

Ground Water Quality under Three Unsewered Subdivisions in Minnesota

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Ground Water and Toxics Monitoring Unit
Environmental Monitoring and Analysis Section
Environmental Outcomes Division
520 Lafayette Road North
St. Paul, Minnesota 55155-4194
(651) 296-6300 or (800) 657-3864

Prepared by

Ground Water Monitoring and Assessment Program (GWMAP)

Table of Contents

Executive Summary

Literature Review

 Nitrate

 Pathogens

 Phosphorus

 Organic Chemicals

 Other Inorganic Chemicals

Materials and Methods

 Description of Study Areas

 Site Characteristics

 Sampling Design and Data Collection

 Statistical Methods

Results and Discussion

 Nitrate

 Bacteria

 Phosphorus

 Other Inorganic Chemicals

Cumulative and Comparative Risk

Summary

References

List of Tables

1. Comparison of chemical concentrations (mg/l) in septic tanks
2. Comparison of chemical concentrations in septic tank effluent, maximum concentrations found in ground water plumes, and background concentrations in ground water. Concentrations are in mg/l.
3. Characteristics of the three study areas.
4. Number of samples collected for different chemicals in each study area.
5. Summary of regression analysis using Eq. [1]. UCL = the 95th percent upper confidence limit of the slope.
6. Comparison of geochemistry in samples having detectable nitrate and having no detectable nitrate.
7. Geochemistry for samples in which nitrate was detected and not detected in temporary well samples screened at the water table.
8. Date summary for total coliform bacteria from temporary well and domestic well samples. Concentrations are MPN/100-mL.
9. Summary information for temporary well samples at Sites 1, 2, and 3. Values represent median concentrations. Concentrations are in mg/l (ppm), except Eh (mV), pH, specific conductance (umhos/cm). Within a row, different letters indicate median concentrations that differ at a significance level of 0.05. Values shown in bold indicate concentrations that differed between undeveloped and unsewered land uses.
10. Concentrations of chemicals in domestic wells at Sites 1, 2, and 3. Values represent median concentrations. Concentrations are in mg/l (ppm), except Eh (mV), pH, specific conductance (umhos/cm). Within a row, different letters indicate median concentrations that differ at a significance level of 0.05. Background concentrations are for water table aquifers from the statewide baseline study (MPCA, 1998b).

List of Figures

1. Location of study sites.
2. Sampling locations at Site 1.
3. Sampling locations at Site 2.
4. Sampling locations at Site 3.
5. Median nitrate concentrations at Sites 1, 2, and 3.
6. Distribution of nitrate in temporary well samples screened at the water table.
7. Distribution of nitrate and tritium with depth at Site 1.
8. Distribution of nitrate with depth at Site 2.
9. Distribution of nitrate as a function of age of well. Year represents the year that a well was drilled.
10. Distribution of Eh and nitrate with depth at Site 3.
11. Distribution of Eh with depth at Site 3.
12. Distribution of nitrate as a function of year of well installation at Site 3.
13. Locations where total coliform bacteria were detected in samples from domestic wells at Site 3. Contours show year in which the domestic wells were drilled..

Executive Summary

Ground water quality under subdivisions with Individual Sewage Treatment Systems (unsewered areas) often differs from water quality under areas serviced by municipal sewers. We conducted a study to evaluate water quality under three unsewered areas in Minnesota. The three sites are located in areas underlain by unconfined sand and gravel aquifers that are considered vulnerable to contamination. Site 1, located in central Minnesota, comprises approximately 1.5 square miles and consists primarily of single family dwellings, located on 0.25 to 1 acre lots, that are less than 15 years old. Site 2, located in north-central Minnesota, comprises approximately 2.5 square miles and consists primarily of single family dwellings, located on 1 acre lots, that vary in age from the 1970's to 1990's. Site 3 is an approximately 5 square mile area located in east-central Minnesota that consists primarily of single-family dwellings that vary in size from 0.25 to 0.5 acres and are generally more than 15 years old, with some lots being more than 30 years old.

Median nitrate concentrations at the water table were 4.10 mg/l at Site 1, 6.42 mg/l at Site 2, and 5.92 mg/l at Site 3. These concentrations were statistically equal between the three sites. Nitrate concentrations decreased rapidly with depth at Sites 1 and 2, with denitrification apparently accounting for most of the nitrate loss. At Site 3, nitrate concentrations slowly decreased with depth, probably because of both dilution and denitrification. Nitrate concentrations in domestic wells from all three sites increased with increasing age of the septic system, with a slope of approximately 0.17 mg/l/year. The correlation with age may be due to several factors. First, nitrate may gradually build up in ground water over time. Second, the ammonium adsorption capacity of the soil

treatment zone may be fully utilized in older systems, leading to increased leaching of ammonia. Third, ammonia may be weakly adsorbed as saturated and reducing conditions occur in the soil treatment zone under ISTS that are nonconforming. Ammonia reaching ground water may subsequently be oxidized, leading to nitrate contamination. Nitrate concentrations were not correlated with most other measured parameters. Twenty-seven percent of samples collected at the water table exceeded the drinking water standard of 10 mg/l, compared to a predicted value of 25 percent based on knowledge of the nitrate distribution at the three sites. Only one domestic well (1 percent of all samples) had a concentration greater than 10 mg/l, but an additional 20 wells had concentrations between 5 and 10 mg/l. Fifteen of these 20 wells were at Site 3.

Concentrations of other inorganic chemicals in ground water were below levels of concern. Phosphate concentrations were less than 0.05 mg/l. Concentrations of chloride were elevated at the water table at each of the three sites, but decreased with depth at Sites 1 and 2. Impacts from ISTS penetrated more deeply into the aquifer at Site 3, probably because of the older median age of ISTS and because the aquifer is well mixed and oxidized to depths of 100 feet or more.

Coliform bacteria were present in most samples collected at the water table. Concentrations were highest at Site 3, with a median concentration of more than 2000 MPN/100-mL. These elevated concentrations, combined with elevated concentrations of ammonia and Kjeldahl nitrogen at Site 3, suggest that nonconforming systems are present, since these chemicals should be observed in low concentrations under properly functioning systems.

Impacts from ISTS are evident in shallow ground water. These impacts can extend deeply into ground water if an aquifer is well mixed and oxidized conditions are maintained with depth. Nitrate and pathogens are the primary concerns for drinking water under unsewered developments. Proper maintenance of septic systems will minimize the potential impact from pathogens. Impacts from nitrate can be minimized by balancing lot size and well placement. In some aquifers, such as the aquifer at Site 3, larger lot sizes and stringent controls on maintenance of ISTS are needed to minimize impacts from septic systems.

Literature Review

Individual septic treatment systems (ISTSS) are designed to treat human waste. Properly functioning systems are effective at filtering raw waste, bacteria, viruses, and most chemicals (such as phosphorus and metals), thus preventing them from reaching ground water. They are not effective at attenuating nitrogen within the unsaturated zone, however, because ammonia-nitrogen in septic effluent is converted to nitrate at the bottom of the treatment system. This treatment system, called the biological mat, primarily attenuates organic matter and microbes (Noss and Billa, 1988; Kaplan, 1987; Brown, 1980; Walker et al., 1973a,b).

Table 1 summarizes the chemistry of effluent in septic tanks from several studies. Compared to ground water, septic effluent is enriched in ammonia, chloride, organic carbon, phosphate, potassium, sodium, and Kjeldahl (total) nitrogen. Septic effluent also contains pathogens and may contain elevated concentrations of volatile organic compounds (VOCs) and trace elements, such as boron, compared to ground water.

Chemical	Robertson et al., 1991	Harman et al., 1996	MPCA, 1999	Ground water¹
Alkalinity	365	802	432	249
Ammonia	30	128	56	0.17
Calcium	40	137	60	51
Chloride	45	207	51	0.8
Dissolved organic carbon	37	19.2	11	2.9
Magnesium	14	25	15	18
Nitrate-N	1	< 0.10	0.025	0.12
Phosphate	8	9	8	0.03
Potassium	12	43	16	0.5
Sodium	98	107	69	3.6
Sulfate-S	27	59	0.9	2.6
Total Kjeldahl nitrogen	-	152	70	0.7

¹ For unconfined sand and gravel aquifers (from MPCA, 1998)

Table 1: Comparison of chemical concentrations (mg/l) in septic tanks.

The primary concerns for ground water occur when septic systems are built on coarse-textured soils overlying a shallow unconfined aquifer, or when noncompliant¹ systems overlie a shallow unconfined aquifer. Chemicals that pass through the biological mat and into underlying coarse-textured soils (soil treatment zone) proceed vertically through the unsaturated zone and into ground water. When ISTSs fail, saturated conditions occur in the soil treatment zone and organic matter, phosphorus, and ammonia can leach to ground water. Once in ground water, a septic plume develops and moves with ground water flow. Approximate times for septic effluent to pass through the unsaturated zone to ground water range from a few hours to fifty days, depending on the volume of effluent and the distance to ground water (Robertson et al., 1991; Robertson, 1994; Robertson and Cherry, 1995). A septic plume may be characterized by concentrations of chemicals, such as chloride and sodium, which are higher than concentrations outside the plume.

We included a comprehensive literature review in this report. The review is divided into discussions of nitrate, pathogens, phosphorus, organic chemicals, and other inorganic chemicals.

Nitrate

Table 1 indicates that about 80 percent of the nitrogen in septic effluent is in the form of ammonia, with most of the remainder in organic forms. There is little nitrate in septic effluent. Wilhelm et al. (1994) and Harman et al. (1996) observed rapid

¹ Minnesota Rule 7080.0060 defines minimum technical standards and criteria for compliant ISTS. Systems not meeting these requirements are considered noncompliant. These include systems with inadequate

transformation of ammonia to nitrate directly beneath septic tile lines. Using data from Table 1 and assuming no organic nitrogen is oxidized to nitrate, about 25 percent of ammonia-nitrogen in septic effluent is transformed to nitrate-nitrogen (Robertson et al., 1991; Harman et al., 1996; MPCA, 1999a). The remainder may be lost as ammonia gas or adsorbed as ammonium. Assuming about 18 pounds of nitrogen per person are released to a septic system annually and considering a family of four on 0.5 acre lots, about 45 pounds of nitrogen per acre are added to ground water from ISTS. This may be a conservative value if organic nitrogen is transformed to nitrate or if soil adsorption capacity decreases over time. This value is less than the amount of nitrogen added under irrigated agriculture, where nitrogen losses range from 70 to more than 150 pounds per acre, depending on cropping practices and climate (Randall and Iragavarapu, 1995; Jemison and Fox, 1994; Steinhemmer et al., 1998). Even under fertilized pasture, nitrogen losses may exceed 50 pounds per acre (Owens et al., 1994). Under fertilized turfgrass, nitrogen losses are typically about 25 pounds per acre, while under unfertilized turfgrass, losses are about 10 pounds per acre (Geron et al., 1993).

There are significant differences in the distribution of nitrate under unsewered areas compared to other land uses. Nitrate is concentrated within a plume that originates beneath the septic drainfield. Consequently, nitrate concentrations may be very high, exceeding 50 mg/l, in portions of an aquifer impacted by ISTSs, while other portions of the aquifer may not be impacted.

Nitrate within the plume is conservative when oxygen is present and there is an insufficient food supply to sustain intensive microbial activity (Wilhelm et al., 1996;

vertical separation in the soil treatment area, systems that are not failing, and systems that do not pose an

Steinheimer et al., 1998). Nitrate within a septic plume moves at a rate similar to the ground water velocity. Nitrate plumes slowly attenuate due to plant uptake, dilution from recharge water, dispersion within the aquifer, and denitrification. None of these typically leads to rapid attenuation of nitrate and concentrations can exceed drinking water criteria at distances of 300 feet or more from the drainfield.

