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REPORT OF INVESTIGATIONS 117

WATER QUALITY NEAR WASTEWATER TREATMENT SYSTEMS IN ALLUVIAL AND KARST HYDROGEOLOGIC SETTINGS, BLACK HILLS, SOUTH DAKOTA

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GEOLOGICAL SURVEY PROGRAM

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ABSTRACT

Water quality was investigated near wastewater treatment systems in alluvial and karst hydrogeologic settings in the Black Hills of western South Dakota. The primary goal of the study was to provide information regarding the potential effects of wastewater treatment systems on ground-water quality within specific hydrogeologic settings. Three residential on-site wastewater treatment systems, the wastewater treatment lagoons at Hill City, South Dakota, and sources of recharge and discharge for the Madison aquifer were included in this investigation. Parameters that were evaluated as indicators of effects from wastewater treatment systems included nitrate-nitrogen, ammonia, phosphorus, total Kjeldahl nitrogen, fecal coliform bacteria, *Escherichia coli* bacteria, chloride, conductivity, and caffeine. An effort was made to investigate properly maintained on-site wastewater treatment systems with no known functional problems.

Results of ground-water quality analyses near the three individual on-site wastewater treatment systems that were investigated included nitrate-nitrogen concentrations from less than 0.1 to 4.2 milligrams per liter, ammonia concentrations from less than 0.02 to 0.25 milligrams per liter, total phosphorus concentrations from less than 0.083 to 10.0 milligrams per liter, total Kjeldahl nitrogen concentrations from less than 0.11 to 0.92 milligrams per liter, chloride concentrations from less than 3 to 187 milligrams per liter, and conductivity values from 281 to 1,070 microsiemens. *Escherichia coli* bacteria were detected in ground water at all three of the study sites. Caffeine was not detected at any site. Detections of bacteria were in water samples from monitoring wells that were in very close proximity to the on-site wastewater treatment system drain fields where drinking-water wells could not be legally installed due to set-back requirements. None of the other parameters that were analyzed near individual on-site wastewater treatment systems exceeded drinking-water standards established by the U.S. Environmental Protection Agency and the South Dakota Department of Environment and Natural Resources.

Water-quality analyses from ground water and spring flow from the Madison aquifer included nitrate-nitrogen concentrations from less than 0.01 to 1.2 milligrams per liter, total phosphorus concentrations from 0.007 to 0.096 milligrams per liter, total Kjeldahl nitrogen concentrations from less than 0.11 to 0.28 milligrams per liter, and chloride concentrations from less than 3 to 18 milligrams per liter. Conductivity values ranged from 251 to 601 microsiemens, and *Escherichia coli* bacteria were present in samples from two of the sampled public water-supply wells. Ammonia and caffeine were below laboratory detection limits or absent in all samples from the Madison aquifer.

Results of ground-water quality analyses near the wastewater treatment lagoons at Hill City included nitrate-nitrogen concentrations from less than 0.1 to 10.1 milligrams per liter, ammonia concentrations from less than 0.02 to 10.0 milligrams per liter, total phosphorus concentrations from 0.054 to 1.96 milligrams per liter, total Kjeldahl nitrogen concentrations from 0.37 to 12.6 milligrams per liter, chloride concentrations from 17 to 89 milligrams per liter, and conductivity values from 373 to 1,156 microsiemens. *Escherichia coli* and fecal coliform bacteria were each detected one time during the investigation, and caffeine was not detected in ground-water samples at the wastewater treatment lagoons.

ACKNOWLEDGEMENTS

The author would like to thank all of the people who contributed to this project including coworkers within the South Dakota Department of Environment and Natural Resources, colleagues with other organizations, and property owners and public water-supply operators who allowed monitoring well installation on their properties or granted permission to sample their wells. In particular, Derric L. Iles and Arden D. Davis provided extensive input and assistance with all phases of the project from the design of the initial approach through final review of published figures and text. Funding for the investigation was provided by the Geological Survey Program and the Water Resources Assistance Program, South Dakota Department of Environment and Natural Resources.

INTRODUCTION

On-site wastewater treatment systems, commonly referred to as septic systems, currently serve approximately 23 percent of the existing population in the United States, and approximately 37 percent of new development in the United States will be served by on-site wastewater treatment systems (U.S. Environmental Protection Agency, 1997a, 2002a). Although frequently viewed in the past as temporary facilities until a centralized approach could be implemented, on-site wastewater treatment systems are increasingly used as permanent solutions to decentralized sewage treatment, particularly in areas affected by factors such as low-density development, limited water supplies, and rugged topography (Siegrist, 2001b). More than half of the existing on-site wastewater treatment systems in the United States are greater than 30 years old, and estimates of functional failure rates range from 10 to 20 percent (Hogye and others, 2001; U.S. Environmental Protection Agency, 2002b). With increased proliferation of on-site wastewater treatment systems, there also have been increasing concerns regarding the potential effects of these facilities on water quality. On-site wastewater treatment systems are frequently identified as a primary suspected contributor to elevated concentrations of constituents of concern such as nitrate-nitrogen, phosphorus, and pathogenic microorganisms (Davis, 1979; Bad Moccasin, 1986; Clark and others, 2001; Cliver, 2001; Geary and Whitehead, 2001; Johnson and others, 2001).

In the Black Hills of western South Dakota, increasing residential development in unsewered areas is raising similar concerns regarding the potential detrimental effects of on-site wastewater treatment systems on water quality. Figure 1 shows approximately 9,000 identified on-site wastewater treatment systems located on or up-gradient from major aquifer recharge areas in the Black Hills (South Dakota Department of Environment and Natural Resources, 2001). Opinions range from the viewpoint that increasing numbers of on-site wastewater treatment systems are of little or no concern because outbreaks of disease have not been prevalent, to the viewpoint that on-site wastewater treatment systems already significantly affect the quality of water resources in the Black Hills and other areas.

Local management of on-site wastewater treatment systems in the Black Hills region, like other areas of the United States, is typically performed by county and city governments which rely either on state regulations or more stringent local ordinances. Chapters 74:53:01 and

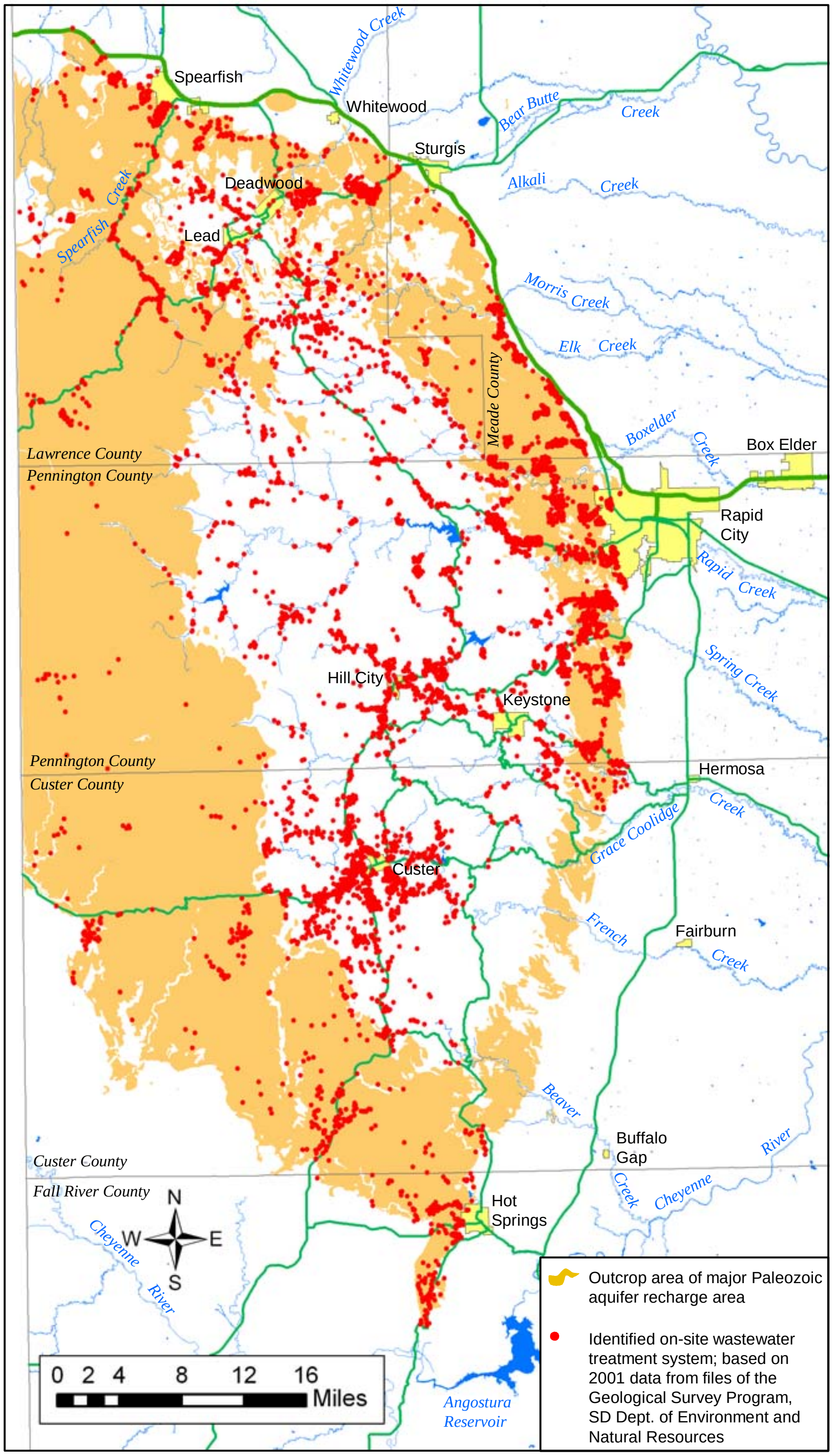


Figure 1. Map showing identified on-site wastewater treatment systems located on or up-gradient from major aquifer recharge areas in the Black Hills.

74:53:02 of the Administrative Rules of South Dakota which govern on-site wastewater treatment systems are applicable statewide and are intended to provide treatment of wastewater and protection for the environment and public health. Similar to surrounding states, South Dakota's regulations are based on criteria such as system design, soil percolation tests, and vertical and horizontal separation distances from ground water, bedrock, wells, and other features and boundaries. However, these "one size fits all" regulations often are called into question as being either overly protective or not protective enough of ground water in certain circumstances, and decisions regarding development issues in sensitive aquifer recharge areas often are made in the absence of specific scientific data related to water quality.

Decision makers at the county and city level in the Black Hills have expressed increased interest in learning more about the sensitivity of aquifers used by local public water-supply systems, particularly with respect to the potential effects of on-site wastewater treatment systems. County commissions frequently raise questions regarding the maximum number of on-site wastewater treatment systems that an area can sustain, and they also have expressed a desire for understandable scientific standards by which to evaluate requests for new housing developments that are outside of established sanitary districts. Unfortunately, there are no simple answers to these questions. Geologic, hydrologic, and soil conditions vary greatly in the Black Hills, and standards that might be suitable for one area might not be appropriate for another area. Further complicating the issue, on-site wastewater treatment systems vary greatly in design, age, efficiency, and maintenance practices. Additionally, systems that were installed prior to the existence of regulations may not meet current requirements.

Overall, the situation is one of increasing residential development and environmental pressure, extremely heterogeneous hydrogeologic conditions, variable government management of existing on-site wastewater treatment systems, variable owner maintenance of such systems, and a lack of scientific information regarding the water-quality effects of on-site wastewater treatment systems. Furthermore, homeowners and developers generally resist any increase in expense or any increase in governmental intervention, particularly when faced with a paucity of scientific data with regard to this issue. Therefore, one objective of this project was to examine the performance of on-site wastewater treatment systems in the field to begin to evaluate the effects of system effluent on ground-water resources in the Black Hills area. Another objective was to investigate potential effects on ground-water quality from the wastewater treatment lagoons at Hill City, South Dakota.

Because of the hydrogeologic heterogeneity of the Black Hills, this work attempted to evaluate the effects of on-site wastewater treatment systems with respect to specifically selected hydrogeologic settings in the Black Hills region. The initial approach was to examine several on-site wastewater treatment systems located within relatively sensitive aquifer recharge areas. This was in lieu of studying all possible permutations of system design and hydrogeologic settings. An attempt was made to investigate properly maintained on-site wastewater treatment systems, with no known functional problems, to provide information to decision makers regarding the number of on-site wastewater treatment systems that specific hydrogeologic settings can sustain. Results of this initial study then can be used as a basis for further research that is needed to more completely characterize effluent effects and behavior within various hydrogeologic settings. This investigation focused primarily on investigating and understanding the potential effects from individual on-site wastewater treatment systems, and it did not attempt to quantify the effects of large numbers of densely spaced on-site wastewater treatment systems on ground-water quality. This investigation also was limited to select water-quality parameters that serve as indicators of potential effects from wastewater treatment systems, and it does not fully characterize the effects, or lack of effects, from potential contaminants that were not investigated.

