulations were primarily due to increased recharge in the shallow aquifer from septic systems. The greatest water level changes (about 2 feet) occurred where water recharging from septic systems mounded above the shale at the base of the St. Peter. Mounding increased with increased housing density, and reached almost 5 feet above predeveloped conditions in a simulation of high-density development where the shale was present across the whole site. This finding demonstrates the transfer of water from one aquifer to another that could occur as a result of combining deep municipal or community wells with unsewered development.

We simulated the effects of decreased recharge across the site as a result of drainage basin installation and subsequent diversion of surface water that would have previously recharged the shallow aquifer. Varying recharge rates at the subdivision site by 25% had less effect on water levels than increasing or decreasing regional recharge by 5%. All water level changes predicted by these simulations were smaller than those that have been observed since monitoring began in 2001. This suggests that seasonal and year-to-year variations in recharge have much more impact on groundwater levels in this area than any changes caused by residential development. However, small changes in recharge, regionally or at the site scale, did have the effect of changing the water table configuration. This, in turn, affected the direction of septic plume migration predicted by MODPATH, making it very difficult to propose well locations throughout the site that could never be impacted by septic effluent.

Figure 4. Simulated septic effluent flow paths and well capture zones.



CONCLUSIONS AND RECOMMENDATIONS

For this study, we monitored water quality and quantity at a developing rural subdivision site in south-central Wisconsin and developed groundwater flow models to predict the effects of subdivision development on groundwater. Nitrate concentrations decreased in shallow groundwater beneath the subdivision site after agricultural loading sources were removed, and a mass balance model showed that nitrogen loading from septic systems may be similar, or even less than previous agricultural land use. Acetaminophen, a caffeine metabolite, two hormones, and elevated concentrations of nitrate and chloride were detected in septic effluent, and nine of ten septic samples contained estrogenically-active compounds. No target organic compounds or estrogenicity have been detected in groundwater at the site, but the possibility of future detections cannot be ruled out at this point. Groundwater monitoring should continue at the Savannah Valley site as more homes are built and septic systems come into use. Additional research focusing on the processes affecting contaminant mobility – including denitrification, sorption, and microbial degradation – in the unsaturated and saturated zones beneath septic systems would be useful for improved assessment of the effects of unsewered development on groundwater and drinking-water quality.

Groundwater modeling has showed that domestic drinking-water wells may pump at least some water that was recharged within the subdivision boundaries. Well location, well depth, and the vertical conductivity of soil or bedrock at the site play significant roles in determining whether septic plumes will intersect the open borehole of a drinking-water well. Net transfer of water from deeper to shallow aquifers occurs when water pumped at depth is recharged as septic effluent to the vadose zone. Water levels across the site were not significantly different in simulations of municipal, community, and individual wells as the primary water supply. From a water-quality standpoint, municipal or community wells may be preferable because these wells can be drilled deeper than is economically feasible for most individual homeowners. Finally, we found that seasonal and year-to-year variations of recharge have much more impact on groundwater levels in this area than any changes caused by residential development.

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Appendix A. Awards, Publications, Reports, Patents, and Presentations

Publications

Wilcox, J.D., K.R. Bradbury, C.L. Thomas, and J.M. Bahr, 2005, Assessing background ground water chemistry beneath a new unsewered subdivision: Ground Water, vol. 43, no. 6, *(in press).*

Presentations

Wilcox, J.D., K.R. Bradbury, and J.M. Bahr, 2005, Implications for the use of shallow private versus deep municipal wells for water supply at the urban fringe: GSA Annual Meeting Abstracts with Programs, vol. 37.

Wilcox, J. D., K. R. Bradbury, J. M. Bahr and J. A. Pedersen, 2004, Pharmaceuticals and hormones as potential groundwater contaminants from on-site wastewater-treatment systems: Proceedings of the 4th International Conference on Pharmaceuticals and Endocrine Disrupting Chemicals, National Ground Water Association, pp. 81-93 (CD-ROM).

Wilcox, J.D., J.M. Bahr, and K.R. Bradbury, 2004, Viruses, hormones, pharmaceuticals, and other household products as potential groundwater contaminants from on-site wastewater treatment systems: American Water Resources Association - Wisconsin Section 28th Annual Meeting Program and Abstracts, p. 29.

Awards

2004 Best Student Presentation Award, American Water Resources Association – Wisconsin Section, 28th Annual Meeting.

Appendix B. Target compounds for the LC-MS/MS analytical method

Compound	Compound Description	Limit of quantitation (ng/L)	Recovery (out of 100%)
	Hormones		
β-estradiol	Natural hormone	0.25	92.1 ± 1.7
17-α ethynyl estradiol	Synthetic hormone, contraceptive	0.25	90.8 ± 1.4
Estriol	β-estradiol metabolite	0.25	88.7 ± 0.6
Estrone	Natural hormone	0.25	88.6 ± 5.6
	Pharmaceuticals		
Carbamazepine	Antiepileptic	5.0	99.3 ± 4.4
Carisoprodol	Muscle relaxant	1.0	109.5 ± 3.6
Chlorpropamide	Prescription drug for Type II diabetes	1.0	93.6 ± 3.4
Erythromycin	Antibiotic	N/A*	2473 ± 83.3*
Fenofibrate	Cholesterol-lowering drug	5.0	111.2 ± 6.6
Fluoxetine	Antidepressant (Prozac)	1.0	90.1 ± 4.9
Gemfibrozil	Blood-lipid regulator	N/A*	0.0 ± 0.0*
Warfarin	Anticoagulant	5.0	59.1 ± 1.7
	Other		
Acetaminophen	Pain reliever	0.25	60.1 ± 18.5
Atrazine	Herbicide	1.0	78.2 ± 1.0
Bisphenol A	Plasticizer	N/A*	50.8 ± 59.7*
Caffeine	Stimulant	N/A*	11.3 ± 19.5*
Cotinine	Nicotine metabolite	N/A*	87.8 ± 40.1
Paraxanthine	Caffeine metabolite	2.5	75.6 ± 16.6

* indicates compounds requiring futher optimization