Predicting nitrate concentrations in an unsewered area with ISTSs consists of quantifying the contribution of each system. It is difficult, however, to predict the nitrate concentration in specific locations, because several factors affect nitrate leaching through the unsaturated zone and subsequent movement in ground water. These factors include density and age of septic systems, water use patterns, aquifer hydrogeology and geochemistry, and performance of individual systems. Mixing of individual plumes, seasonal changes in ground water flow and nitrogen inputs, effects of surface water bodies, and local pumping from wells further confound our interpretation of nitrate distribution in ground water.

Hantzsche and Finnemore (1992) drilled several wells in three different unsewered developments and reported mean nitrate concentrations ranging from 9.6 to 13.9 mg/l. Concentrations in individual wells ranged from less than 3.0 mg/l to 65.0 mg/l. Quan et al. (1974) sampled several domestic wells in a deep sand and gravel aquifer and found concentrations between 5.0 and 11.0 mg/l. Harmsen et al. (1996) installed multilevel wells along two hydrologic transects in two separate unsewered subdivisions with lot sizes ranging from 0.5 to 0.7 acres. Concentrations of nitrates ranged from 5.0 to 15.0 mg/l, with average concentrations of 6.0 to 8.0 mg/l. Tinker

(1991) measured nitrate concentrations in residential wells in five unsewered subdivisions with lot sizes ranging from 0.5 to 1.1 acre. Concentrations gradually increased from 1.0 to 13.0 mg/l from the up-gradient to the down-gradient edge of the subdivisions. Miller (1975) observed median nitrate concentrations of 2.0 and 13.0 mg/l in ground water under 0.5 acre lot developments that were two and 15 years old, respectively.

Ground water quality within an aquifer will continue to change until plumes from individual systems stabilize. Research results suggest that lot sizes of 1.0 acre or less in sandy soils result in nitrate concentrations exceeding the drinking water criteria of 10 mg/l over more than 10 percent of the aquifer (Hantzsche and Finnemore, 1992; Anderson et al., 1987; Baumann and Schafer, 1984). Considering one acre lots and a 30-foot wide plume where nitrate concentrations exceed 10 mg/l for a distance of 300 feet, about 20 percent of the upper few feet of an aquifer will be contaminated above the drinking water standard. Increasing lot size is an effective way of minimizing nitrate impacts to ground water, but this leads to urban sprawl and creates a variety of other environmental problems, such as habitat destruction and increased fuel consumption for transportation.

Pathogens

Human waste contains about 3×10^7 coliform bacteria/100-ml and may contain similar concentrations of viruses following viral infections (DeBorde et al., 1998). The most common water-borne diseases are typhoid fever, bacillary dysentery, infectious hepatitis, and amoebic dysentery (Brown, 1980). Most bacteria and parasites are filtered out of the soil, but this mechanism is not very effective for viruses. The main removal

mechanism for viruses is adsorption, primarily to organic matter and clay (Kaplan, 1987). Bacterial survival in the drainfield is typically less than 100 days. A septic system in a sandy soil can be very effective at decreasing bacteria numbers because of filtration in the bio-mat followed by unsaturated flow through small pores below this, as well as low soil moisture. However, a strong recharge pulse or a fluctuating water table can wash many of the bacteria from the bio-mat if they remain viable (Reneau, 1978). Viruses survive longer than bacteria, but their life expectancy is less than one year on average. High temperatures, dry soils, and low pH will decrease concentrations of bacteria and viruses. Effective removal of bacteria requires 3 to 4 feet of unsaturated soil and a stable water table. If bacteria do pulse through with recharge, they are somewhat conservative and rapid breakthrough is observed.

Bacteriophage or poliovirus are good indicators of virus transport, but coliform bacteria are the most common indicator used in research. Survival time of viruses in ground water are usually days or less, but viruses are very mobile. Virus and bacterial transport in ground water can be rapid and occur over distances of 400 feet or more (Yates and Yates, 1988; Brown, 1980; Hagedorn et al., 1978; Reneau, 1978).

Hagedorn et al. (1978) observed movement of *Streptococcus* and *E. coli* primarily in down-gradient flow directions in ground water. They observed that peak bacterial counts in shallow water table wells located within 10 feet of the drainfield corresponded with the peak precipitation events in the wells. There was a lag of around 8 days in deeper water table wells. Bacteria always moved 10 feet or more within 24 hours of a rain event. This is faster than predicted and reflects some dispersion mechanism in ground water. Bacterial counts were similar within three feet of the disposal area, but

then decreased by 2 to 3 orders of magnitude within 45 feet. Reneau (1978) observed about an order of magnitude decrease in populations in ground water for every 10 feet down-gradient of the drainfield. MPCA (1999a) observed concentrations of total coliform bacteria in excess of 200 MPN (most probable number)/100-mL in septic tank effluent from seven systems in Baxter, Minnesota, but concentrations were typically less than 1 MPN/100-mL within 30 feet of the treatment system.

Phosphorus

Phosphorus is the chemical of greatest concern for septic systems located adjacent to lakes, rivers, and wetlands. Concentrations in excess of about 0.050 mg/l can stimulate growth of algae in surface waters. Table 1 indicates that phosphorus concentrations in septic effluent are about two orders of magnitude higher than this. Wilhelm et al. (1994) and Brown (1980) report that most phosphorus goes into the septic tank in organic forms, but that orthophosphate accounts for about 80 percent of the total phosphorus in the tank effluent. We observed similar results in samples from septic tanks in Baxter, Minnesota (MPCA, 1999a).

Phosphorus precipitates and is adsorbed in the unsaturated zone. Calcium carbonate, metal oxides, and organic matter are the primary adsorption media. Aluminum and iron species control precipitation reactions (Harman et al., 1996). Adsorbed phosphorus is sparingly soluble, and 50 percent or more cannot be desorbed. Within the soil treatment system, particularly in close proximity to the septic pipes, precipitation reactions control the fate of phosphorus (Robertson et al., 1998).

Phosphorus moves in ground water but, because it is strongly adsorbed, movement is retarded compared to the velocity of water and other solutes such as chloride and nitrate. Retardation rates vary from 20 to 100 (Robertson et al., 1991; Robertson et al., 1998; Harman et al., 1996). Robertson et al. (1998) observed phosphorus migration exceeding 30 feet in six of ten plumes investigated. The greatest movement occurred in calcareous sands. Harman et al. (1996) found phosphorus concentrations of less than 1 mg/l directly beneath a 44-year old system servicing an elementary school. Rea and Upchurch (1980) observed adsorption of phosphorus within 45 feet of the drainfield. Reneau and Pettry (1976) found little detectable phosphate within 40 feet of the drainfield. Reneau (1979) observed a logarithmic decrease in phosphate down-gradient of the drainfield. MPCA (1999a) found that phosphorus in septic plumes approached background concentrations within 40 feet of the treatment system. Movement of phosphorus rarely exceeds 15 to 40 feet, even in very old systems (Wilhelm, et al., 1994; MPCA, 1999a). The literature suggests that compliant septic systems located 100 or more feet from a surface water body will effectively minimize the risk of phosphorus loading.

Organic chemicals

Synthetic organic chemicals are not a component of human waste. Oil and optical brighteners, however, are commonly found in laundry effluent (Fay et al., 1995; Alhajjar et al, 1990; Kaplan, 1987). Some Volatile Organic Chemicals (VOCs) have been used for odor control (Robertson, 1994). VOCs may also be introduced into septic systems after household use (e.g. paints, varnishes, degreasers). Most organic compounds will be

attenuated within the soil treatment system because they are strongly adsorbed to organic matter. If synthetic organic chemicals pass below the treatment zone, they will be conservative in ground water and may therefore travel long distances within the aquifer. We observed VOCs in septic tank effluent from seven systems sampled in Baxter, Minnesota (MPCA, 1999a). As many as 11 different chemicals were detected in individual plumes, with the maximum total concentration of VOCs being 0.80 mg/l. Toluene, xylene, methyl ethyl ketone, and chloroform were the most frequently detected chemicals. Toluene, xylene, and methyl ethyl ketone are associated with oils, paints, and household solvents. Chloroform occurs when chlorine is used as a disinfectant in water supplies. Despite the presence of VOCs in septic effluent, they were detected in only a few samples at very low concentrations in ground water down-gradient of the treatment system.

Other Inorganic Chemicals

Human waste contains a variety of other chemicals, including chemicals found at high concentrations (sodium, potassium, and chloride) and several elements found at relatively low concentrations. Table 2 summarizes chemical information from our study of septic plumes in Baxter, Minnesota (MPCA, 1999a). Boron, chloride, iron, potassium, and sodium concentrations were much higher in septic effluent than in ground water not impacted by a septic plume. Concentrations of calcium, chromium, zinc, and manganese were somewhat higher in septic effluent. Chloride, potassium, and sodium are potentially good indicators of septic impacts to ground water. Chloride is by far the most conservative of these in ground water. We observed chloride concentrations in plumes

greater than background concentrations for distances of up to 400 feet in Baxter, Minnesota, although the average distance was about 150 feet. Sodium was attenuated in ground water, but was often observed at elevated concentrations more than 100 feet from the drainfield. Concentrations of these major ions were well below drinking water standards.

Chemical	Tank effluent	Maximum in plume	Ground water¹
Aluminum	< 0.050	< 0.050	< 0.050
Boron	0.62	0.27	0.033
Cadmium	< 0.0015	< 0.0015	< 0.0015
Calcium	60	46	48
Chloride	51	107	20
Chromium	0.0067	< 0.0031	0.0041
Copper	< 0.0051	0.0055	0.0058
Iron	1.4	0.23	0.075
Lead	< 0.025	< 0.025	< 0.025
Magnesium	15	12	10
Manganese	0.35	0.065	0.10
Nickel	< 0.0060	0.013	< 0.0060
Potassium	16	13	4.7
Sodium	69	145	12
Sulfate	0.90	9.3	4.1
Zinc	0.030	0.0098	0.0074

¹ Data for a shallow sand and gravel aquifer, Baxter, Minnesota (from MPCA, 1999a)

Table 2: Comparison of chemical concentrations in septic tank effluent, maximum concentrations found in ground water plumes, and background concentrations in ground water. Concentrations are in mg/l.

Table 2 indicates that concentrations of trace elements under septic drainfields are similar to background concentrations, except for boron. There is little information in the literature on the fate of trace elements from septic waste. Harman et al. (1996), Robertson and Blowes (1995), and MPCA (1999a) observed concentrations of trace metals that approached background concentrations in ground water within a few feet of the septic drainfield. Most trace elements are strongly adsorbed in the soil treatment

system, except in poorly buffered soils where the pH is less than 6.0. Boron is a somewhat conservative chemical in ground water (MPCA, 1999a), but concentrations within the septic plume are typically below the drinking water standard of 0.60 mg/l.

MATERIALS AND METHODS

In 1998 and 1999 we sampled ground water under three unsewered subdivisions in Minnesota. Our objectives were to

1. determine the spatial distribution of nitrate in ground water under unsewered subdivisions; and
2. identify factors affecting the distribution of nitrate under unsewered communities.