Two of the most important hydrogeologic settings in the Black Hills area are unconsolidated alluvial deposits and karst limestone formations. Alluvial deposits often are chosen as development sites because of their accessibility, relatively gentle topography, scenic beauty, fertility, and shallow ground-water resources. However, the sands and gravels composing the alluvial deposits frequently serve both as a source of drinking water for dwellings and as the receiving environment for wastewater effluent from the dwellings. Furthermore, the shallow, unconfined water table within alluvial deposits, and the shallow depths of drinking-water wells in these deposits (often less than 30 feet), may not afford appropriate treatment of the contaminants in wastewater introduced by on-site wastewater treatment systems. Therefore, areas underlain by alluvial deposits were one of the hydrogeologic settings that were selected for evaluation in this study.

Another highly significant hydrogeologic setting is that of the karst limestone formations (including the Madison aquifer) which provide water to cities, towns, home owner associations, and private residences on the flanks of the Black Hills uplift. These highly productive aquifers provide the largest single source of drinking water in the Black Hills; however, because of their karstic characteristics, recharge areas for these aquifers are highly sensitive to contamination, and the situation poses difficult challenges to drinking-water protection efforts. Therefore, an attempt also was made to evaluate the potential effects of on-site wastewater treatment systems on water-supply systems that rely on karst limestone aquifers.

LOCATION OF STUDY AREA

The study area was located in the central and eastern regions of the Black Hills of western South Dakota (fig. 2) because of the proliferation of on-site wastewater treatment systems and the occurrence of large aquifer recharge areas in that vicinity. Specific alluvial hydrogeologic settings were targeted within three major drainage basins in the Black Hills, including the Rapid Creek, Spring Creek, and French Creek watersheds. Two of the alluvial sites were located immediately adjacent to perennial streams and one was located within a smaller tributary valley with no perennial streamflow and no nearby surface-water bodies. Many drinking-water systems that use the Madison aquifer are located along the eastern flank of the Black Hills uplift; therefore, sampling localities also were selected in that area. To evaluate water quality of surface water entering the recharge areas of the Madison aquifer, several sample localities were located along perennial streams just upstream of exposures of the Pahasapa (Madison) Limestone. The municipal wastewater treatment lagoons at the city of Hill City, South Dakota, also were included in this study because of their hydrogeologic setting, potential effects to ground-water and surface-water quality, and information they can yield on this topic (fig. 2).

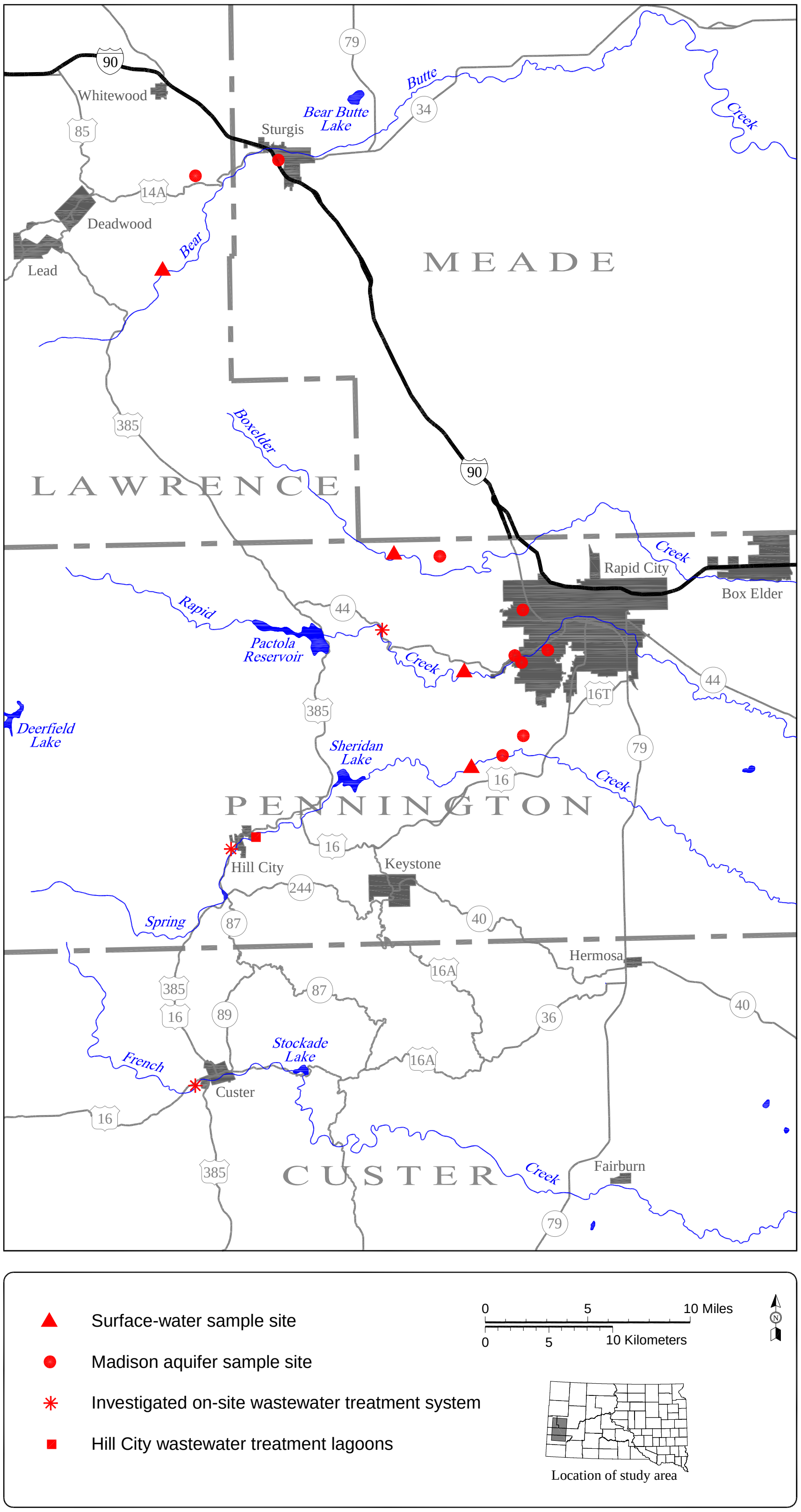


Figure 2. Map showing location of study area.

PREVIOUS WORK

There is a considerable knowledge base with respect to on-site wastewater treatment system design, implementation, and performance; however, understanding of the science and engineering of these systems lags behind that of centralized systems, and significant gaps still remain in understanding the effects of these systems. Furthermore, the current knowledge base does not support system design well enough to consistently and reliably achieve specific performance goals. Therefore, it is still difficult to account for many of the relevant influences on system performance including design, siting, usage, installation, operation, and environmental factors (Siegrist, 2001a).

A national conference on research priorities pertaining to on-site wastewater treatment systems identified 61 significant topics of research. Conference participants also concluded that quantitative analyses of long-term treatment performance, formal assessment of risks, and selection of appropriate management actions are difficult at present (Electric Power Research Institute, 2001).

Information on potential effects to drinking-water aquifers is even sketchier, and study of the topic is complicated by a host of variables ranging from hydrologic and geologic heterogeneity to changing environmental factors to system design, age, and condition. In the Black Hills area, previous studies have generally focused on relatively localized issues such as contamination within a subdivision, community, or with respect to a particular public water-supply system (South Dakota Department of Health, 1969; Davis, 1979, 1983; Coker, 1981; Hafi, 1983; Bad Moccasin, 1986).

Siegrist and others (2001) described and summarized current research needs pertaining to the design and performance of on-site wastewater soil absorption systems in the United States. Although an absence of documented adverse effects led these authors to conclude that system design and performance are usually satisfactory, they also concluded that there is a lack of understanding and predictability of performance with respect to design, installation, operation, environmental factors, and the risks and effects of inadequate system function. This lack of understanding and predictability is attributed primarily to the complex relationships between purification and hydraulic processes and the factors that control their behavior.

The following four areas of high-priority research needs were identified by these authors: “(1) fundamental understanding of clogging zone genesis and unsaturated zone dynamics and their effects on treatment efficiency, particularly for pathogens, (2) development of modeling tools for predicting wastewater soil absorption system function and performance as affected by design and environmental conditions, (3) identification of indicators of performance and methods of cost-effective monitoring, and (4) development of valid accelerated testing methods for evaluating long-term wastewater soil absorption system performance.” Siegrist and others (2001) further specified that at scales encompassing communities and watersheds, there is a pressing need for information regarding the effects of on-site wastewater treatment systems on ground-water and surface-water quality. They suggested that information at this scale is critical for questions pertaining to minimum lot sizes, determination of setback distances, discrimination of on-site wastewater treatment system contributions of pathogens and nutrients to receiving waters, and for providing information for total maximum daily load studies, modeling, and decision-support tools for water management.

Cliver (2001) summarized current research needs with respect to the fate and transport of pathogens associated with on-site wastewater treatment systems. The most crucial research needs involved comprehension of the distribution and infiltration of pathogens into the soil treatment medium, and the necessity to further understand the retention or inactivation of pathogens in the unsaturated zone of the soil treatment medium.

In a similar synopsis pertaining to research needs for nutrients associated with on-site wastewater treatment systems, Gold and Sims (2001) recommended increased emphasis on micro-scale analysis of the fate and transport of nutrients from individual on-site wastewater treatment systems, and watershed scale analysis to estimate nutrient loading for use in developing total maximum daily load values and source-specific pollution control strategies. These authors also encouraged the development of risk categorization models or site indexing approaches to classify and describe the vulnerability of aquifers and watersheds to nutrient losses from on-site wastewater treatment systems.

Numerous authors have published results of studies regarding specific contaminants found in association with on-site wastewater treatment systems. Nitrate-nitrogen has been identified as a significant potential contaminant in many investigations, and the effects and behavior of nitrate-nitrogen in ground water have been documented for many specific localities (Yates, 1985; Ayers Associates, 1991; Hantzsche and Finnemore, 1992; Shaw and Turyk, 1994; Robertson and Cherry, 1995; Minnesota Pollution Control Agency, 1999; U.S. Environmental Protection Agency, 2002b; Taylor, 2003). Bacteria, viruses, protozoa, and parasites all have been investigated with respect to survival rates and travel distances (McGauhey and Krone, 1967; Bouma and others, 1972; Gerba and others, 1975; Wellings and others, 1975; Schaub and Sorber, 1977; Vaughn and Landry, 1977; Yeager and O’Brien, 1977; Hain and O’Brien, 1979; Harkin and others, 1979; Vaughn and Landry, 1980; Vaughn and others, 1981; Hagedorn, 1982; Kowal, 1982; Vaughn and others, 1982, 1983; Bicki and others, 1984; Pekdeger, 1984; Cantor and Knox, 1985; Jansons and others, 1989; Yates and Yates, 1989; Anderson and others, 1991; Tchobanoglous and Burton, 1991; Ayers Associates, 1993; Anderson and others, 1994; Higgins and others, 2000; U.S. Environmental Protection Agency, 2002b). Treatment of phosphorous by on-site wastewater treatment systems and subsequent migration of phosphorous in soil and ground water also have been examined in a number of studies (Sikora and Corey, 1976; University of Wisconsin, 1978; Rezek and Cooper, 1980; U.S. Environmental Protection Agency, 1984a, 1984b, 2002a).

Little research has been performed pertaining to toxic organic compounds in wastewater effluent; however, some information on migration and retention of toxic organic compounds is available (Dagan and Bresler, 1984; DeWalle and others, 1985; Hillel, 1989; Preslo and others, 1989; Wilhelm, 1998; U.S. Environmental Protection Agency, 2002b). The fate and transport of metals in ground water also are not well documented, although a number of metals in wastewater effluent, and limited information on their migration and retention, have been reported in the literature (Feige and others, 1975; Bennett and others, 1977; Segall and others, 1979; Whelan and Titmanis, 1982; Cantor and Knox, 1985; Ayers Associates, 1991; Evanko and Dzombak, 1997; Lim and others, 2001; U.S. Environmental Protection Agency, 2002b). Reports concerning total suspended solids, biochemical oxygen demand, and dissolved inorganic constituents also have been published (University of Wisconsin, 1978; Anderson and others, 1994; U.S. Environmental Protection Agency, 2002b).

A number of publications in the literature pertaining to on-site wastewater treatment systems have focused on evaluating and predicting the effects of nitrate-nitrogen on ground-water quality, usually from high-density residential developments, through the use of mathematical mass-balance models (Wehrmann, 1984; Bauman and Schafer, 1985; Center for Environmental Research, 1985; Tinker, 1991; Hantzsche and Finnemore, 1992; Taylor, 2003). Models by Wehrmann (1984) and Bauman and Schafer (1985) allowed consideration of nitrogen contributions from lateral ground-water flow. Tinker (1991) combined the approach of Wehrmann with that of the BURBS model developed by the Center for Environmental Research (1985) to include nitrogen contributions from lawn fertilizer. The method of Hantzsche and Finnemore (1992) is a simplified approach that assumes the only source of nitrate-nitrogen is from wastewater effluent and that nitrate-nitrogen is reduced strictly by dilution from recharge resulting from infiltration of precipitation and from on-site wastewater treatment systems. Taylor (2003) presented an approach in which background nitrogen concentrations can be considered in addition to nitrogen from lawn fertilizer and on-site wastewater treatment systems. All of these authors stress that the mass-balance approach is more accurate when applied to large scale housing developments over long-term, steady-state conditions, rather than attempting to predict nitrate-nitrogen concentrations at a down-gradient location from an individual on-site wastewater treatment system.