Description of Study Areas

Site 1 is a 1.5 square mile area located near St. Joseph, Minnesota (Figure 1). This area of the state has experienced rapid urban growth within the past 10 years, primarily associated with the nearby metropolitan area of St. Cloud. The surficial geology of the area consists primarily of outwash deposits of sand and gravelly sand (Myer and Knaeble, 1995). Localized alluvium and organic deposits less than six feet thick occur along Watab Creek and Watab Lake, which dissect the study area. Soils consist of the Hubbard-Dickman association, which comprises nearly level to gently sloping, excessively and well-drained, moderately coarse- and coarse-textured deposits on outwash plains and stream terraces (United States Department of Agriculture, 1985). Average annual precipitation is 27.6 inches, with 17.6 inches occurring during the period May through September. Average annual recharge is approximately 9.8 inches.

Site 2 (Figure 1) is a 2.5 square mile area located in Baxter, Minnesota. Baxter is part of an urban area of about 40000 people. Considerable changes in land use have occurred in the past 10 years, including expansion of areas served by ISTSs. Surficial geology consists of outwash deposits of sand and gravelly sand. Localized alluvium consisting of less than six feet of silt loam and loamy sand occurs along rivers in the area. Isolated peat deposits and sandy till lenses are found in the study area but comprise small areas. Peat deposits, where they are mapped, are associated with organic deposits greater than three feet in thickness and are found in marshes, often adjacent to lakes and drainageways. Till deposits are sandy loam, unsorted, and become very dense with depth. Precambrian age deposits underlie the area, but these are more than 100 feet below the land surface and are unimportant hydrologically. Soils belong to the Menahga and Zimmerman series. These soils consist of light colored, well drained, medium acid loamy soils 4 to 8 inches thick over fine and medium sand. Sands are limy below 3 to 5 feet. Peat occurs in depressions over approximately five percent of the area. Poorly drained sandy tills, such as those from the Lino series, occur as lenses within the well-drained sands (USDA). Average annual precipitation in the study area is about 28.3 inches, 10.6 of which falls between June and August. Average July and January temperatures are approximately 69.8 and 10.4 °F, respectively. The frost-free period extends from mid-May to late-September. A shallow sand and gravel aquifer occurs within 15 feet of the land surface. The aquifer is unconfined and ranges in thickness from 30 to 90 feet. Annual recharge to the aquifer is approximately 5.9 inches.

Site 3 is a 5 square mile area located in eastern Washington County near Lakeland, Minnesota (Figure 1). The Lakeland area is a small, unsewered community

consisting of approximately 5000 people. Surficial geology consists of coarse sand, gravel, and gravelly sand terrace deposits (Minnesota Geological Survey, 1990). Bedrock units of Early Paleozoic Age underlie the township. Soils from the Sparta-Dickman-Hubbard association cover the area. These are level to moderately steep, well drained and excessively drained, medium and coarse textured soils (USDA). Average annual precipitation is 27.6 inches, with about 17.7 inches falling between April and September. Depth to bedrock varies from zero to over 180 feet, but is between 70 and 100 feet across most of the area. An unconfined sand and gravel aquifer underlies the area and is found at a depth of 45 to 75 feet below the land surface.

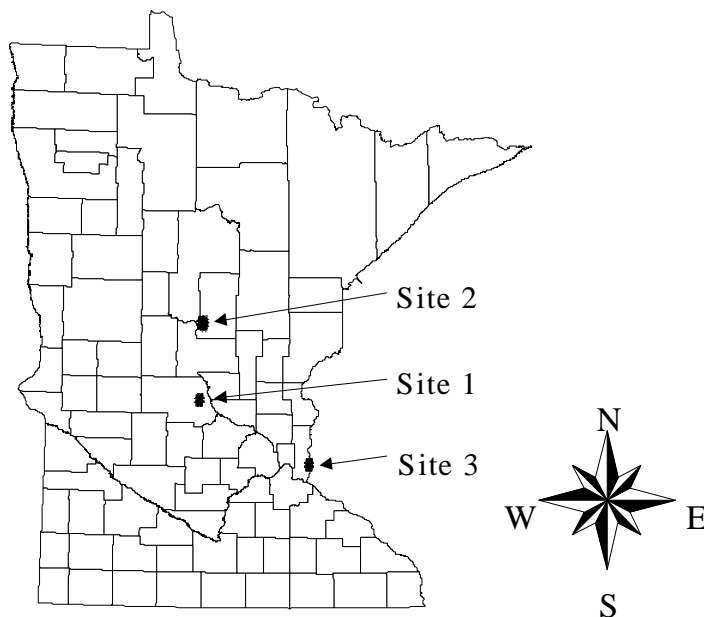


Figure 1: Location of study sites.

Site Characteristics

Ground water underlying each of the three study areas is classified as being vulnerable to contamination. These designations are based on the presence of coarse textured soils that overlie unconfined aquifers. Water thus travels quickly through the unsaturated zone and into ground water.

In addition to the hydrogeologic vulnerability, the three study areas were chosen because they represent a variety of characteristics that could potentially affect ground water quality. These characteristics include age of development, lot size, type of use (seasonal or year round), family size, likelihood of having compliant systems, and physical factors such as depth to water and type of aquifer underlying each area. Table 3 summarizes some of these factors.

Characteristic	Site 1	Site 2	Site 3
Median age of system (years)	8	6	20
Median lot size (acres)	0.25	1	0.25 to 0.5
Type of use	Permanent	Seasonal/permanent	Permanent
Median home size	3.49 people	3.5	3 bedroom
% Compliant systems ¹	60% have permit	50%	Probably < 50%
Median depth to water	16	15	45
Aquifer type	Medium sand	Fine sand	Sand over limestone

¹ Estimates provided by local water managers

Table 3: Characteristics of the three study areas.

Site 1 (St. Joseph's) is a small subdivision surrounded by row crop agriculture. Most of the homes were built between 1985 and 1995. The area is slowly transitioning to unsewered land use. Lot sizes of newer developments are typically one acre or more, but most of the area sampled for this study represents older developments, with the average lot size being less than 0.5 acre.

Site 2 (Baxter) includes a mixture of old and new homes. Although the median age of a home in the Baxter area is about 6 years, core areas in Baxter developed in the 1970's, typically on 1 acre lots, often along lakes. Development in the Baxter area has increased rapidly in recent years.

Site 3 (Lakeland) consists of homes that were built on 0.25- to 0.5-acre lots more than 20 years ago. It is likely that many of the older systems are not compliant. High nitrate concentrations have been documented for several years.

Sampling Design and Data Collection

Sampling at each site included temporary and permanent wells screened at various depths in the aquifers. Figures 2 through 4 illustrate sampling locations. Temporary wells consisted of using direct push technology to install 1.25 inch wells that were screened at the water table or within ten feet of the water table. Permanent domestic wells comprised deeper sampling points.

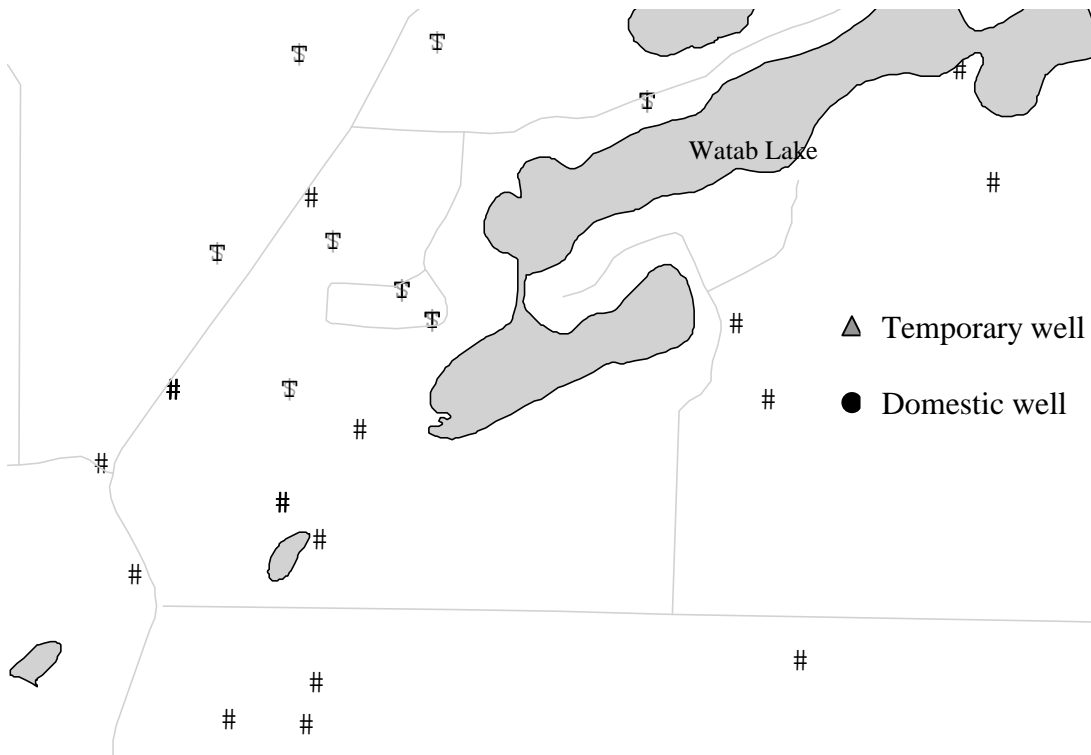


Figure 2: Sampling locations at Site 1 (St. Joseph's).

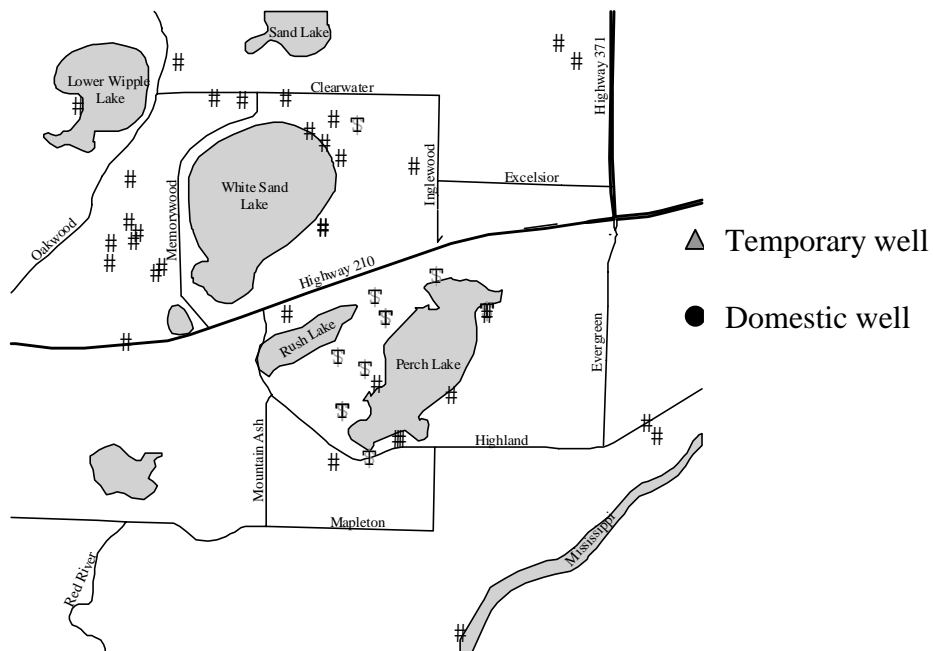


Figure 3: Sampling locations at Site 2 (Baxter).