Although much published literature exists pertaining to a variety of performance factors for on-site wastewater treatment systems, relatively few studies have been completed on the variable effects of on-site wastewater treatment system effluent within different hydrogeologic settings. One such study in the Turkey Creek watershed in central Colorado attempted to address this issue through an improved conceptual understanding of the ground-water system within the watershed (Jefferson County, Colorado, 2000). In that study, hydrogeologic characteristics such as rock type, fracture characteristics, storage capacity, and permeability were used to delineate unique areas called “hydrologic response units,” which then were used to calculate quantities of stored and available ground water. After hydrologic response units were delineated, they were combined with other factors such as slope, soils, vegetation, and precipitation for modeling purposes. Development and incorporation of an appropriate conceptual model characterizing the hydrologic system within the watershed was critical to the understanding of the resulting data in this approach (Jefferson County, Colorado, 2000).

Several previous studies completed in the Black Hills area have provided information useful in the study of on-site wastewater treatment systems. Many of these studies utilized approaches that included terms such as aquifer sensitivity, aquifer susceptibility, and aquifer vulnerability. In the hydrologic literature, the terms aquifer sensitivity and aquifer susceptibility usually are used to characterize intrinsic aquifer features, such as rock type, transmissivity rates, fractures, or karst features, that affect the ability of water and potential contaminants to move through an aquifer. More specifically, Focazio and others (2002) defined the intrinsic susceptibility of a ground-water system as the measure of ease with which water enters and moves through an aquifer. Aquifer vulnerability is generally used to refer to the risk of contamination from potential pollution sources, and was defined by the National Research Council (1993) as the potential or likelihood for contaminants to reach a specified position in the ground-water system after introduction at some location above the uppermost aquifer. In the following paragraphs, the terms aquifer sensitivity, aquifer susceptibility, and aquifer vulnerability are used as they appeared in each referenced publication.

The South Dakota Department of Health (1969) published a report on pollution within the Spring Creek drainage, and determined that there were significant problems with bacteria, nitrate-nitrogen, phosphate, siltation, and biologic oxygen demand at that time. Although the largest single source of these contaminants was attributed to municipal wastewater discharged directly into Spring Creek by the city of Hill City, a practice that has been discontinued, on-site wastewater treatment systems and cattle grazing within the watershed collectively were determined to be an equal if not greater source of pollutants to the creek.

Putnam (2000) published information on sensitivity of ground water to contamination in Lawrence County, South Dakota, based primarily on the DRASTIC approach (Aller and others, 1987) and prior work in the Rapid Creek watershed by Davis and others (1994). Davis and others (2000) published a modification of the DRASTIC approach, known as KARSTIC, which included additional provisions for karst terrain and which can be used for general ground-water sensitivity characterizations needed for land management and planning decisions in karst settings. Wiles (1992) studied infiltration patterns in caves in the Black Hills, and Long and Putnam (2002) presented a conceptual model of flow within the Madison and Minnelusa aquifers in the Rapid City area. Several ground-water tracing investigations of the Madison aquifer in the Black Hills also have contributed to understanding of anisotropic conditions, ground-water flow paths and travel times, and sensitivity of ground-water resources in this karst aquifer (Rahn, 1971; Rahn and Gries, 1973; Greene, 1997, 1999; L. Putnam and A. Long, Rapid City, S. Dak., U.S. Geological Survey, oral commun., 2007).

Hargrave (2005) presented susceptibility and vulnerability maps for the Minnelusa aquifer within the Rapid City West 7.5 minute quadrangle. Hargrave used hydraulic properties of the aquifer, including factors affecting permeability and porosity such as fracturing and karst features, to derive the susceptibility map. The vulnerability map then was generated by overlaying potential contaminants, including on-site wastewater treatment systems and transportation routes, onto the susceptibility rankings. Miller (2005) completed similar susceptibility and vulnerability maps for the Madison aquifer for an area covering portions of five 7.5 minute topographic quadrangles west of Rapid City, South Dakota, and concluded that the Madison aquifer is inherently highly susceptible to contamination and locally it is highly vulnerable to contamination. Miller (2005) also evaluated the influence of large geologic structures and stratigraphic characteristics on specific ground-water flow paths in the Madison aquifer west of Rapid City, South Dakota.

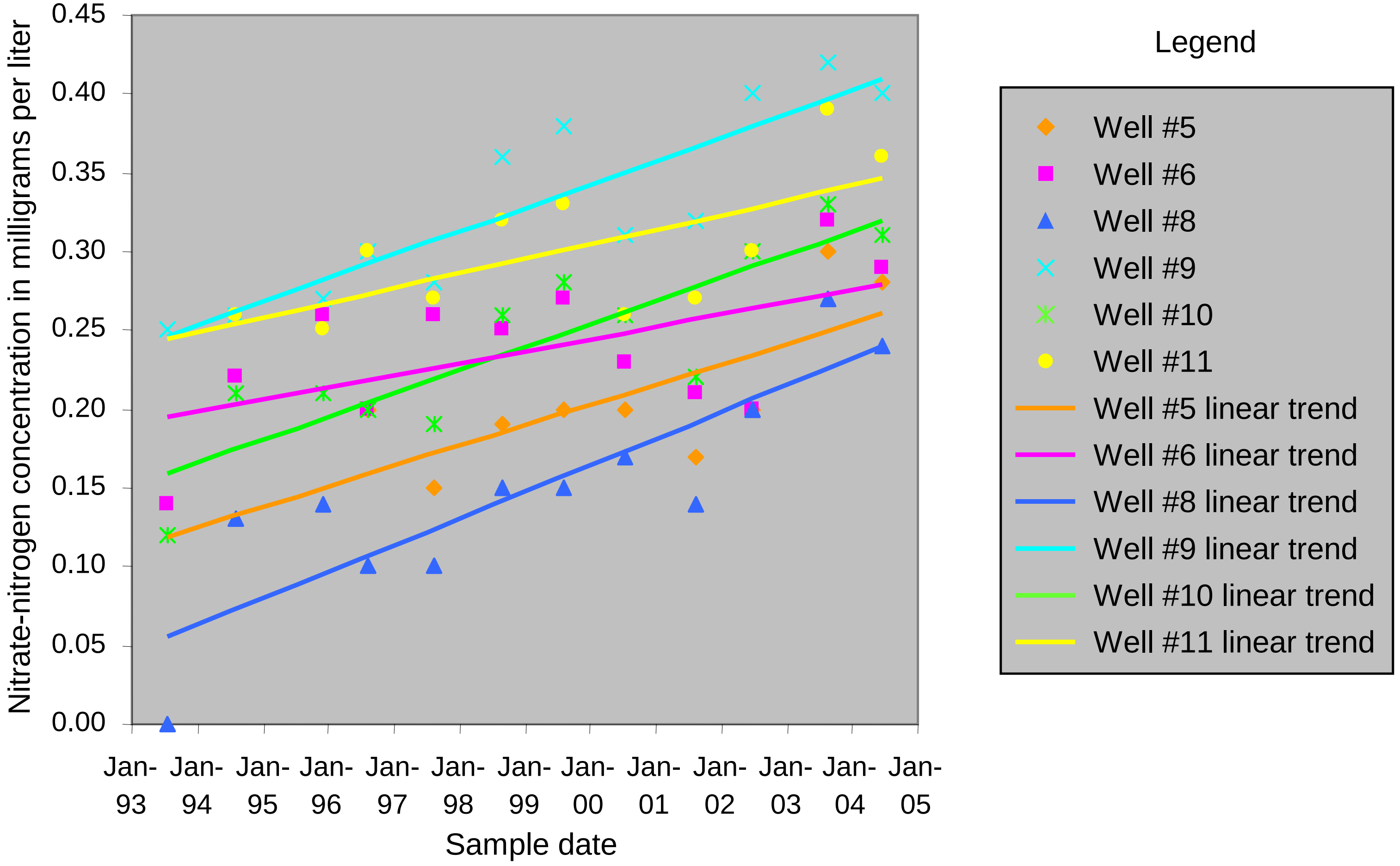
Mott and others (2004) studied a residential development of 261 homes served by on-site wastewater treatment systems underlain by the shallow alluvial aquifer associated with Rapid Creek east of Rapid City, South Dakota. Of those 261 homes, 240 occurred within an area of approximately 0.5 square miles (mi^2) which was interspersed with several large animal pastures.

Elevated concentrations and values of chloride, nitrate-nitrogen, and coliform bacteria in the shallow ground water correlated closely with areas of high-density on-site wastewater treatment systems, rather than with the animal pastures. Coliform bacteria were detected in greater than 90 percent of the tested residential wells. A similar investigation of effects on shallow ground water from on-site wastewater treatment systems in the community of Piedmont, South Dakota, analyzed ground water from 428 private and public drinking-water wells in a variety of unconfined and confined aquifers ranging from 10 to 2,000 feet in depth (Bartlett & West Engineers, Inc., 1998). Results of water-quality analyses from this study indicated that fecal coliform bacteria occurred in 4 percent of the sampled wells, total coliform bacteria were present in 28 percent of sampled wells, and nitrate-nitrogen concentrations were greater than or equal to 5.0 milligrams per liter (mg/L) in approximately 13 percent of the wells that were included in the investigation (Bartlett & West Engineers, Inc., 1998).

In a recent investigation, Rahn (2006) noted an increase in nitrate-nitrogen concentrations over a 12-year period from 1993 through 2004 in Rapid City public water-supply wells into the Madison aquifer (fig. 3). Rahn concluded that the nitrate-nitrogen was anthropogenic in origin and that sources included on-site wastewater treatment systems and fertilizer from home sites within aquifer recharge areas and nitrate-nitrogen in streamflow that recharges the Madison aquifer (Rahn, 2006). Rahn (2006) also suggested that commercial agriculture and mining probably were not the primary source of the increased nitrate-nitrogen concentrations because there are no feedlots, very few farm lands, and very little mining with explosives in the source water areas for these wells. Long and others (2006) also reported that population growth and development on and up-gradient from aquifer recharge areas in recent years may have influenced plumes of elevated nitrate-nitrogen concentrations that were identified within or near conduit flow paths in the Madison aquifer on the eastern flank of the Black Hills.

Other sources of water-quality data include information resulting from the ground-water quality monitoring network maintained by the South Dakota Department of Environment and Natural Resources and numerous water-quality publications released by the U.S. Geological Survey (Peter, 1985; Kyllonen and Peter, 1987; Goddard and Lockner, 1989; Zogorski and others, 1990; Freeman and Komor, 1991; Driscoll and Hayes, 1995; Williamson and others, 1996; Williamson, 2000; Williamson and Hayes, 2000; Naus and others, 2001; Williamson and Carter, 2001).

The South Dakota Department of Environment and Natural Resources currently is developing a total maximum daily load analysis for fecal coliform bacteria in Spring Creek. As part of this evaluation, deoxyribonucleic acid (DNA) ribotyping has indicated that 35 percent of samples in which fecal coliform bacteria were detected had a human source for the bacteria and 65 percent had an animal source (Schwickerath, 2004). Schwickerath also used computer models to apply combinations of best management practices to the Spring Creek watershed to meet the requirements of the total maximum daily load. Carter (2002) applied a similar approach to model the Whitewood Creek watershed and to develop best management practices that would meet total maximum daily load requirements for fecal coliform bacteria in Whitewood Creek. Swanson (2004) also evaluated nutrient loads to Sheridan Lake and Spring Creek and used a computer model to determine phosphorus reductions necessary to attain desired nutrient levels within Sheridan Lake. The South Dakota Department of Environment and Natural Resources (1999)



Modified from Rahn (2006)

Figure 3. Graph showing nitrate-nitrogen concentrations from Rapid City public water-supply wells in the Madison aquifer.

also recently completed the Source Water Assessment and Protection Program in the Black Hills area, and resulting data such as locations of dense concentrations of on-site wastewater treatment systems and delineations of source-water areas are extremely useful in understanding the effects of urbanization and potential effects of on-site wastewater treatment system proliferation.

The U.S. Environmental Protection Agency has published numerous guideline documents pertaining to protection of drinking-water resources (U.S. Environmental Protection Agency, 1991, 1993, 1997b), although these reports generally have not specifically addressed topics such as differing levels of development sustainability with respect to specific heterogeneous hydrogeologic settings. Other previous reports on ground-water vulnerability in the central Black Hills (Davis and others, 1994), and on design and construction criteria for on-site wastewater treatment systems within aquifer recharge areas (Rahn and others, 1993), also have contributed important information that could be assimilated into similar guideline documents. In the Black Hills, karst environments are significant because of the prevalence of limestone and dolomite in the region, and several guidance documents with aquifer protection information for karst settings are available (Zokaites, 1997; Doerfliger and Zwahlen, 1998; Veni and others, 2001).

The U.S. Environmental Protection Agency's *Onsite Wastewater Treatment Systems Manual* (2002a) is an excellent source of comprehensive information regarding on-site wastewater treatment systems. Major topics covered in this document include historic information on the use and management of on-site wastewater treatment systems, current management options and information, wastewater treatment performance requirements, and wastewater treatment processes and systems. Miscellaneous topics such as alternative systems, enhanced treatment techniques, commercial additives, and selection of system design and capacity also are described in this manual (U.S. Environmental Protection Agency, 2002b).

OBJECTIVES

The primary objective of this project was to respond to a fundamental societal need for information regarding the effects of effluent from wastewater treatment systems on ground-water quality in a heterogeneous hydrogeologic environment such as the Black Hills area. Such information may be useful for future land-management decisions regarding housing density in new developments and for educational purposes for homeowners with on-site wastewater treatment systems.

A number of scientific and societal challenges are inherent in the study of the environmental effects of on-site wastewater treatment systems. Some of the challenges include:

- Obtaining representative samples of ground water potentially affected by effluent from on-site wastewater treatment systems;

- Determination of the actual effects of on-site wastewater treatment systems on water resources, both on a local (system by system) scale and on a more regional scale such as a watershed or a geographic area such as the Black Hills uplift;

- Incorporation and implementation of protection measures by local governmental entities and acceptance of management measures by the general public;

- Balancing the rights of property owners with societal needs for environmental protection, addressing changes in land valuation and development potential, increased costs, regulations, and taxes;

- Sampling treated effluent from on-site wastewater treatment systems in the unsaturated zone within a karst environment;

- Accurately characterizing the effects of varying age and design of on-site wastewater treatment systems with respect to system effluent;

- Developing accurate, usable, land management guidelines, given the extremely variable hydrogeologic settings of development sites; and

- Incorporating hydrogeologic uncertainty and site specific investigations into regulations.

Although several of these fundamental challenges pertain to societal issues that are beyond the scope of this project, some of these challenges can be addressed through hydrogeologic investigations. Other identified challenges can be addressed only after scientific information has been supplied as a basis for public policy and educational efforts. Therefore, it is important to focus on the scientific quantification of the potential effects of treated effluent from on-site wastewater treatment systems so that the proper information is available to better understand and manage the hydrologic resources of the Black Hills and other similar regions. It is also important not to overstate the conclusiveness of sample results from a study such as this. Many more hydrogeologic settings, system designs, and maintenance techniques must be investigated, and more comprehensive contaminant analyses must be performed in order to provide the information necessary to properly characterize these complex issues.

A total maximum daily load project for the Spring Creek watershed was performed concurrently with this project by a consortium of agencies including the South Dakota Department of Environment and Natural Resources, the South Dakota School of Mines and Technology, the South Dakota Department of Game, Fish and Parks, and the U.S. Forest Service. One of the primary goals of this cooperative project was to establish a total maximum daily load for fecal coliform bacteria for surface-water bodies in the Spring Creek watershed. Potential sources for fecal coliform bacteria within this watershed include on-site wastewater treatment systems and wastewater treatment lagoons operated by the city of Hill City. These circumstances presented a unique opportunity to coordinate efforts between multiple agencies to examine the effect of wastewater treatment facilities on ground water simultaneously with the total maximum daily load project which is focused on surface water.

METHOD OF STUDY

The objectives described previously were addressed primarily through identification of specific hydrogeologic settings in the Black Hills area and by direct evaluation of on-site wastewater treatment system effluent within those settings. Because of their sensitivity to contamination, desirability with respect to residential development, and importance to local drinking-water supplies, alluvial aquifers and karst limestone aquifers were both selected for analysis in this study. To integrate results from this project with the concurrent Spring Creek total maximum daily load project, study locations for each of these hydrogeologic settings were selected from within the Spring Creek watershed. For comparison purposes, study locations for both of these hydrogeologic settings also were selected from other watersheds in the central Black Hills. Additionally, sample locations were selected and monitoring wells were installed to monitor the Hill City wastewater treatment lagoons, which are located immediately adjacent to Spring Creek. Surface-water quality data resulting from the Spring Creek total maximum daily load project, and other pre-existing water-quality data, also were evaluated to maximize comprehension of the effects of on-site wastewater treatment systems on water resources. As previously stated, an effort was made to investigate properly maintained on-site wastewater treatment systems, with no known functional problems.

Specific tasks that were completed as part of this study to achieve the stated objectives included the following:

- Locate appropriate study and sample locations, obtain permission from facility owners to perform research, and confirm access to springs and surface-water sampling locations;

- Design monitoring-well networks and install monitoring wells at selected locations in unconsolidated alluvial deposits;

- Measure water levels to allow estimation of potential ground-water flow direction and interaction between surface water and ground water;

- Collect and analyze water samples for chemical constituents and microorganisms indicative of treated wastewater from on-site wastewater treatment systems and municipal wastewater lagoons;

- Use sample results to begin to characterize the effects of on-site wastewater treatment systems and wastewater treatment lagoons on ground-water quality; and

- Coordinate efforts with the total maximum daily load project in the Spring Creek watershed involving the South Dakota Department of Environment and Natural Resources, the South Dakota School of Mines and Technology, the South Dakota Department of Game, Fish and Parks, and the U.S. Forest Service.

Funding for laboratory analyses of water samples was provided by the Water Resources Assistance Program, South Dakota Department of Environment and Natural Resources, as part of the total maximum daily load project for the Spring Creek watershed. All laboratory analyses were completed by the South Dakota State Health Laboratory in Pierre, South Dakota. The Geological Survey Program, South Dakota Department of Environment and Natural Resources, also provided personnel, drilling equipment, and supplies for installation of monitoring wells and for the collection of water-quality samples.

All installed monitoring wells were surveyed with respect to a site datum, and all water-quality samples were collected in accordance with standard industry practices (South Dakota Department of Environment and Natural Resources, 2000). At least 10 percent of the collected water samples consisted of either blanks or duplicate samples to provide quality assurance and control. Each monitoring well was constructed so that the screened interval intersected the water table in an effort to maximize the opportunity to encounter contaminants such as coliform bacteria.

Parameters that were included in the evaluation of each site were selected for their usefulness as indicators of adverse effects on ground-water quality from on-site wastewater treatment systems. These parameters included several types of nutrients, bacteria, and chemical constituents, and field measurements for pH, temperature, and conductivity. A complete list of analyzed water-quality parameters, their units of measurement, and drinking-water standards for the parameters that have them are given in table 1. Drinking-water standards listed in table 1 are from U.S. Environmental Protection Agency (1994, 2002b) and the South Dakota Department of Environment and Natural Resources (2003). Maximum contaminant levels are enforceable standards that refer to the highest level of a contaminant that is allowed in drinking water. Secondary maximum contaminant levels are unenforceable standards generally related to contaminants that can adversely affect the taste, odor, or appearance of water (U.S. Environmental Protection Agency, 1994). The dates of sample collection were from September 2002 to November 2003 and samples were collected on a quarterly basis so that seasonal variations could be observed.

Nutrients that were evaluated in this investigation included nitrate-nitrogen, ammonia, total Kjeldahl nitrogen, and total and dissolved phosphorus. Dissolved nitrogen occurs in ground water in the forms of nitrate-nitrogen, nitrite-nitrogen, ammonium, ammonia, nitrogen, nitrous oxide, and organic nitrogen in various organic substances (Freeze and Cherry, 1979). Nitrogen is discharged from septic tanks primarily in the form of ammonia, which is converted to nitrite-nitrogen and then to nitrate-nitrogen through biological aerobic nitrification processes in the drain field and underlying infiltrative surfaces (Taylor, 2003). According to the U.S. Environmental Protection Agency (2002a), nitrate-nitrogen is the most significant documented threat to ground water from on-site wastewater treatment systems, migrating readily with ground water and posing human health hazards. Therefore, nitrate-nitrogen was selected as one of the parameters for laboratory analysis in this study. Ammonia also was selected as a parameter for analyses as another form of dissolved nitrogen associated with wastewater treatment systems which can cause algal blooms and excessive plant growth and is toxic to fish under certain conditions (Fetter, 1980; Williamson and Carter, 2001). Total Kjeldahl nitrogen, the sum of organic nitrogen and ammonia, also can become concentrated as a result of wastewater effluent, and is a useful parameter for determination of organic nitrogen content (Swanson, 2004; Minnesota Pollution Control Agency, 2006).

Table 1. Analyzed water-quality parameters, units of measurement, and drinking-water standards

Water-quality parameter	Unit of measurement	Drinking-water standard
Ammonia	Milligrams per liter	--
Phosphorus, total	Milligrams per liter	--
Phosphorus, dissolved	Milligrams per liter	--
Nitrate-nitrogen	Milligrams per liter	10[1]
Total Kjeldahl nitrogen	Milligrams per liter	--
Chloride	Milligrams per liter	250[2]
Caffeine	Micrograms per liter	--
Fecal coliform bacteria	Colonies per 100 milliliters	zero[1]
Escherichia coli (*E. coli*)	Most probable number per 100 milliliters	zero[1]
Conductivity	Microsiemens	--
pH	$-\log_{10}$ of hydrogen ion activity in solution	6.5-8.5[2]
Temperature	Degrees Celsius	--

-- No drinking water standard established
[1] Maximum contaminant level
[2] Secondary maximum contaminant level

Phosphorus is another plant nutrient that can become concentrated as a result of human and animal wastes and fertilizers, contributing to eutrophication of surface waters and reduction of dissolved oxygen (Williamson and Carter, 2001; U.S. Environmental Protection Agency, 2002b). Phosphorus occurs in on-site wastewater treatment system effluent either dissolved or as suspended particulate matter, and it is primarily in the form of orthophosphates and organically bound phosphates (University of Wisconsin, 1978; American Public Health Association, 1998). Total phosphorus and dissolved phosphorus were both evaluated in this investigation as additional indicators of the potential effects of on-site wastewater treatment systems on ground-water and surface-water quality in the study areas.

Fecal coliform bacteria are enteric bacteria associated with human and animal wastes (U.S. Environmental Protection Agency, 2002b). They are used as specific indicators of fecal contamination and as indicators for the possible presence of other potentially harmful pathogens (Droste, 1997; U.S. Environmental Protection Agency, 2002a). *Escherichia coli* (*E. coli*) is the predominant member of the fecal coliform group, and if ingested by humans, strains of this species can cause illnesses which can result in vomiting, diarrhea, cramping, headache, kidney failure, and death in susceptible populations (Droste, 1997; U.S. Environmental Protection Agency, 2002b; Standridge, 2008). Laboratory analyses were performed specifically for *E. coli* bacteria and for the more general category of fecal coliform bacteria in this investigation. Although *E. coli* bacteria are fecal coliform bacteria, results of laboratory analyses are presented and discussed separately for these two parameters. It was originally planned to perform DNA ribotyping if fecal coliform bacteria were found to evaluate whether the source was human or animal. However, detections of fecal coliform bacteria were sporadic and rare, so DNA ribotyping was not attempted.

Chloride is a significant inorganic anion in wastewater resulting from human diets, water softeners, and other sources. Chloride also is biologically inactive, highly soluble, non-reactive in soil, and it can leach readily to ground water (American Public Health Association, 1998; Meyer, 2000). Although the effects of elevated concentrations of chloride in drinking water are largely limited to undesirable taste, chloride has been useful as a ground-water tracer and as an indicator of impacted ground-water quality from sources such as sewage lagoons, on-site wastewater treatment systems, livestock facilities, landfills, and road salt (Meyer, 2000; Mott and others, 2004). Therefore, chloride was included in this investigation of potential effects on ground-water quality from on-site wastewater treatment systems. Caffeine also is peculiar to the human diet and was included in this investigation as an indicator of the presence of effluent from on-site wastewater treatment systems.

Field measurements were recorded for conductivity, temperature, and pH for each sample collected in this investigation. Conductivity is a measure of the electrical conductance of water, and it is used to determine the approximate concentration of dissolved solids (Williamson and Carter, 2001). Temperature and pH are important to document because these parameters affect other constituents and conditions. The pH is the measure of the hydrogen ion concentration, indicating acidic, neutral, or alkaline solutions, and it can affect pathogen survival rates, toxicity of ammonia, phosphorus sorption and precipitation reactions, and the mobility of metals in ground water. Temperature also affects pathogen survival rates, ammonia toxicity, and measurement of conductivity concentrations (Fetter, 1980; Williamson and Carter, 2001; U.S. Environmental Protection Agency, 2002b).

Precipitation data over the duration of the study were obtained from four stations in reasonably close proximity to the sites of investigation (app. A). Streamflow data from streams that were either directly sampled or were immediately adjacent to on-site wastewater treatment systems included in this investigation are presented in appendix B. Subsurface lithologic data and other information pertaining to monitoring wells that were installed as part of this investigation are presented in appendix C.