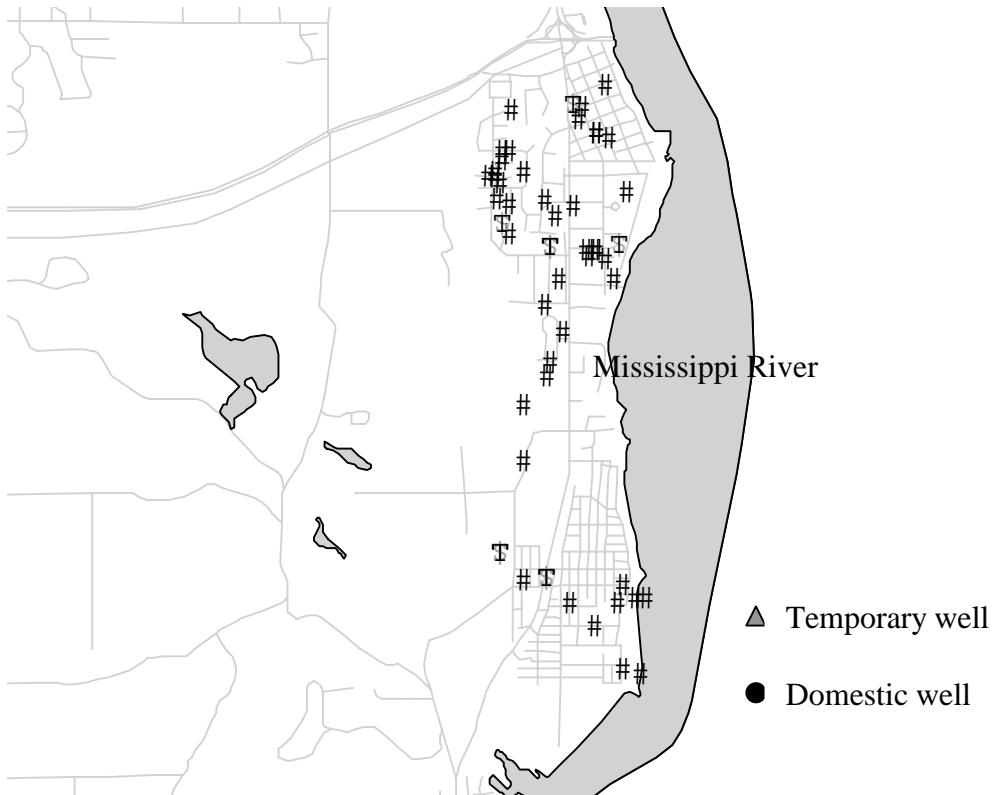


Figure 4: Sampling locations at Site 3 (Lakeland).

Samples from domestic wells were collected from an outside spigot. Once the spigot was turned on, continuous measurements of oxidation-reduction potential (mV), temperature ($^{\circ}\text{C}$), pH, specific conductance ($\mu\text{mhos/cm}$), and dissolved oxygen (mg/l) were taken using a YSI 600 XL multiparameter probe. Samples were collected once temperature, specific conductance, and pH had stabilized to 0.1°C , 10%, and 0.1 pH unit, respectively, for three successive readings. Samples for inorganic chemicals were analyzed at the University of Minnesota Research Analytical Laboratory. MPCA (1998b) presents laboratory analytic methods and reporting limits. Samples for total coliform bacteria and *E. coli* bacteria were analyzed at the Minnesota Department of Health. Alkalinity, chloride, nitrate, reduced iron, and sulfate were measured in the field using

commercially available field test kits. MPCA (1998c) describes field sampling methods. All samples for laboratory analysis were packed in coolers at 4°C and delivered to laboratories within required holding times.

At temporary wells, we collected continuous soil samples in four foot intervals and performed textural analysis. For samples collected at the water table, four-foot screens (0.010 slot, stainless steel) were installed approximately halfway across the water table. A two-foot screen was used at deeper depths to provide a more discrete sampling interval. Samples were pumped using a peristaltic pump until sample turbidity became uniform and field parameters stabilized. Sample parameters were the same as for domestic wells.

Decontamination procedures for bacteria samples included scrubbing the screen, screen sheath, and any probe rod or connections that intersected the water column with a bleach solution (approximately 1 cup bleach per 5 gallons water). The equipment was then rinsed with tap water and then with deionized water. Sampling tubing was discarded after each use. Latex gloves were worn during sampling.

Field and laboratory duplicates were performed at a rate of ten percent. There were no significant differences in concentrations between primary and duplicate samples. Laboratory spikes were within 5 percent of complete recovery for anions and within 10 percent for cations. Table 4 summarizes the number of samples collected for each study.

	Site 1	Site 2	Site 3
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Chemical	Temporary well	Domestic well	Temporary well	Domestic well	Temporary well	Domestic well
Alkalinity	0	24	12	0	0	42
Aluminum	11	24	0	0	0	29
Ammonia	11	24	12	0	9	44
Bacteria	6	23	0	0	9	43
Boron	11	24	12	0	0	29
Cadmium	11	24	0	0	0	29
Calcium	11	24	12	0	0	29
Chloride	11	24	12	32	9	29
Chromium	11	24	0	0	0	29
Copper	11	24	0	0	0	29
Dissolved organic carbon	10	24	12	0	9	2
Dissolved oxygen	11	24	12	32	8	43
Eh	11	24	12	31	9	43
Iron	11	24	12	33	0	29
Lead	11	24	0	0	0	29
Magnesium	11	24	12	0	0	28
Manganese	11	24	12	0	0	29
Nickel	11	24	0	0	0	29
Nitrate	11	24	12	33	9	44
pH	11	24	12	32	8	43
Phosphate	11	24	12	0	0	29
Phosphorus	11	24	12	0	9	0
Potassium	11	24	12	0	0	28
Sodium	11	24	12	0	0	29
Specific conductance	11	24	12	32	8	43
Sulfate	11	24	12	0	9	29
Temperature	11	24	12	31	8	43
Total Kjeldahl nitrogen	11	24	12	0	9	0
Tritium	0	24	0	0	0	0
Zinc	11	24	12	0	0	29

Table 4: Number of samples collected for different chemicals in each study area.

Statistical Methods

Statistical methods included the Kruskal-Wallis and Mann-Whitney tests for group comparisons. Factors included study area and well depth. We used the Spearman rank method to identify relationships between sampled chemicals and parameters.

We used two independent methods to test for normality in our sample populations. These included regression analysis and the Komogorov-Smirnov or Shapiro-Wilk tests. If a population is normally distributed and there are no samples below the reporting limit, we can estimate the percentage of samples that will exceed a certain concentration by calculating the standard error of the population. Usually, we want to calculate the percent of samples that will exceed the drinking water standard. When some of the samples were below the reporting limit for a particular chemical, but we knew the population was normally distributed, we used the Helsel Robust Method (Newman et al., 1995). This method assumes a normal distribution and calculates or predicts concentrations for the samples below the reporting limit by using information from the data that were above the reporting limit. We then calculated the standard error with these new values and determined the percent of samples above the drinking water standard.

Results and Discussion

We present results separately for nitrate, phosphorus, bacteria, and for other inorganic chemicals.

Nitrate

Figure 5 illustrates median concentrations of nitrate in temporary well samples from the three study sites. These represent nitrate concentrations near the water table, where we expect impacts from septic systems to be greatest. There were no significant differences in nitrate concentrations between the three study areas ($p = 0.612$).

Concentrations were 4.10 mg/l at Site 1, 6.42 mg/l at Site 2, and 5.92 mg/l at Site 3. The drinking water standard of 10 mg/l was exceeded in 3 temporary wells at Site 1 (25 percent of sampled wells), 3 wells at Site 2 (33 percent), and 1 well at Site 3 (9 percent).

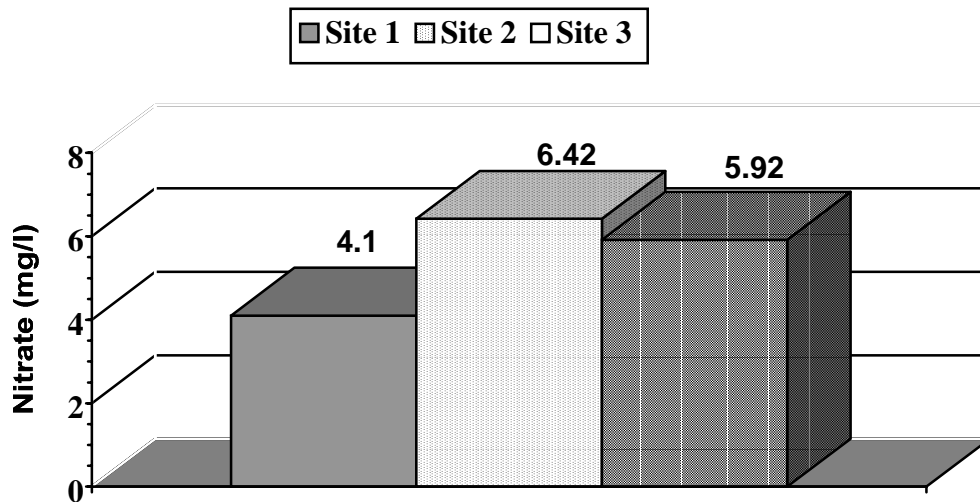


Figure 5: Median nitrate concentrations in temporary wells at Sites 1, 2, and 3.

A few temporary well samples at each site were below the reporting limit. In samples with detectable levels of nitrate, we ordered the samples from low to high nitrate concentration and found that the concentration of nitrate increased linearly with the sample number at Sites 2 and 3 ($p < 0.01$) and was described by

$$\text{Nitrate Concentration} = a + b * \text{sample number} \quad [\text{Eq. 1}]$$

where a is the intercept and b the slope (Figure 6). Eq. [1] was not a strong indicator of the distribution at Site 1. This may be due to the presence of denitrifying conditions in some samples at Site 1 or to the presence of a single high concentration in one of the sampled wells. Eq. [1] assumes that denitrifying conditions are not present in a sample. Table 5 provides a summary of the regression analysis using Eq. [1]. Slopes of the regression lines differed significantly between sites. The significant correlation between nitrate and sample number at Sites 2 and 3 indicates a normal distribution of nitrate in ground water at these sites. Further tests of normality using the Komolgorov-Smirnov and Shapiro-Wilk tests confirmed the normal distribution of nitrate at these two sites ($p > 0.100$). This means we can use the Helsel Method to estimate the percent of samples that are likely to exceed the drinking water standard of 10 mg/l. Using this method, we calculated that 28 percent of samples at Site 2 would be expected to exceed the drinking water standard, and 22 percent at Site 3. Comparing this to the percentages that we observed, we find that predicted (28 percent) and measured (33 percent) values are relatively close at Site 2. At Site 3, our sampling (9 percent) greatly underestimated the expected percent (22 percent).

Site	p-value	Correlation coefficient	Slope	UCL of slope
1	0.0501	0.606	1.99	2.97
2	0.00078	0.944	3.47	3.94
3	0.00084	0.795	1.42	1.79

Table 5: Summary of regression analysis using Eq. [1]. UCL = the 95th percent upper confidence limit of the slope.