Specific methods that were followed within the different hydrogeologic settings are described in further detail under the following three subheadings which are: 1) alluvial deposits, 2) karst limestone, and 3) municipal wastewater treatment lagoons, Hill City, South Dakota. Although the Hill City wastewater treatment lagoons occur in an alluvial setting, it is described separately because of the differences between this system and residential on-site wastewater treatment systems.

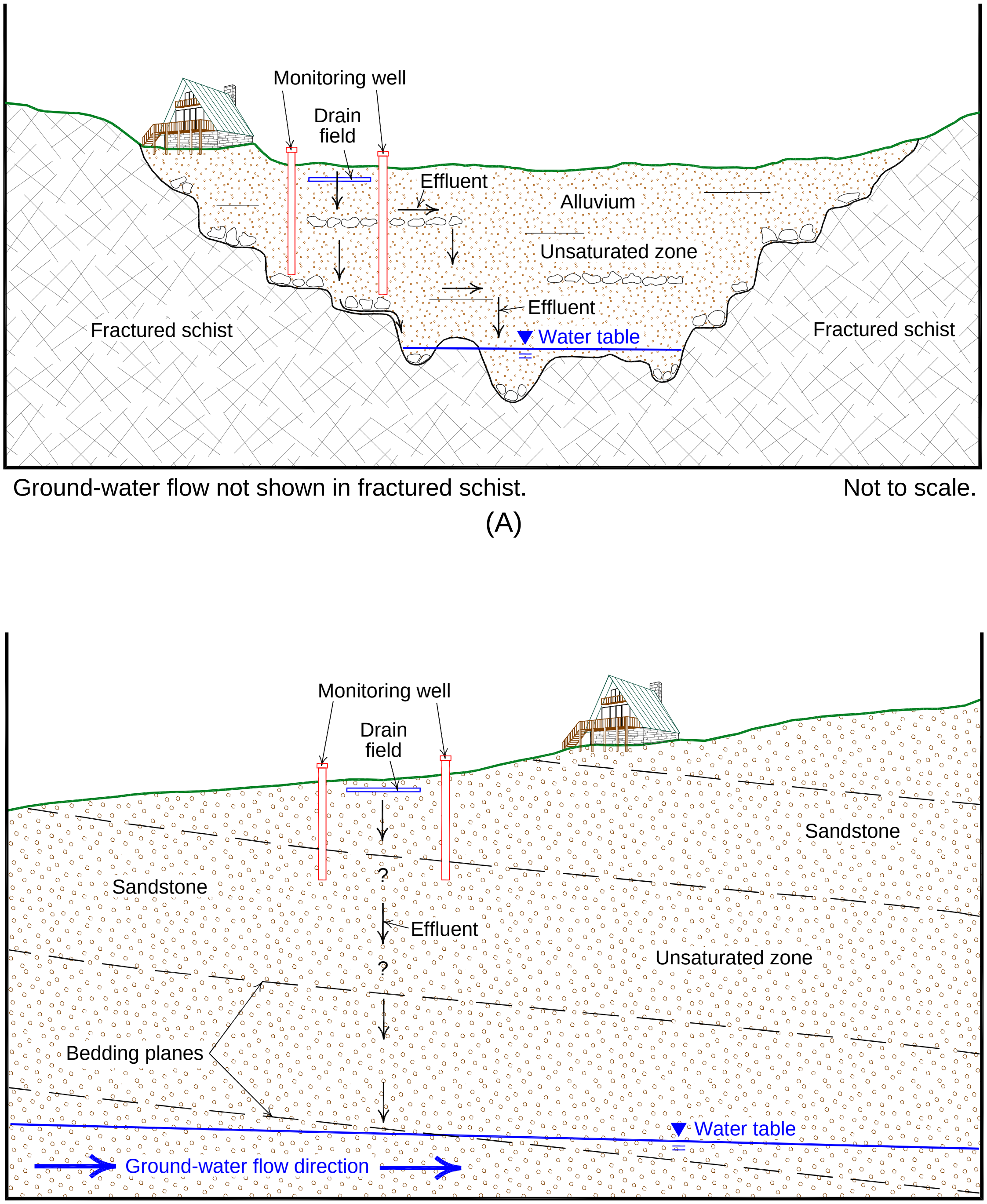
Alluvial Deposits

Permission was obtained to evaluate three privately owned on-site wastewater treatment systems constructed in alluvial settings suitable for analysis with shallow, ground-water monitoring-well networks. Locations of the three study areas are shown in figure 2. The initial approach for analysis of on-site wastewater treatment systems within alluvial deposits was to first install three monitoring wells at each study location to determine the configuration of the local water table, which was to be followed by installation of additional monitoring wells in appropriate locations once water-table conditions were defined. Samples for water-quality analyses were then to be collected from an up-gradient well to establish background conditions and from several down-gradient wells to evaluate for chemical and biological constituents contributed by the on-site wastewater treatment systems. However, difficulties with site access, drilling conditions (auger refusal), and the need to minimize disturbance of private land limited potential well locations. For example, a fourth potential study location in an alluvial setting was abandoned after three auger tests drilled to bedrock or large boulders and did not encounter ground water.

Another difficulty with investigation of ground-water quality near on-site wastewater treatment systems occurs because of unsaturated subsurface conditions. Figure 4 (A) shows a schematic cross section of subsurface conditions typical of the alluvial hydrogeologic setting in the central Black Hills. The diagram shows effluent migrating downward and laterally through unsaturated, heterogeneous alluvial sediments and that the water table is below the base of the alluvium immediately beneath the on-site wastewater treatment system. Such a setting precludes the use of shallow alluvial wells as a monitoring option. Another unsuccessful attempt to encounter saturated conditions down-gradient from an on-site wastewater treatment system located within an unsaturated sandstone formation is illustrated in figure 4 (B). In this example the water table was several hundred feet below ground surface and thus, no shallow ground water was encountered in the vicinity of the on-site wastewater treatment system.

After numerous attempts with the drilling rig, three monitoring wells were eventually installed at each study location, although well installation varied from excellent to poor with respect to establishing sampling locations both up-gradient and down-gradient from each on-site wastewater treatment system. Two of the on-site wastewater treatment systems that were investigated experienced year-round, residential usage, and one on-site wastewater treatment system that was investigated was used on a seasonal basis at a municipal golf course. One of the residences was located near the town of Hill City in an alluvial valley containing a small tributary to Spring Creek, and the other residence was located downstream from the community of Johnson Siding on alluvium immediately adjacent to Rapid Creek. The golf-course site was located in the city of Custer, South Dakota, on alluvial material immediately adjacent to a small impoundment on French Creek named West Dam.

Following installation of the monitoring-well network, ground-water samples were collected at each site. The samples were analyzed for parameters indicative of effects on ground water resulting from treated wastewater discharged by the on-site wastewater treatment systems.



Ground-water flow not shown in fractured schist. Not to scale.

(A)

Not to scale.

(B)

Figure 4. Schematic diagrams showing hydrogeologic settings that are difficult to investigate with traditional drilling methods. (A) Unsaturated alluvial deposit over fractured schist, and (B) Unsaturated sandstone.

Karst Limestone

Quantification of the effects of on-site wastewater treatment systems located within karst limestone settings is difficult because of the depth of the water table and the extent of unsaturated and anisotropic conditions in the subsurface. However, knowledge of the mechanics of karst hydrogeology can be used in the effort to study the effects of on-site wastewater treatment systems in this setting. The approach taken in this study was first to evaluate water-quality conditions for surface water that enters and recharges the karst system directly through swallow holes along streams underlain by the limestone. Water-quality samples also were obtained from down-gradient wells and artesian springs, which are supplied by water from the karst aquifer, and which also are down-gradient from significant unincorporated developments served largely by individual on-site wastewater treatment systems. By comparing the water quality of the up-gradient surface water with the ground water down-gradient of developed areas, it was attempted to document some of the chemical and biological constituents that are entering the ground water within the karst limestone as a result of on-site wastewater treatment systems. It should be noted, however, that variable ground-water travel times and mixing ratios of different age water are potential complicating factors in any ground-water study within karst limestone terrain. Furthermore, sample results in this setting may represent transient or emerging effects of contaminants from on-site wastewater treatment systems due to the potential lag time between recharge and discharge, and variations caused by changing climatic conditions.

Samples to evaluate surface-water quality prior to recharge to karst limestone aquifers were collected just up-gradient of major outcrop areas of the Pahasapa (Madison) Limestone along Bear Butte Creek, Boxelder Creek, Rapid Creek, and Spring Creek (fig. 2). Water-quality samples from sources located down-gradient of residential development were collected from eight public water-supply system wells that obtain water from the Madison aquifer and from two Madison aquifer artesian springs, one of which (Jackson Spring) is also used as a public water supply. Analyzed water-quality parameters were the same for the karst limestone setting as for the unconsolidated alluvial deposits (table 1). Samples were collected on a quarterly schedule from January 2003 through November 2003.

Municipal Wastewater Treatment Lagoons, Hill City, South Dakota

The wastewater treatment lagoons serving the city of Hill City is estimated to be discharging up to approximately 38 million gallons per year (gal/yr) of wastewater to ground water within alluvial deposits along Spring Creek, and possibly to the creek itself, because of an imbalance between known inflows and calculated evaporation rates (McLaughlin Water Engineers, Ltd., 2000). This system of lagoons was included in this study to better understand effects of human wastewater on shallow ground water in an alluvial hydrogeological setting and to coordinate with another study in the Spring Creek watershed having the goal to establish total maximum daily loads to the creek. Because these lagoons occur within unconsolidated alluvial deposits, the study approach was similar to that for the previously described residential on-site wastewater treatment systems which also were located in unconsolidated alluvial deposits. However, the larger size of the facility and the much greater quantity of wastewater being treated at the site are significantly distinct from other localities in this study. Similar to the residential on-site wastewater treatment systems, an up-gradient monitoring well was installed to document background conditions, and four additional monitoring wells were installed where access allowed around the perimeter of the three lagoons. Analyzed water-quality parameters were the same as for the previous hydrogeologic settings (table 1), and samples were collected on a quarterly sampling schedule from September 2002 through August 2003.

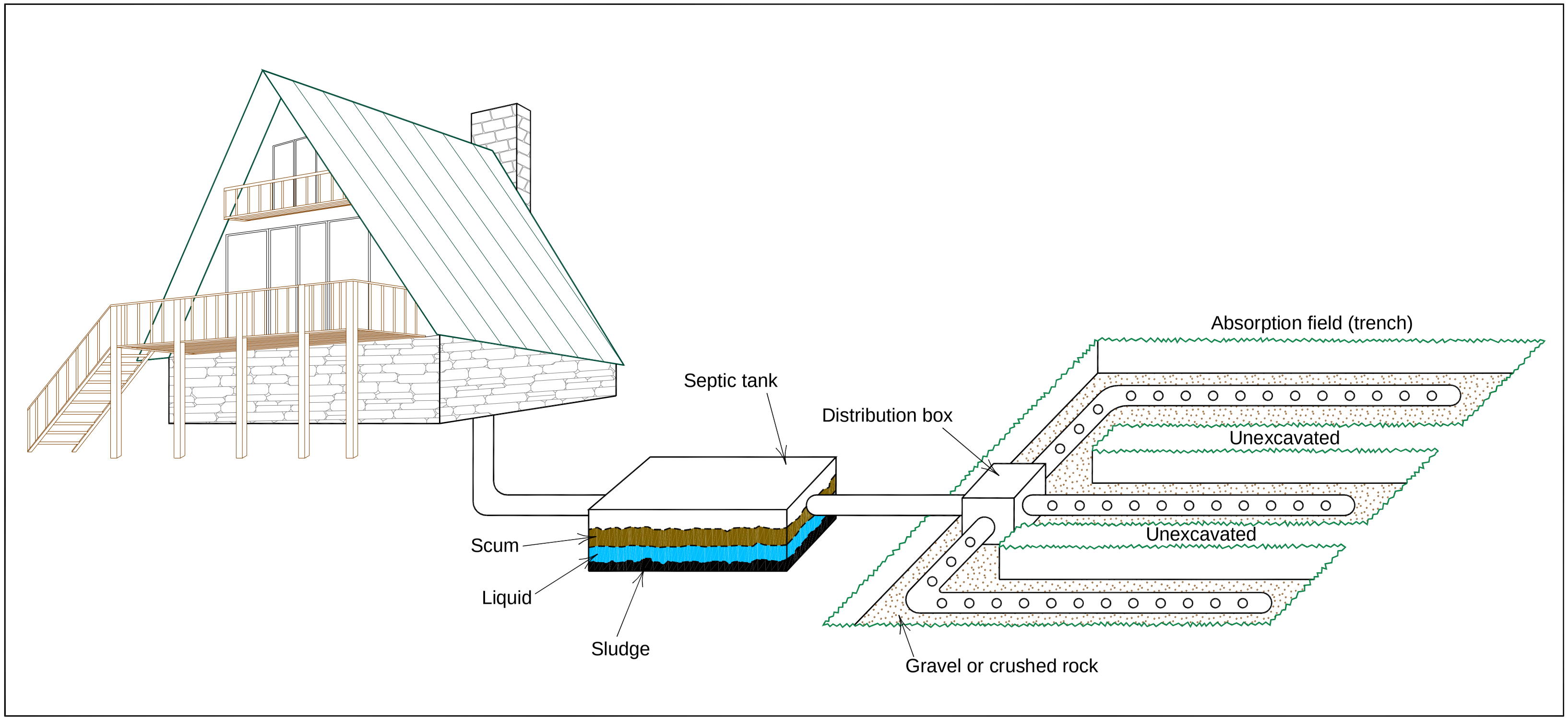
ON-SITE WASTEWATER TREATMENT SYSTEM COMPONENTS AND PROCESSES

Conventional on-site wastewater treatment systems typically consist of a septic tank and a soil absorption field, also known as a subsurface wastewater infiltration system or drain field (fig. 5). The primary function of on-site wastewater treatment systems is the removal and storage of settleable solids, floatable grease and scum, nutrients, and pathogens from effluent that is discharged to the environment. The tank component of the system removes most settleable and floatable materials and functions as an anaerobic bioreactor that partially digests organic matter in the tank (U.S. Environmental Protection Agency, 2002b). The drain-field component of the system typically consists of perforated distribution pipes installed within gravel, crushed rock, or other porous media (fig. 6). The porous media in the drain field promotes further treatment through adsorption, filtration, and biological processes in an aerobic environment. The porous media also facilitates delivery of effluent to the underlying infiltrative surface, and it provides storage for wastewater during peak flows (Ayers Associates, 1991).