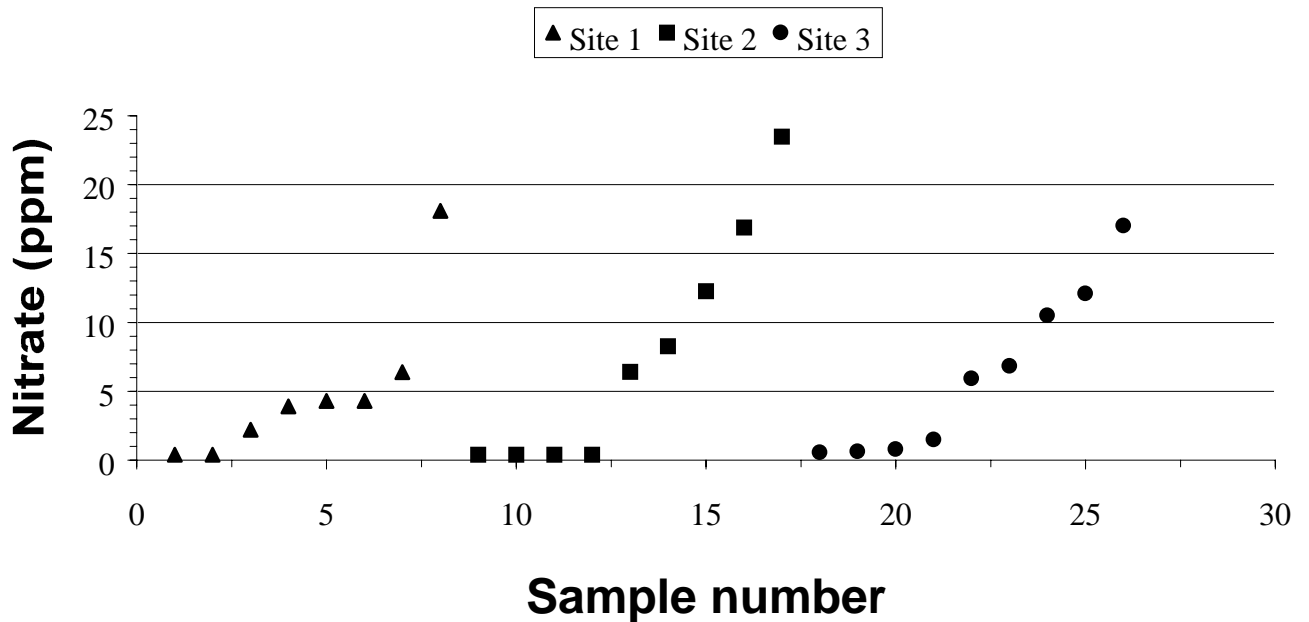


Figure 6: Distribution of nitrate in temporary well samples screened at the water table.

At Site 1, the distribution of nitrate at the water table was affected by aquifer geochemistry. The median Eh was 227 mV and the median concentration of dissolved oxygen was 0.8 mg/l. These are conditions favorable for denitrification. Nitrate was present only in samples with Eh values greater than about 235 mV. Nitrate concentrations in these samples were strongly correlated with concentrations of dissolved oxygen ($p < 0.001$, $R^2 = 0.835$) and negatively correlated with concentrations of manganese ($R^2 = -0.518$). Because of the presence of nitrate-reducing conditions, the impact of septic systems on the distribution of nitrate was masked by the geochemistry of ground water. Nitrate was negatively correlated with dissolved organic carbon ($R^2 = -0.369$), total Kjeldahl nitrogen ($R^2 = -0.376$), and total coliform bacteria ($R^2 = -0.627$).

The correlations with carbon, Kjeldahl nitrogen, and coliform bacteria may provide evidence of the presence of noncompliant septic systems. Only 60% of the systems are considered to be in compliance within the study area. Many of the homes are more than 15 years old, although the median age was 8 years. Organic matter and bacteria are introduced to ground water when septic systems are not functioning properly. Under these conditions, ammonia, rather than nitrate, will be the form of nitrogen most prevalent in ground water. Table 6 summarizes a comparison of geochemistry in samples with detectable nitrate and those with no detectable nitrate. The data show that total concentrations of nitrogen in the upper part of the aquifer are similar in oxygenated and anaerobic samples, but that the form of nitrogen differs. The total concentration of about 1 mg/l-nitrogen is significantly higher than the 0.29 mg/l-nitrogen observed in nearby samples collected from areas with undeveloped land use (MPCA, 1999b). These results indicate that about 0.7 mg/l of nitrogen have been added to the aquifer from septic systems (about 3.1 mg-nitrate/l).

Characteristic	Detectable nitrate	No detectable nitrate
Nitrate (mg/l)	4.3	< 0.50
Dissolved oxygen (mg/l)	3.75	0.48
Eh (mV)	285	186
Ammonia (mg/l)	0.07	1.10
Kjeldahl nitrogen (mg/l)	0.22	2.06
Nitrogen as N (mg/l) ¹	1.06	0.96
% of Nitrogen as nitrate	91	6

¹ Does not include Kjeldahl nitrogen

Table 6: Comparison of geochemistry in samples having detectable nitrate and having no detectable nitrate.

Nitrate concentrations at Site 1 decreased rapidly with depth (Figure 7). Within about 10 to 15 feet of the top of the aquifer, nitrate concentrations decreased to below the reporting level of 0.50 mg/l. In 12 of 24 samples collected more than 10 feet below the

top of the aquifer, tritium was detected at a concentration greater than 10 TUs (MPCA, 1999b). The water from these samples is thus considered to be younger than 1953. Since there is “young” water deeper in the aquifer but nitrate concentrations are low, nitrates are being attenuated with depth. Denitrification and dilution are the two most likely attenuation processes. Denitrification is probably the more important mechanism, since nitrate concentrations decrease abruptly below 10 feet. Dilution would lead to a more gradual reduction in nitrate concentrations. Most wells at Site 1 are screened below the oxidation-reduction (redox) boundary, where nitrate is denitrified, and are therefore not vulnerable to nitrate contamination.

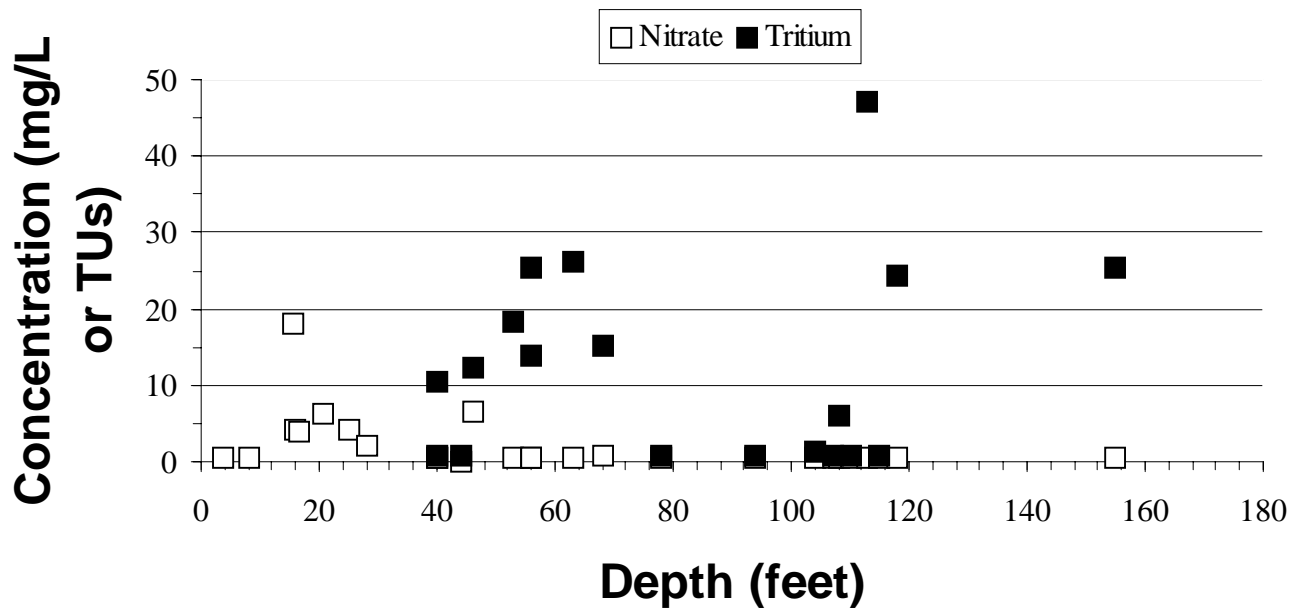


Figure 7: Distribution of nitrate and tritium with depth at Site 1.

In temporary well samples from Site 2, there were no differences in redox chemistry between samples with detectable nitrate and samples with no detectable nitrate (Table 7). Eh and dissolved oxygen concentrations in samples with no detectable nitrate were 268 mV and 9.33 mg/l, respectively, compared to 245 mV and 4.08 mg/l in samples

with detectable nitrate. The upper few feet of the aquifer is oxygenated throughout the study area and is therefore vulnerable to nitrate contamination. The highest nitrate concentrations were located between White Sand and Perch lakes. These are heavily developed areas where homes are typically more than 15 years old. Although Table 3 indicated most septic systems are in compliance in the study area, three of four sites we inspected showed inadequate soil thickness over the septic drainfield (MPCA, 1999a).

Site	Nitrate	Ammonia (mg/l)	DOC ¹ (mg/l)	TKN (mg/l)	DO (mg/l)	Eh (mV)	Chloride (mg/l)
2	Detected	0.060	3.7	0.42	4.16	224	14.2
	Not detected	0.048	3.1	0.15	5.51	221	8.9
3	Detected	0.74	3.2	1.54	10.10	453	66.4
	Not detected	0.46	3.1	1.52	7.40	476	34.1

¹ DOC=Dissolved organic carbon, TKN=Total Kjeldahl nitrogen, DO=dissolved oxygen

Table 7: Geochemistry for samples in which nitrate was detected and not detected in temporary well samples screened at the water table.

Nitrate decreases abruptly at a depth of about 15 feet below the water table at Site 2. Within the upper 15 feet of the aquifer, the median concentration of dissolved oxygen was 2.62 mg/l, median Eh was 261 mV, the median iron concentration was 0.17 mg/l, and the median nitrate to chloride ratio was 0.08. Below 15 feet, the median concentration of dissolved oxygen was 0.39 mg/l, median Eh was 77 mV, median iron concentration was 2.6 mg/l, and the median nitrate to chloride ratio was 0.03. Conditions below 15 feet are nitrate-reducing, but the decrease in nitrate to chloride ratio with depth is not particularly rapid. It is possible that there is little mixing throughout the aquifer, as was evident at Site 1. We did not collect tritium samples at Site 2 and cannot therefore suggest a mechanism for nitrate attenuation with depth.

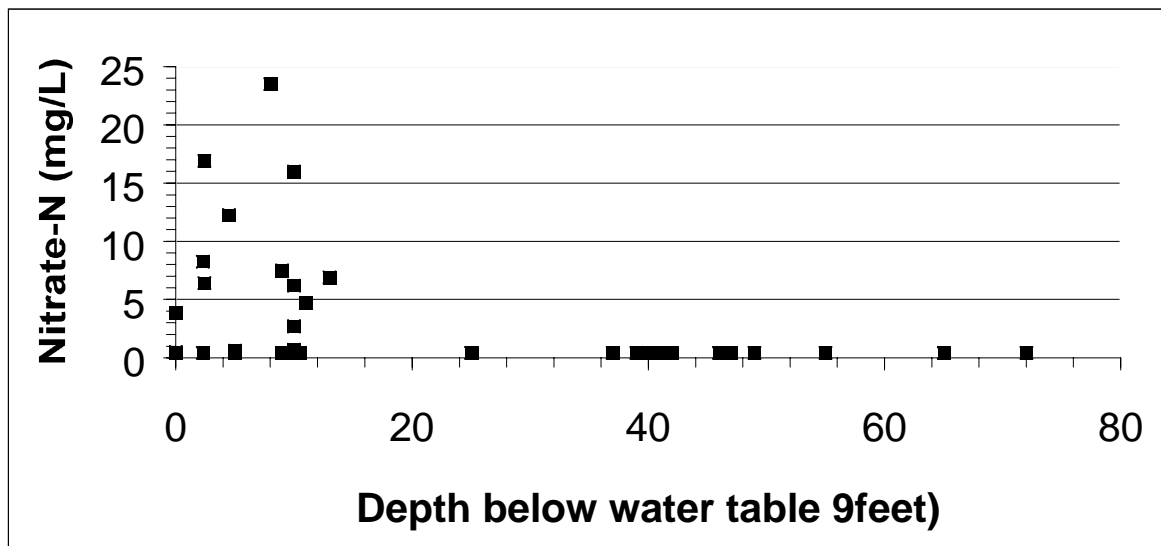


Figure 8: Distribution of nitrate with depth at Site 2.

Domestic wells with detectable nitrate at Site 2 were associated with septic systems that were more than 20 years old, while samples with no detectable nitrate were associated with septic systems that were generally less than 10 years in age (Figure 9). It is unclear if the differences in nitrate concentration are due to performance of the septic systems or to gradual accumulation of nitrates in ground water. Most of the systems in older areas of Site 2 are likely to be noncompliant. This should have little effect on nitrate concentrations unless the ammonium adsorptive capacity of soils has been utilized. There are many shallow domestic wells at Site 2 and these may be vulnerable to nitrate contamination. Wells screened more than 15 feet below the top of the aquifer do not appear to be vulnerable to nitrate contamination.

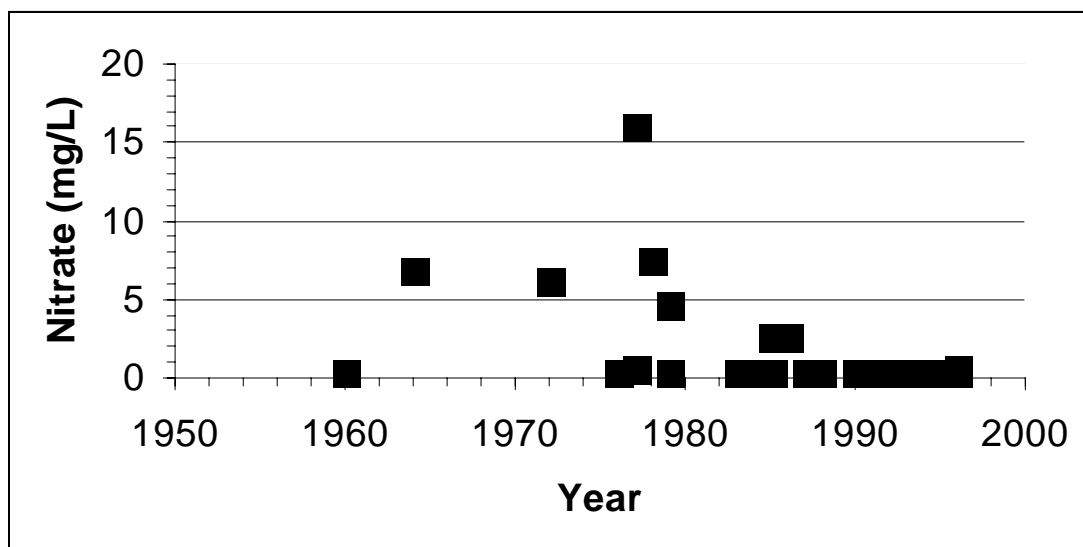


Figure 9: Distribution of nitrate as a function of age of well. Year represents the year that a well was drilled.

In temporary well samples at Site 3, redox differences were not evident between samples having detectable nitrate and those with no detectable nitrate (Table 7). As with Site 2, the distribution of nitrate appears to be controlled by inputs of nitrogen. Nitrate concentrations in temporary well samples were positively correlated with chloride, sulfate, depth to water, and total coliform bacteria ($p < 0.01$ for all tests).

Nitrate concentrations do not decrease with depth in the upper 130 feet of the sand and gravel aquifer (Figure 10). Concentrations in the underlying bedrock aquifer are below the reporting limit of 0.50 mg/l. Figure 11 summarizes Eh concentrations with depth in the sand and gravel aquifer and for the bedrock aquifer. The entire sand and gravel aquifer is oxidized and nitrate is not reduced. The aquifer appears to be well mixed as evidenced by the presence of nitrate to 130 feet. There is no correlation between nitrate concentration and the year in which a well was drilled (Figure 12). The entire sand and gravel appears to be vulnerable to contamination with nitrate.

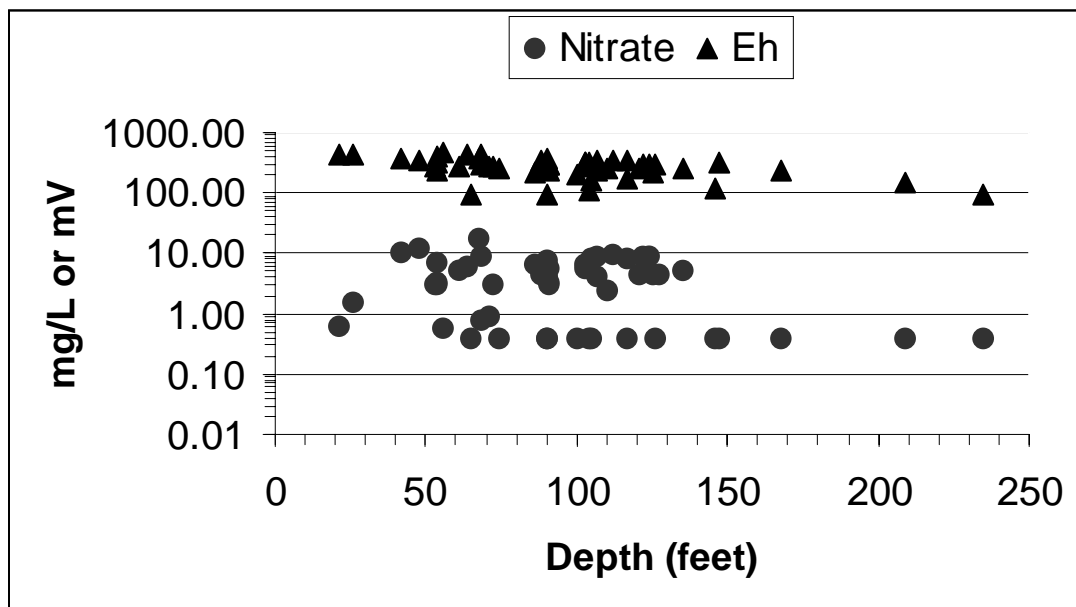


Figure 10: Distribution of Eh and nitrate with depth at Site 3.

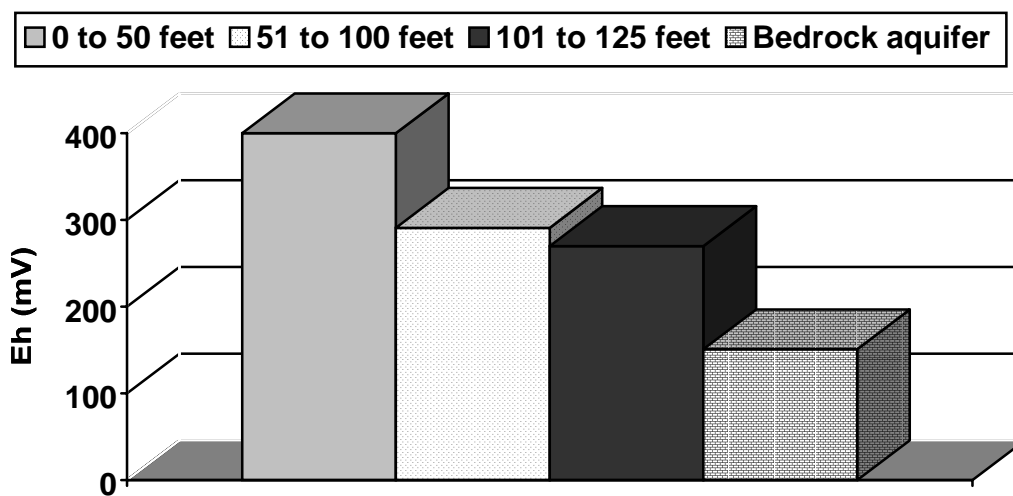


Figure 11: Distribution of Eh with depth at Site 3.

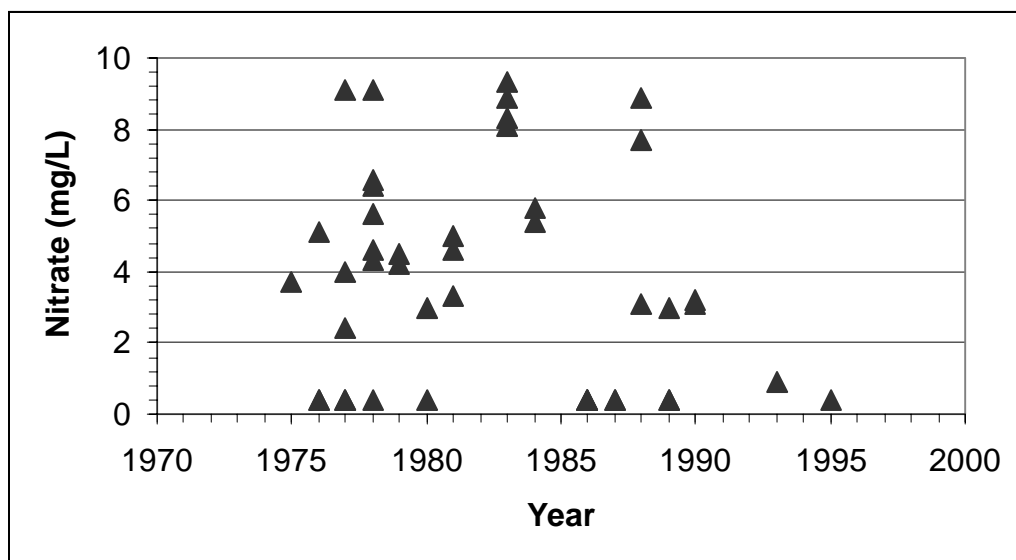


Figure 12: Distribution of nitrate as a function of year of well installation at Site 3.

Bacteria

There were 46 samples for *E. coli* and total coliform bacteria collected from domestic wells at Site 3 and 25 samples at Site 1. Six samples were collected from temporary well samples at both sites. Bacteria samples were not collected at site 2.

Coliform bacteria were detected in all six temporary well samples at Site 1 (Table 8). The concentration in one sample exceeded 200 MPN/100-mL, but concentrations in the remaining samples were less than 100 MPN/100-mL. Concentrations of coliform bacteria in temporary well samples were positively correlated with sodium ($R^2 = 0.578$), total Kjeldahl nitrogen ($R^2 = 0.549$), phosphorus ($R^2 = 0.401$), ammonia ($R^2 = 0.344$), and negatively correlated with specific conductance ($R^2 = -0.766$), Eh ($R^2 = -0.352$), and depth to water ($R^2 = -0.342$). The correlations with reduced forms of nitrogen and with phosphorus indicates that noncompliant systems may exist within the study area.