Beneath the porous media in the drain field, a biologically active infiltration zone (fig. 6), usually a few centimeters thick, provides significant physical, biological, and chemical treatment of effluent. This zone is characterized by an accumulation of particulate material and development of an active biomass, or “biomat,” sustained by nutrients in the particulate matter and effluent. Metabolic by-products also accumulate in the infiltration zone, and carbonaceous material is degraded. There is a sharp decline in hydraulic conductivity at this zone because of soil pore blockage and microbiological growth and by-products, and fluid flow changes from saturated above to unsaturated below the infiltration zone. Nitrification occurs immediately below the infiltration zone if oxygen is present in sufficient quantity (Ayers Associates, 1991; U.S. Environmental Protection Agency, 2002b).

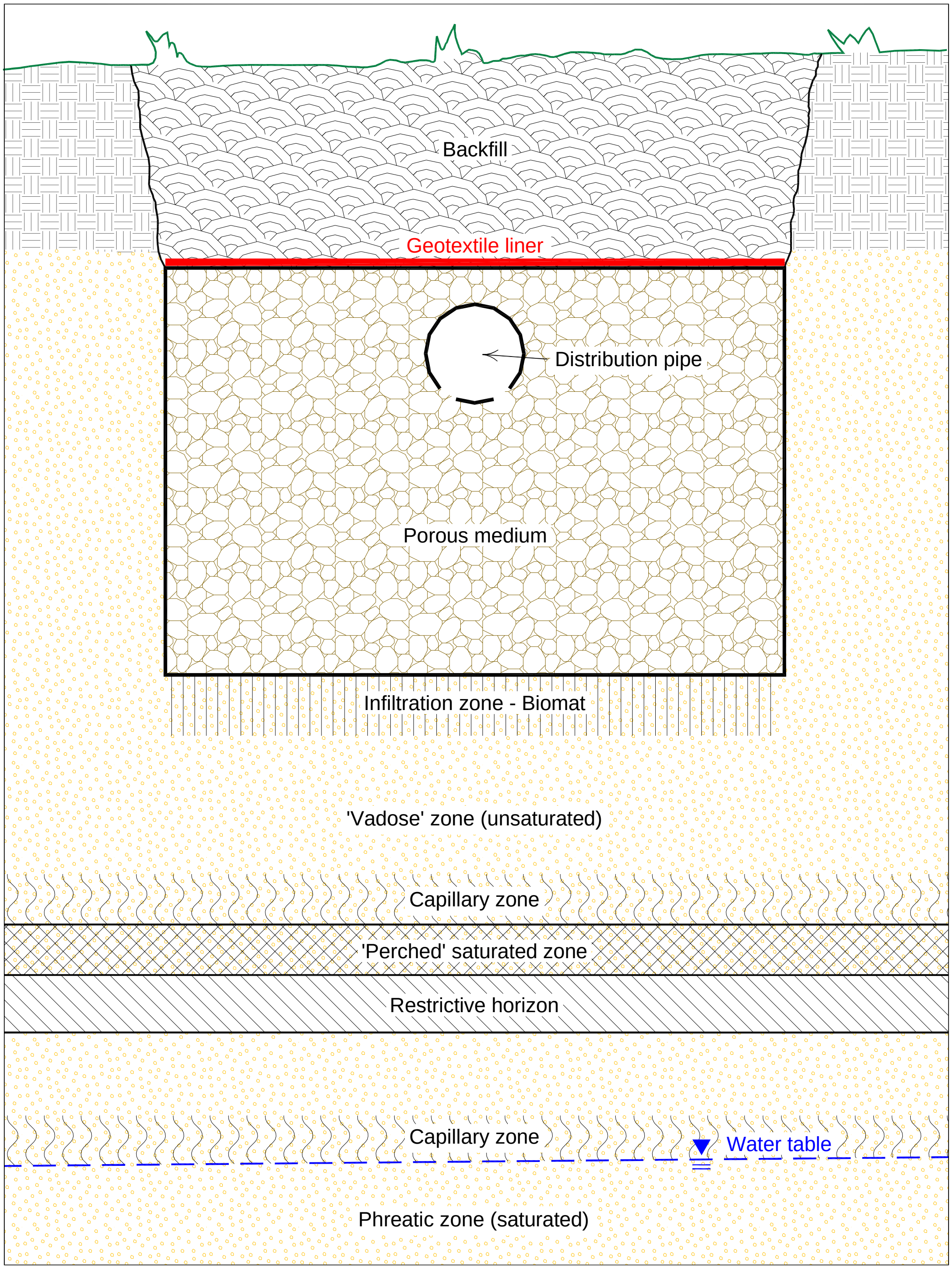
The vadose (unsaturated) zone occurs below the infiltration surface (fig. 6), usually within native soils, providing additional physical, biological, and chemical treatment of the effluent before it reaches ground water in the underlying saturated zone. Because of capillary and adsorptive forces of the soil, fluid flow in the vadose zone is under a negative pressure potential (less than atmospheric), and flow is primarily over soil particle surfaces and through smaller pores, leaving larger pores air-filled. This unsaturated environment allows oxygen to reach microbes that grow on soil particle surfaces, and it is where most sorption reactions occur and where much of the phosphorus and pathogen removal takes place (Ayers Associates, 1991; U.S. Environmental Protection Agency, 2002b).

After passing through the vadose zone, effluent enters the zone of saturation, or ground water, and is transported from the site by fluid movement in response to positive pressure gradients. Ground-water flow is usually laminar, and mixing of effluent with ground water is



Adapted from U.S. Environmental Protection Agency (2002a)

Figure 5. Schematic diagram showing major components of a typical on-site wastewater treatment system.



Adapted from Ayres Associates (1993)

Figure 6. Schematic cross section showing soil treatment zones for a typical on-site wastewater treatment system drain field.

often limited, resulting in distinct wastewater plumes for some distance from the point of origin (fig. 7). Such plumes may descend into the ground water because of precipitation recharge, and some dispersion occurs; however, migration of solutes in the plume varies with hydrogeologic conditions and soil-solute reactivity (U.S. Environmental Protection Agency, 2002b).

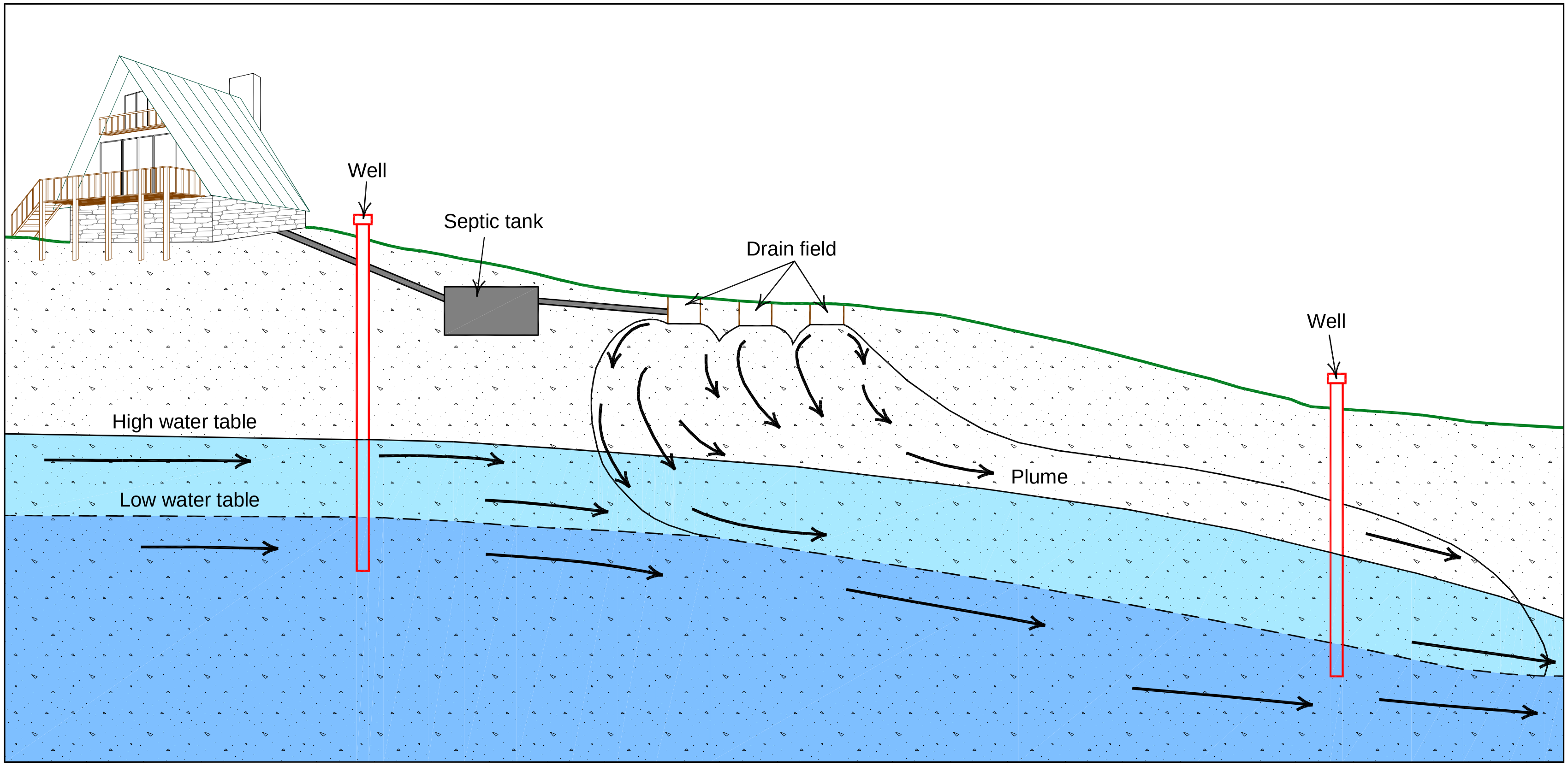
Effluent Characteristics

Wastewater, or effluent, discharged from on-site wastewater treatment systems originates from several different sources and activities, including toilet usage, bathing, dishwashing, laundry, cleaning activities, faucets/sinks, floor drains, garbage disposals, and water conditioners. Effluent flow rates are affected by family size, efficiency of water fixtures, age and socioeconomic status of residents, water composition, and other factors (U.S. Environmental Protection Agency, 2002b). According to one of the largest and most comprehensive studies of effluent flow rates to date (Mayer and others, 1999), median daily per capita flow for residential on-site wastewater treatment systems ranges from 54 to 67 gallons/person/day.

Typical pollutants in residential wastewater consist of suspended solids, organic compounds, pathogenic microorganisms, nitrogen, phosphorus, heavy metals, and dissolved inorganics. Table 2 shows constituent mass loadings and concentrations of contaminants in typical residential wastewater (U.S. Environmental Protection Agency, 2002b). The performance and efficiency of on-site wastewater treatment systems, and the underlying soil horizons, in reducing and removing these constituents determines the quality of the effluent that enters the receiving ground-water environment. Properly designed, located, installed, operated, and maintained on-site wastewater treatment systems are capable of almost complete removal of suspended solids, biodegradable organic compounds, and fecal coliform bacteria; however, if treatment is incomplete, contamination of ground water can occur from these constituents (U.S. Environmental Protection Agency, 2002b). Further risks to ground-water quality are posed by pollutants such as nitrogen, phosphorus, pathogenic parasites, bacteria, viruses, toxic organic compounds, metals, and dissolved inorganics. The age and condition of on-site wastewater treatment systems, and the suitability of soils beneath the infiltrative surface, also are critical factors for treatment of effluent before reaching the ground-water environment (U.S. Environmental Protection Agency, 2002b).