Coliform bacteria were detected in 11 of 25 domestic wells sampled at Site 1. One well had a concentration greater than 200 MPN/100-mL, but concentrations were less than 25 MPN/100-mL in the remaining wells. There were no correlations between concentrations of coliform bacteria and other sampled parameters, including depth to water and well depth. Since we do not know the age of each system we sampled, we used the age of a well as a surrogate for age of the septic system. In most cases, this is an accurate approximation, since both well and septic system would have been installed about the time the home was constructed. The median age of wells for samples with detectable bacteria and no detectable bacteria were equal (8 years). *E. coli* were detected at a concentration of >200 MPN/100-mL in the temporary well sample that had total coliform concentrations greater than 200 MPN/100-mL. *E. coli* were not detected in any other temporary well or domestic well sample.

	Site 1		Site 3	
	Temporary wells	Wells	Temporary wells	Wells
No. samples	6	25	6	46
No. detections	6	11	6	9
Median	34.5	< 1	1800	< 1
Minimum	1	< 1	116	< 1
Maximum	> 200	> 200	> 2000	> 200

Table 8: Data summary for total coliform bacteria from temporary well and domestic well samples. Concentrations are MPN/100-mL.

Coliform bacteria were detected in all temporary well samples at Site 3. Concentrations in three samples at Site 3 exceeded 2000 MPN/100-mL. Concentrations in the four temporary well samples collected in the northern part of the study area (Figure 4) exceeded 1000 MPN/100-mL, while the two samples collected from the southern part

of the study area had lower concentrations (>200 and 31 MPN/100-mL). The median age of septic systems in the southern half of the study was 9 years, compared to a median age of 21 years in the northern half. Although lot sizes are somewhat larger in the southern portion of the study area, age of system appears to be an important factor affecting concentrations of coliform bacteria. The results suggest that many of the systems in the northern part of the study area are noncompliant.

Coliform bacteria were detected in 9 of the 46 domestic wells sampled at Site 3. A concentration of more than 200 MPN/100-mL was detected in one well, but concentrations in the remaining 8 wells were less than 100 MPN/100-mL. All nine detections were located in the northern part of the study area, where older systems occur (Figure 13). There were no correlations between concentrations of coliform bacteria and other sampled parameters, including well depth or depth to water.

There were two temporary well samples in which *E. coli* was detected at Site 3 (11 and 7 MPN/100-mL). These were from samples that had total coliform concentrations of more than 2000 MPN/100-mL. *E. coli* was not detected in any domestic well sample.



Figure 13: Locations where total coliform bacteria were detected in samples from domestic wells at Site 3. Contours show year in which the domestic wells were drilled.

Phosphorus

Median concentrations of phosphate were 0.03 mg/l at Site 1 and 0.02 mg/l at Sites 2 and 3. There were no differences in concentrations of phosphate between sites or between temporary and domestic well samples. There were no correlations between phosphate concentrations and concentrations of other sampled chemicals. There was no effect of date of well installation on phosphate concentrations. The maximum concentration was 0.17 mg/l and only five of 76 samples exceeded 0.05 mg/l. Phosphate does not appear to represent a concern in ground water under unsewered communities, but our data do not represent potential concerns of phosphate in individual plumes.

Other Inorganic Chemicals

Table 9 summarizes ground water quality information for temporary well samples at each of the three sites. Concentrations of chloride and nitrate were higher under unsewered land use than under undeveloped land use (MPCA, 1999b). Site 3 is characterized by high concentrations of chloride, ammonia, Kjeldahl nitrogen, and sulfate, and by high Eh. Site 3 also had high concentrations of coliform bacteria. Site 3 appears to be the most impacted of the three sites. This is probably due to the older age of the septic systems at Site 3. As plumes continue to expand with time, chemicals such as nitrate and sulfate accumulate to higher concentrations in ground water. The presence of reduced nitrogen and coliform bacteria indicate the presence of noncompliant systems, and these are more likely with older systems. Concentrations of ammonia at Site 3 (0.54 mg/L) may represent a concern for surface water, since the Class II Aquatic Life Standard for unionized ammonia is 0.040 mg/l.

Table 10 summarizes chemical concentrations in domestic wells at each of the three sites. Concentrations at Sites 1 and 2 are within the range of background concentrations for similar aquifers statewide (MPCA, 1998b). Impacts from ISTSs at Site 3 extend deeply into the aquifer as evidenced by the elevated concentrations of chloride and nitrate. Drinking water standards for boron were exceeded in four wells at Site 1. These elevated concentrations are probably associated with Cretaceous shale, which underlies the study area and has high concentrations of boron. Glacial deposits that comprise the aquifer at Site 1 likely contain weathered shale. The drinking water standard for nitrate was exceeded in only one domestic well. This occurred at Site 2.

There were, however, an additional 20 wells with concentrations of 5 mg/l or greater.

Fifteen of these were at Site 3.

Chemical	Site 1	Site 2	Site 3	Background
Alkalinity	-	164	-	241
Ammonia	0.085 ab	0.055 b	0.54 a	0.050
Boron	0.026	0.016	-	0.016
Calcium	89	59	-	66
Chloride	19.6 ab	12.6 b	41.6 a	1.85
Dissolved organic carbon	2.90	3.10	3.10	5.10
Dissolved oxygen	2.88	4.16	3.30	5.10
Eh	247 b	234 b	437 a	318
Iron	< 0.100	< 0.100	-	0.0055
Magnesium	26	11	-	23
Manganese	0.091	0.206	-	0.0050
Nitrate	4.10	6.42	5.92	0.70
No. samples	8	9	9	-
pH	7.00	7.29	6.74	7.31
Phosphorus	0.095	0.030	-	0.019
Potassium	1.63	1.15	-	1.09
Sodium	24.0	6.44	-	5.77
Specific conductance	628	439	389	427
Sulfate	6.65 a	2.68 b	6.91 ab	2.92
Total Kjeldahl nitrogen	0.29 b	0.31 b	1.52 a	0.200
Zinc	< 0.001	0.007	-	0.0075

Table 9: Summary information for temporary well samples at Sites 1, 2, and 3. Values represent median concentrations. Concentrations are in mg/l (ppm), except Eh (mV), pH, specific conductance (umhos/cm). Within a row, different letters indicate median concentrations that differ at a significance level of 0.05. Values shown in bold indicate concentrations that differed between undeveloped and unsewered land uses.

Chemical	Site 1	Site 2	Site 3	Background
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Alkalinity	246 a	-	210 b	238
Aluminum	< 0.050	-	< 0.050	0.0012
Ammonia	0.37 a	-	0.075 b	-
Boron	0.078 a	-	0.027 b	0.024
Cadmium	< 0.0010	-	< 0.0010	< 0.020
Calcium	63	-	66	74
Chloride	5.6 c	26 b	43 a	5.8
Chromium	< 0.0030	-	< 0.0030	0.55
Copper	< 0.0054	-	< 0.0054	6.3
Dissolved organic carbon	1.05	-	0.75	2.4
Dissolved oxygen	0.83 b	0.82 b	7.1 a	< 0.50
Eh	211 b	154 b	266 a	187
Iron	0.13 b	0.40 a	0.10 c	0.81
Lead	0.020	-	0.020	0.00018
Magnesium	23 b	-	27 a	22
Manganese	0.14 a	-	0.0018 b	0.18
Nickel	< 0.0061	-	< 0.0061	< 0.0061
Nitrate	< 0.50 b	< 0.50 b	3.5 a	< 0.50
pH	7.47	7.58	7.56	7.21
Phosphate	0.020	-	0.020	-
Phosphorus	0.02	-	-	0.056
Potassium	2.4 a	-	1.8 b	1.8
Sodium	9.5	-	13	5.0
Specific conductance	490 b	247 c	634 a	465
Sulfate	9.18 a	-	5.28 b	4.3
Temperature	9.9 b	9.3 c	10.7 a	8.8
Total Kjeldahl nitrogen	0.42	-	-	-
Zinc	0.0074 b	-	0.038 a	0.012

Table 10: Concentrations of chemicals in domestic wells at Sites 1, 2, and 3. Values represent median concentrations. Concentrations are in mg/l (ppm), except Eh (mV), pH, specific conductance (umhos/cm). Within a row, different letters indicate median concentrations that differ at a significance level of 0.05. Background concentrations are for water table aquifers from the statewide baseline study (MPCA, 1998b).

Cumulative and Comparative Risk

The Minnesota Pollution Control Agency is attempting to implement a multimedia approach to assessing environmental quality, with the objective of identifying environmental priorities. Two concepts that are part of this approach are cumulative risk and comparative risk. We define the cumulative risk to an individual (human or

ecological) as the sum of risks from different exposure pathways, such as drinking water, contact with soil, and inhalation of air. If we add the risks from each pathway, or media, we can identify the total risk and determine which media is most important. This priority setting is a form of comparative risk, that is, comparing the risk between different media. Within a particular media, we can also compare different types of human activity. For example, we can ask the question “What is the ground water risk associated with different land uses, such as unsewered residential, sewerred residential, and agricultural land use?”.

It is beyond the scope of this paper to present a comprehensive assessment of cumulative and comparative risk for the three study areas. Our data indicate that, in shallow ground water, the median concentration of nitrate is about 6 mg/l. This is 60 percent or 0.6 of the drinking water criteria. A risk value of 1.0 (100 percent) indicates the potential for adverse health impacts. Although “average” ground water under unsewered communities is below this risk value, septic systems contribute significantly to the risk value. Additionally, about 25 percent of shallow ground water exceeds the drinking water criteria for nitrate (risk value is greater than 1.0). There do not appear to be impacts from other chemicals, although we tested for a limited number of chemicals. Of the chemicals we sampled for, nitrate is the only chemical of concern in ground water under unsewered residential areas. Since nitrate affects the blood system, exposure to other chemicals that also affect the blood system can raise the risk value above the target level of 1.0. This exposure includes all media (air, water, soil). We conclude therefore, that in shallow ground water, septic systems have a sufficient impact on water quality to warrant additional analysis of other media or to implement management strategies to decrease exposure risk.

One management strategy is to drill deeper wells. At Sites 1 and 2, there was no increased risk from septic systems below depths of about 20 feet. At Site 3, drilling deeper wells will not decrease exposure risk. Additional risk analysis of other media is warranted or other management strategies must be implemented.

Although ground water under unsewered areas is impacted by nitrate, concentrations of nitrate are considerably less than those under irrigated agriculture (MPCA, 1999b), where the risk value is typically greater than 1.0 and often greater than 2.0. Nitrate concentrations under dryland agriculture are similar to those under unsewered residential communities with lot sizes of about 1 acre. Concentrations of nitrate under sewerred residential areas are much less than under unsewered residential areas. There may be concentrations of other chemicals, however, such as volatile organic compounds, that pose risks to drinking water receptors in sewerred areas (MPCA, 1999b). Treatment of municipal wastes in sewerred areas may also lead to surface water pollution, which may affect ecological receptors or diminish the aesthetic value of the surface water. Consequently, it is important for communities dealing with the question of development and whether or not to sewer to compare potential impacts from different land uses, of different management strategies, and from different media.

Summary

Nitrate represents the chemical of greatest concern in ground water under unsewered developments. Median nitrate concentrations in shallow ground water under three unsewered developments fell within a range of 4.1 to 6.4 mg/l. The probability of exceeding the drinking water standard of 10 mg/l was about 25 percent. Nitrate

concentrations decreased abruptly within 15 feet of the top of the aquifer at Sites 1 and 2, probably as a result of denitrification. Concentrations did not decrease with depth in the sand and gravel aquifer at Site 3, and no nitrate-reducing zone was observed. Nitrate concentrations were correlated with age of development. This is primarily due to the gradual buildup of nitrate as plumes expand in the aquifers underlying the study areas, but may also be related to the performance of individual septic systems.