Fate and Transport of Contaminants in Ground Water

Movement of contaminants discharged by on-site wastewater treatment systems is dependent on many variables, including soil composition and layering, underlying geologic units, topography, precipitation, and physical and chemical properties of the contaminants. The shape of effluent plumes also are affected by the uniformity of effluent distribution in the drain field, the position of the drain field with respect to ground-water flow and direction, and preferential flow paths in the vadose and unsaturated zones. Effluent plumes defined by nitrate-nitrogen tend to be long and narrow, with little dispersion, and can extend for hundreds of feet, while infective viruses have been documented to travel over 1,300 feet (U.S. Environmental Protection Agency, 2002b). Areas underlain by karst terrain or sandy soils are particularly likely to experience rapid



Adapted from National Small Flows Clearinghouse (2000)

Figure 7. Schematic cross section showing possible plume movement under an on-site wastewater treatment system drain field.

Table 2. Constituent mass loadings and concentrations in typical residential wastewater

Constituent	Mass loading (grams per person per day)	Concentration (milligrams per liter)
Total solids	115-200	500-800
Volatile solids	65-85	280-375
Total suspended solids	35-75	155-330
Volatile suspended solids	25-60	110-265
5-day biochemical oxygen demand	35-65	155-286
Chemical oxygen demand	115-150	500-660
Total nitrogen	6-17	26-75
Ammonia	1-3	4-13
Nitrite- and nitrate-nitrogen	<1	<1
Total phosphorous[1]	1-2	6-12
Fats, oils, and grease	12-18	70-105
Volatile organic compounds	0.02-0.07	0.1-0.3
Surfactants	2-4	9-18
Total coliforms[2]	---	10^8-10^{10}
Fecal coliforms[2]	---	10^6-10^8

Information from U.S. Environmental Protection Agency (2002a)

The information in this table is for typical residential dwellings equipped with standard water-using fixtures and appliances; assumed water use of 60 gallons per person per day (227 liters per person per day)

[1] The detergent industry has lowered the total phosphorous concentrations since early literature studies; therefore, Sedlak (1991) was used for total phosphorous data

[2] Concentrations presented in “most probable number” of organisms per 100 milliliters

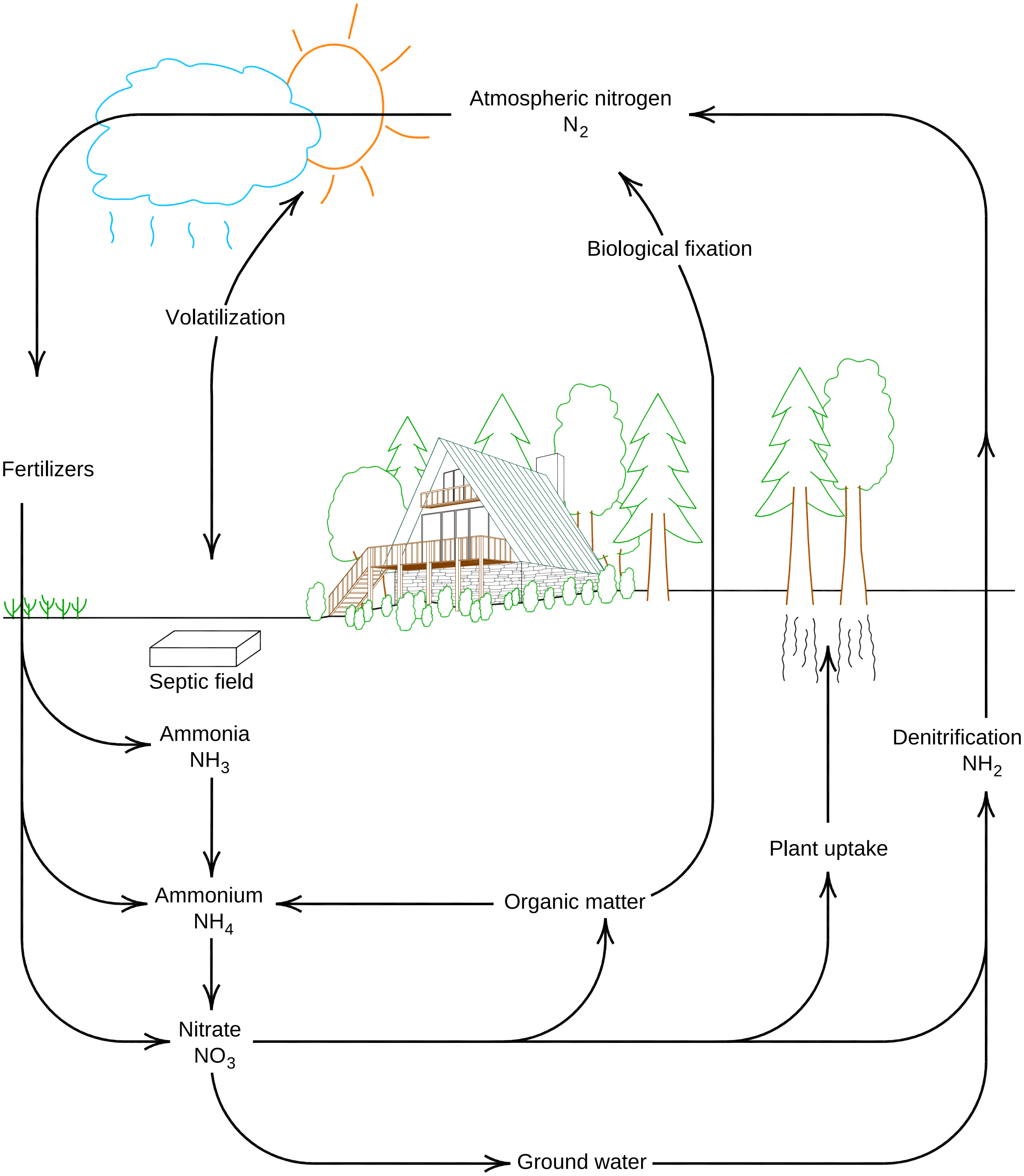
movement of bacteria, viruses, nitrate-nitrogen, and other contaminants. The following paragraphs address the typical fate and transport in ground water for the principal categories of effluent contaminants.

According to the U.S. Environmental Protection Agency (2002a), nitrate-nitrogen is the most significant documented threat to ground water from on-site wastewater treatment systems. Primary health threats from excessive nitrate-nitrogen include methemoglobinemia (blue-baby syndrome) in infants and complications for pregnant women. In surface water nitrate-nitrogen can contribute to eutrophication, dissolved oxygen loss, and degradation of aquatic habitat. Furthermore, the effects of the discharge of nitrate-nitrogen from long-term, high-density residential development is difficult to estimate and is a source of concern among many communities, particularly those with sensitive aquifer recharge areas or sandy soils (Hantzsche and Finnemore, 1992).

Nitrogen enters on-site wastewater treatment systems primarily as human waste containing urea, organic matter, and ammonia, most of which is converted to ammonia and ammonium in the septic tank (Taylor, 2003). Ammonia and ammonium then are discharged to the infiltrative surface and upper vadose zone where they are converted by aerobic bacteria almost entirely to nitrite-nitrogen, then to nitrate-nitrogen (fig. 8). Nitrate-nitrogen is a highly soluble, negatively charged ion which is transported through negatively charged soil particles with little soil adsorption, and it moves readily with infiltrating wastewater, precipitation recharge, and ground water (U.S. Environmental Protection Agency, 2002b; Taylor, 2003). Nitrate-nitrogen plumes in ground water have been documented to migrate over 300 feet laterally from discharge points, and they can persist for years or decades (Robertson and Cherry, 1995; Minnesota Pollution Control Agency, 1999; Taylor, 2003). Near on-site wastewater treatment systems, nitrate-nitrogen concentrations in ground water are usually found to exceed the 10 mg/L drinking-water standard, and in areas with higher development densities nitrate-nitrogen has been documented in excess of 130 mg/L (Yates, 1985; Shaw and Turyk, 1994; Minnesota Pollution Control Agency, 1999; Taylor, 2003). Biological denitrification can remove nitrogen from percolating wastewater under anaerobic conditions in the presence of an organic carbon source; however, it is generally considered that these conditions seldom occur at most sites (Ayers Associates, 1991; U.S. Environmental Protection Agency, 2002b; Taylor, 2003).

Nitrate-nitrogen concentrations in ground water also have been observed to vary with climatic and seasonal changes (Taylor, 2003). Previous investigations have reported higher nitrate-nitrogen concentrations in the winter and lower concentrations in summer months, possibly as a result of plant growth uptake and variations in soil leaching (Lindsey and others, 1997). Nitrate-nitrogen concentrations also have been found to increase during storm events and wet climatic conditions and to decrease during drought and dry conditions (Wehrmann, 1984; Taylor, 1996).

Sources of nitrogen in ground water other than from on-site wastewater treatment systems include industrial processes, agricultural activities, livestock waste, and fertilizer application (Taylor, 2003). Most of the residential developments served by on-site wastewater treatment systems in the Black Hills area are not associated with industrial processes or large-scale agricultural activities; however, fertilizer is commonly used for lawn care around many



Adapted from Taylor (2003)

Figure 8. Flow chart showing primary components of the nitrogen cycle.

residences. Petrovic (1990) reported that leaching of nitrate-nitrogen from lawn fertilizer is highly variable with some studies finding little to no effects while others report significant nitrogen loading from fertilizer. Hantzsche and Finnemore (1992) estimated that nitrate-nitrogen concentrations of approximately 0.37 to 1.1 mg/L result from leaching of fertilizer from a typical fertilized lawn with an assumed rainfall recharge rate of 12 inches per year. Taylor (2003) summarized data from numerous previous investigations and reported that 5 to 80 percent of the nitrogen in lawn fertilizer reaches the water table. Factors that affect nitrogen leaching from fertilizer include soil type, nitrogen application rate, season of application, lawn watering, and nitrogen source (Petrovic, 1990). Livestock and wildlife also are present in Black Hills watersheds and are additional potential sources of nitrogen to surface waters which recharge local aquifers.

Another serious health threat associated with on-site wastewater treatment systems is posed by the introduction of pathogenic microorganisms, including bacteria, viruses, protozoa, and parasites, which can cause a wide range of gastrointestinal, neurological, respiratory, renal, and other diseases (U.S. Environmental Protection Agency, 2002b). The occurrence and concentration of pathogenic microorganisms in wastewater are extremely variable, depending on the source of the wastewater, occurrence of infected persons in the population, and pathogen survival rates in the receiving environment. Waterborne pathogens found in human waste are shown in table 3, along with their associated diseases and effects of infection.

Primary environmental factors that influence pathogen survival rate include the initial quantity and types of organisms, temperature, humidity, sunlight, and soil conditions (U.S. Environmental Protection Agency, 2002b). Survival rates of enteric bacteria, those associated with human and animal waste, in the subsurface are generally diminished by higher temperatures, lower nutrient and organic matter content, acidic conditions (low pH values), decreased moisture, and indigenous soil microflora (Gerba and others, 1975); however, filtration is the primary mechanism of bacterial retention in unsaturated soil. Bacteria range in size from 0.2 to 5 microns, which is large enough to become trapped in soil micropores and surface-water film interstices (Bicki and others, 1984; Pekdeger, 1984; Cantor and Knox, 1985; Tchobanoglous and Burton, 1991). Slow soil permeability rates, unsaturated conditions, uniform wastewater distribution to soils, and periodic lulls in system usage further enhance bacterial filtration. Adsorption of bacteria onto clay and soil colloids and sedimentation of particulate matter also contribute to bacterial retention (U.S. Environmental Protection Agency, 2002b).

Studies have shown that most, if not all, pathogenic bacteria are usually retained or die within 2 to 3 feet of the infiltrative surface (McGauhey and Krone, 1967; Bouma and others, 1972; Ayres Associates, 1993; Anderson and others, 1994); however, improper location, design, installation, or operation can result in bacterial contamination of surface or ground water. Situations that are particularly susceptible to bacterial contamination occur when infiltrative surfaces are installed below ground-water surfaces or too close to fractured bedrock, in karst terrain, or within areas that experience seasonally high water tables that rise above the infiltrative surface (Hagedorn, 1982; Bicki and others, 1984). Bacteria that are not retained by the infiltration surface or underlying soils have been observed to survive up to 63 days and can travel over 100 feet in ground water (Gerba and others, 1975).