Phosphorus concentrations were very low and were less than 0.050 mg/l in most samples. Concentrations of other inorganic chemicals were low throughout the aquifers underlying the study areas, with the exception of chloride. Concentrations in domestic wells were below drinking water standards except for nitrate in a single well at Site 2.

Coliform bacteria were present in shallow ground water samples. Concentrations were highest at Site 3 and exceeded 2000 MPN/100-ml under older parts of the study area. Coliform bacteria were detected in about 30 percent of domestic wells sampled at Sites 1 and 3, but concentrations were typically less than 10 MPN/100-ml.

The upper part of the aquifer at Site 3 is impacted by ISTSs as evidenced by elevated concentrations of coliform bacteria, ammonia, and Kjeldahl nitrogen compared to the other sites. The presence of these chemicals at higher concentrations suggests that noncompliant systems are present. Site 3 was the oldest development, with a median age of 20 years for wells drilled in the area.

Our results for nitrate are similar to those observed in other research throughout the country. There is little information in the literature on the distribution of other chemicals and pathogens in ground water under unsewered developments. While ISTSs impact shallow ground water, two management strategies can minimize these impacts.

First, regular maintenance and inspection of septic systems ensures they are functioning properly, thus minimizing the likelihood of contamination with pathogens. Second, communities must find the proper balance between lot size and well location to minimize impacts from nitrates. Some aquifers, such as the one at Site 3, are more vulnerable to contamination and may require larger lot sizes and more stringent controls on maintenance of ISTSs.

While nitrate and pathogens appear to be the primary concerns for drinking water, we cannot ignore impacts of phosphorus from noncompliant systems that occur in close proximity to surface waters. Estimating impacts from phosphorus requires studies of individual plumes. Currently, we are trying to find appropriate locations for conducting these plume studies.

References

- Alhajjar, B.J., G. Chesters, and J.M. Harkin. 1990. *Indicators of Chemical Pollution from Septic Systems*. Ground Water. 28:559-568.
- Alhajjar, B.J., J.M. Harkin, and G. Chesters. 1989. *Detergent Formula Effect on Transport of Nutrients to Ground Water from Septic Systems*. Ground water. 2:209-219.
- Anderson, D.L., J.M. Rice, M.L. Voorhees, R.A. Kirkner, and K.M. Sherman. 1987. *Ground Water Modeling with Uncertainty Analysis to assess the Contamination Potential from Onsite sewage Disposal Systems (OSDS) in Florida*. Proceedings from the National Symposium on Individual and Small Community Sewage Systems, December 14-15. Chicago, IL. 411 pp.
- Bauman, B.J. and W.M. Schafer. 1984. *Estimating ground-water Quality Impacts from On-site Sewage Treatment Systems*. Proceedings of the Fourth National Home Sewage Treatment Symposium. ASAE pp. 285-294.
- Brown, K.W. 1980. *An Assessment of Impacts of Septic Leachfields, Home Lawn Fertilization, and Agricultural activities on Ground water Quality*. 104 p.
- DeBorde, D.C., W.W. Woessner, B. Lauerman, and P.N. Ball. 1998. *Virus Occurrence and Transport in a School Septic System and Unconfined Aquifer*. Ground Water 36:825-834
- Fay, S.R., R.C. Spong, S.C. Alexander, and E.C. Alexander Jr. 1995. *Optical Brighteners: Sorption Behavior, Detection, Septic System Tracer Applications*. Published in: Proceedings of the International Association of Hydrogeologists XXVI International Congress, Edmonton, Alberta, Canada.
- Geron, C.A., T.K. Danneberger, S.J. Traina, T.J. Logan, and J.R. Street. 1993. *The Effects of Establishment Methods and Fertilization Practices on Nitrate Leaching from Turfgrass*. Jour. Environ. Qual. 22:119-125.
- Hagedorn, C., D.T. Hansen, and G.H. Simonson. 1978. *Survival and Movement of Fecal Indicator bacteria in Soil under Conditions of Saturated Flow*. Jour. Environ. Qual. 7:55-59.
- Hantzsche, N.N., and E.J. Finnemore. 1992. *Predicting Ground-Water Nitrate-Nitrogen Impacts*. Groundwater. 30:490-499.
- Harman, J., W.D. Robertson, J.A. Cherry, and L. Zanini. 1996. *Impacts on a sand aquifer from an old septic system: nitrate and phosphate*. Ground Water 34:1105-1114.

- Harmsen, E.W., J.C. Converse, B.H. Shaw, E.J. Tyler, and J.O. Peterson. 1996. *Spatial and Temporal Distribution of Groundwater Nitrate-N beneath Two Unsewered Subdivisions in the Central Wisconsin Sand Plain*.
- Jemison, J.M. Jr., and R.H. Fox. 1994. *Nitrate Leaching from Nitrogen-Fertilized and Manured Corn Measured with Zero-Tension Pan Lysimeters*. Jour. Environ. Qual. 23:337-343.
- Kaplan, B.O. 1987. *Septic Systems Handbook*. Lewis Publishers. Chelsea, Michigan 290 p.
- Miller, J.C. 1975. *Nitrate Contamination of the Water-table Aquifer by Septic Tank Systems in the Coastal Plain of Delaware*. P. 121-133. In W.S. Jewell and R. Swan (ed.) *Water Pollution Control in Low Density Areas*. Proc. Rural Environ. Eng. Conf., Hanover, N.H. September, 1973. Univ. Press of New England, Hanover, N.H.
- Minnesota Geological Survey. 1990. *Geologic Atlas Washington County, Minnesota*. County Atlas Series C5- 7 plates.
- Minnesota Pollution Control Agency. 1998a. *Water Quality in the Upper Fifteen Feet of a Shallow Sand Aquifer in a Variable Land Use Setting - St. Cloud Area, Minnesota*. St. Paul, MN 52 p.
- Minnesota Pollution Control Agency. 1998b. *Baseline Water Quality of Minnesota's Principal Aquifers*. St. Paul, MN. 145 p. and appendices..
- Minnesota Pollution Control Agency. 1998c. *Data Analysis Protocol for the Ground Water Monitoring and Assessment Program (GWMAP)*. Draft in review.
- Minnesota Pollution Control Agency. 1999a. *Effects of Septic Systems on Ground Water Quality in Baxter, Minnesota*. St. Paul, MN 38 p.
- Minnesota Pollution Control Agency. 1999b. *Effects of Land Use on Ground Water Quality, St. Cloud Area, Minnesota, 1998 Results*. St. Paul, MN. 46 p.
- Myer, G.N., and A.R. Knaeble. 1995. *Surficial Geology, Plate 3*. In: *Geologic Atlas, Stearns County, Minnesota*. Minnesota Geological Survey, County Atlas Series C-10, Part A. St. Paul, MN.
- Newman, M.C., K.D. Greene, and P.M. Dixon. 1995. *Uncensor 4.0*. Savannah River Ecology Laboratory. 91p.

- Noss, R.R., and M. Billa. 1988. *Septic System Maintenance Management*. Journal of Urban Planning and Development. 114:73-90.
- Owens, L.B., W.M. Edwards, and R.W. Van Keuren. 1994. *Groundwater Nitrate Levels under Fertilized Grass and Grass-Legume Pastures*. Jour. Environ. Qual. 23:752-758.
- Quan, E.L., H.R. Sweet, and J.R. Illian. 1974. *Subsurface Sewage Disposal and Contamination of Ground Water in East Portland, Oregon*. Groundwater 12:356-366.
- Randall, G.W., and T.K. Iragavarapu. 1995. *Impact of Long-Term Tillage Systems for Continuous Corn on Nitrate Leaching to Tile Drainage*. Jour. Environ. Qual. 24:360-366.
- Rea, R.A., and S.B. Upchurch. 1980. *Influence of Regolith Properties on Migration of Septic Tank Effluent*. Groundwater 118-1125.
- Reneau, R.B. Jr. 1978. *Influence of Artificial drainage on Penetration of Coliform Bacteria from Septic Tank Effluents into Wet Title Drained Soils*. Jour. Environ. Qual. 7:23-30.
- Reneau, R.B. Jr. 1979. *Changes in Concentration of Selected Chemical Parameters in wet, Tile-drained Soil Systems as Influenced by disposal of septic Tank Effluents*. Jour. Environ. Qual. 8:189-195.
- Reneau, R.B. Jr., and D.E. Pettry. 1976. *Phosphorus Distribution from Septic Tank Effluent in Coastal Plain Soils*. Jour. Environ. Qual. 5:34-39.
- Robertson, W.D. 1994. *Chemical Fate and Transport in a Domestic Septic System: Site Description and Attenuation of Dichlorobenzene*. Environ. Toxicol. Anal. Chem. 13:183-191.
- Robertson, W.D., and D.W. Blowes. 1995 *Major Ion and Trace Metal Geochemistry of an Acidic Septic-System Plume in Silt*. Groundwater. 33:275-283
- Robertson, W.D., and J.A. Cherry. 1995. *In Situ Denitrification of Septic-System Nitrate using Reactive Porous Media Barriers: Field Trials*. Groundwater. 33:99-111.
- Robertson, W.D, J.A. Cherry, and E.A. Sudicky. 1991. *Ground-water contamination from two small septic systems on sand aquifers*. Ground Water. 29:82-92.
- Robertson, W.D., S.L. Schiff, and C.J. Ptacek. 1998. *Review of Phosphate Mobility and Persistence in 10 Septic System Plumes*. Ground Water. 36:1000-1010.

- Steinhemmer, T.R., K.D. Scoggin, and L.A. Kramer. 1998. *Agricultural Chemical Movement through a Field-Size Watershed in Iowa: Subsurface Hydrology and Distribution of Nitrate in Ground Water*. Environ. Sci. Technol. 32:1039-1047.
- Tinker, J.R. Jr. *Impact of Nitrate-Nitrogen from Unsewered Subdivisions on Groundwater*.
- Walker, W.G., J. Bouma, D.R. Keeney, and F.R. Magdoff. 1973a. *Nitrogen Transformations during Subsurface Disposal of Septic Tank Effluent in sands: I Soil Transformations*. Jour. Environ. Qual. 2:475-480.
- Walker, W.G., J. Bouma, D.R. Keeney, and P.G. Olcott. 1973b. *Nitrogen Transformations during Subsurface Disposal of Septic Tank Effluent in Sands: II. Ground Water Quality*. Jour. Environ. Qual. 4:521-525.
- Wilhelm, S.R., S.L. Schiff, and W.D. Robertson. 1994. *Chemical Fate and transport in a Domestic Septic System: Unsaturated and Saturated Zone Geochemistry*. Environ. Toxic.Chem. 13:193-203
- Wilhelm, S.R., S.L. Schiff, and W.D. Robertson. 1996. *Biogeochemical Evolution of Domestic waste Water in Septic Systems: 2. Application of Conceptual Model in sandy Aquifers*. Groundwater 34:853-864.
- United States Department of Agriculture. *Soil Survey of Crow Wing County Minnesota*.
- United States Department of Agriculture. 1985. *Soil Survey of Stearns County Minnesota*. 272 p. plus maps.
- United States Department of Agriculture. *Soil Survey of Washington County Minnesota*.
- Yates, M.V. and S.R. Yates. 1988. *Septic Tank Setback Distances : A Way to Minimize Virus Contamination of Drinking Water*. Ground Water 202-208.