Table 3. Waterborne pathogens found in human waste and associated diseases

Type	Organism	Disease	Effects
Bacteria	*Escherichia coli (enteropathogenic)*	Gastroenteritis	Vomiting, diarrhea, death in susceptible populations
	Legionella pneumophila	Legionellosis	Acute respiratory illness
	Leptospira	Leptospirosis	Jaundice, fever (Well's disease)
	Salmonella typhi	Typhoid fever	High fever, diarrhea, ulceration of the small intestine
	Salmonella	Salmonellosis	Diarrhea, dehydration
	Shigella	Shigellosis	Bacillary dysentery
	Vibrio cholerae	Cholera	Extremely heavy diarrhea, dehydration
	Yersinia enterolitica	Yersinosis	Diarrhea
Protozoans	*Balantidium coli*	Balantidiasis	Diarrhea, dysentery
	Cryptosporidium	Crypotosporidiosis	Diarrhea
	Entamoeba histolytica	Amoebiasis (Amoebic dysentery)	Prolonged diarrhea with bleeding, abscesses of the liver and small intestine
	Giardia lambia	Giardiasis	Mild to severe diarrhea, nausea, indigestion
	Naegleria fowleri	Amebic meningoencephalitis	Fatal disease; inflammation of the brain
Viruses	Adenovirus (31 types)	Conjunctivitis	Eye, other infections
	Enterovirus (67 types, e.g., polio-, echo-, and Coxsackie viruses)	Gastroenteritis	Heart anomalies, meningitis
	Hepatitis A	Infectious hepatitis	Jaundice, fever
	Norwalk agent	Gastroenteritis	Vomiting, diarrhea
	Reovirus	Gastroenteritis	Vomiting, diarrhea
	Rotavirus	Gastroenteritis	Vomiting, diarrhea

Information from U.S. Environmental Protection Agency (2002a)

Viruses differ from bacteria in that they are not normally present in effluent from on-site wastewater treatment systems. Viruses appear intermittently, in varying quantities, as a result of infected users of on-site wastewater treatment systems, and are therefore difficult to monitor and evaluate. When present, however, enteric viruses can occur in significant numbers, and it has been estimated that infected feces can contain 1×10^6 to 1×10^{10} viral particles per gram (Vaughn and Landry, 1977; Yeager and O'Brien, 1977; Hain and O'Brien, 1979; Harkin and others, 1979; Kowal, 1982; Anderson and others, 1991). Viruses in wastewater effluent typically are retained in soil where they may or may not become inactivated. Viruses that are not inactivated can accumulate in soil, and when conditions change due to increased precipitation or peak effluent flow rates, viruses may be released to ground water. Virus survival rates are decreased by low moisture content, warm temperatures, and high organic content. Virus retention rates are increased by high moisture content, low organic content, small soil particle size, and low pH; however, numerous studies have shown that adsorption is the largest mechanism of virus retention (U.S. Environmental Protection Agency, 2002b). Although some studies found effective virus removal within 1 to 2 feet of soil, most reported that viruses penetrate more than 10 feet through unsaturated soils, and travel distances in ground water up to 1 mile have been reported in karst terrain (Anderson and others, 1991; Ayres Associates, 1993; Higgins and others, 2000). Furthermore, viral survival rates up to 130 days have been reported and survival rates as high as 200 days have been estimated from observed virus mortality rates (Wellings and others, 1975; Schaub and Sorber, 1977; Hain and O'Brien, 1979; Vaughn and Landry, 1980; Vaughn and others, 1981, 1982, 1983; Jansons and others, 1989; Yates and Yates, 1989; Anderson and others, 1991).

Phosphorus, like nitrogen, is a significant plant nutrient that contributes to eutrophication and dissolved oxygen depletion in rivers, lakes, and ponds. Approximately 4 to 8 percent of phosphorus in raw wastewater is removed by sedimentation in septic tanks (Rezek and Cooper, 1980; U.S. Environmental Protection Agency, 1984a, 1984b). The remaining phosphorus passes through the tank, either suspended or in solution, in the form of orthophosphate and organic phosphorus (University of Wisconsin, 1978). Primary factors that affect the amount of phosphorus leached to ground water include soil characteristics, thickness of the unsaturated zone, loading rates, and age of the on-site wastewater treatment system (U.S. Environmental Protection Agency, 2002b). Sorption and precipitation reactions control the fate and transport of phosphorus in soil. At low concentrations (less than 5 mg/L), the phosphate ion is chemisorbed onto calcium mineral surfaces in neutral and alkaline systems, and onto iron and aluminum mineral surfaces in neutral and acidic systems. Phosphate precipitates form as phosphorus concentrations increase (Sikora and Corey, 1976; U.S. Environmental Protection Agency, 2002b). With continued loading, the capacity of the soil to retain phosphorus can be exceeded, allowing phosphorus to move deeper into the soil profile. Ultimately, the retention capacity of the soil is determined by particle-size distribution, mineralogy, oxidation-reduction potential, and pH. Fine-textured, unstructured soils have greater phosphorus retention capacity by allowing wastewater to disperse and contact a greater volume of soil than coarse or highly structured soils. Precipitation reactions in soils with significant concentrations of iron, aluminum, and calcium also increase phosphorus retention. Therefore, the risk of phosphorus contamination is greatest in karst regions, in coarse-textured soils, and in soils without significant concentrations of iron, calcium, or aluminum (U.S. Environmental Protection Agency, 2002b).

Effluent from on-site wastewater treatment systems also can contain a number of toxic organic compounds that can pose serious threats to human health and can interfere with biological processes in the environment. Some of the potential health effects from toxic organic compounds include anemia, increased risk of cancer, reproductive difficulties, and liver, kidney, circulatory, nervous system, or adrenal gland problems. The most prevalent toxic organic compounds found in household wastewater are those from solvents and cleaners, including 1,4-dichlorobenzene, methylbenzene (toluene), dimethylbenzenes (xylenes), 1,1-dichloroethane, 1,1,1-trichloroethane, and dimethylketone (acetone) (U.S. Environmental Protection Agency, 2002b). No known investigations to determine treatment efficiency for toxic organic compounds in individual residential on-site wastewater treatment systems have been conducted; however, a study of a community septic tank found that removal efficiency was related to tank detention time and settling efficiency, and that some removal of low molecular-weight alkylated benzenes (e.g., toluene, xylene) was observed, while almost no removal was noted for higher molecular-weight compounds (DeWalle and others, 1985; U.S. Environmental Protection Agency, 2002b).

Very little research has been performed regarding the behavior of toxic organic compounds in unsaturated soil, although it is known that these compounds can migrate in both gaseous and liquid phases. Unsaturated conditions in drain fields can facilitate release of volatile organic contaminants through gaseous diffusion and volatilization (Wilhelm, 1998; U.S. Environmental Protection Agency, 2002b). Toxic organics in gaseous phases diffuse outward in any direction within unobstructed soil voids, and in liquid phase they follow the movement of the soil solution. Organic toxins that are miscible in water also can migrate with soil water (Preslo and others, 1989; U.S. Environmental Protection Agency, 2002b). Certain toxic organic compounds are not electrochemically retained in unsaturated soil because of their nonpolar nature; however, some of the compounds can be transformed by soil microorganisms into less innocuous forms, depending on oxygen availability. Some retention may be achieved through adsorption by solid organic matter in septic tanks and in soils in the receiving environment. Soils with fine textures, abrupt interfaces of distinctly different textural layers, a lack of fissures and other continuous macropores, and low moisture content also tend to retard movement of toxic organic compounds (Hillel, 1989). Toxic organic compounds that reach an aquifer generally follow the direction of ground-water movement. Some compounds stay near the upper surface of the aquifer and experience significant lateral movement, while other compounds with greater molecular weight may show increased vertical movement (Dagan and Bresler, 1984; U.S. Environmental Protection Agency, 2002b).

Metals occur in effluent from on-site wastewater treatment systems and if ingested, can pose human health risks including physical and mental developmental delays, gastrointestinal illnesses, kidney disease, and neurological problems. Sources of metals in effluent include plumbing systems, vegetable matter, and human excreta. Metals that have been identified in effluent include barium, cadmium, chromium, copper, lead, mercury, nickel, selenium, and zinc (Feige and others, 1975; Bennett and others, 1977; Segall and others, 1979; Whelan and Titmanis, 1982; Cantor and Knox, 1985; U.S. Environmental Protection Agency, 2002b). Mobility and retention of metals in effluent is primarily controlled by adsorption on soil particles and interaction with organic molecules. Mobility of metals below the infiltrative surface is dictated by the cation exchange capacity of the soil and the soil solution pH. Sorption of metals in soil is also reduced by acidic conditions, which can increase the risk of ground-water contamination (Evanko and Dzombak, 1997; Lim and others, 2001; U.S. Environmental Protection Agency, 2002b).

Very little information is available regarding the fate and transport of metals in ground water; however, it is thought that metal movement is complexed with organic ligands formed at or near the infiltrative surface. Therefore, the types and amounts of inorganic and organic ligands entering ground water, their chemical and biochemical reactivity, and their transport behavior (diffusion and/or advection) are important factors (Ayres Associates, 1991; U.S. Environmental Protection Agency, 2002b).

Surfactants are anthropogenic organic compounds commonly used in laundry detergents and soaps to decrease the surface tension of water and to enhance wetting and emulsification. They represent the largest class of anthropogenic organic compounds in domestic wastewater (Dental and others, 1993), and if released into the environment they can mobilize otherwise insoluble organic pollutants, change soil structure, and alter wastewater infiltration rates (U.S.

Environmental Protection Agency, 2002b). Some of the compounds of concern in the surfactant category include linear alkylbenzenesulfonate, methylene blue active substance, alkylbenzenesulfonate, and alkylphenol polyethoxylates (endocrine disruptors). Surfactant molecules concentrate at interfaces of the aqueous system such as air, oily material, and particle surfaces. Their behavior in unsaturated soil depends on surfactant type, soil solution chemistry, organic content of the soil, and degradation rate by microorganisms. Surfactants are biodegradable under aerobic conditions, and soils with higher organic matter content and fine textures generally favor retention of these compounds. Alternatively, surfactants are more stable under anaerobic conditions, and it is thought that little retention of anionic and nonionic surfactants occurs in unsaturated soils having low organic matter content (U.S. Environmental Protection Agency, 2002b).

Although not typically problematical in properly functioning on-site wastewater treatment systems, biodegradable organic material and total suspended solids can be constituents of concern in residential wastewater if excessive quantities of these materials migrate beyond the septic tank as a result of poor maintenance (U.S. Environmental Protection Agency, 2002b). Biodegradable organic material creates biochemical oxygen demand which can decrease dissolved oxygen concentrations in surface water, cause odor and taste problems in wells, and can cause metals to leach from soil and rock into ground water and surface water. Total suspended solids can clog infiltrative surfaces and soil interstices, leading to surface seepage of wastewater or plumbing fixture backup. If released to surface waters, total suspended solids cause cloudiness and can result in the development of sludge layers that are harmful to aquatic organisms. In addition, the release of biological oxygen demand and total suspended solids in close proximity to surface waters and drinking water wells may contribute to other problems related to toxic and pathogenic pollutants (U.S. Environmental Protection Agency, 2002b). Under proper operating and site conditions, on-site wastewater treatment systems remove most biodegradable organic compounds and suspended solids, and the risk of ground-water contamination from these constituents is low (University of Wisconsin, 1978; Anderson and others, 1994). Most solids are removed in the septic tank, and the majority of remaining particulate biological oxygen demand is removed at the infiltrative surface and biomat. Colloidal and dissolved biological oxygen demand that passes through the infiltrative surface is further treated through aerobic biological processes in the vadose zone (U.S. Environmental Protection Agency, 2002b).

A final category of potential contaminants found in effluent from residential on-site wastewater treatment systems are those consisting of dissolved inorganic parameters such as chloride, sulfide, boron, sodium, sulfate, and potassium. Chlorides are highly soluble, nonreactive in soil, and can leach readily to ground water. In excess, both chlorides and sulfides can cause taste and odor problems in drinking water. Solutes including boron, sodium, chlorides, and sulfate also may limit wastewater reuse options, and sodium and potassium can damage soil structure and diminish performance of the subsurface wastewater infiltration system (U.S. Environmental Protection Agency, 2002b).

On-Site Wastewater Treatment Systems in Karst Limestone

Karst terrain is characterized by sinkholes, caves, large springs, dry valleys, sinking streams, thin soils, and efficient flow of ground water through conduits in dissolved bedrock (Veni and others, 2001). More specifically, karst limestone aquifers are characterized by solution enhanced fractures, caverns, and conduits which allow extremely rapid ground-water flow velocities in an anisotropic subsurface environment. In karst terrain, surface water commonly drains rapidly into the subsurface in recharge areas, sometimes disappearing entirely at sinkholes or through stream reaches underlain by highly dissolved fractures or caverns in the limestone. Down-gradient ground-water flow occurs through a network of fractures, partings, and caves, and tracing investigations in the Black Hills area have demonstrated that ground water can move large distances at fast velocities in this setting (Rahn, 1971; Rahn and Gries, 1973; Greene, 1997, 1999; L. Putnam and A. Long, Rapid City, S. Dak., U.S. Geological Survey, oral commun., 2007). These previous investigations documented that dye in ground water traveled approximately 6 miles in about 1 month in the karstic Madison aquifer, and in some cases, dye injected into surface waters in streamflow loss zones reached public water-supply wells in the Madison aquifer within a matter of hours.

In karst limestone settings pathogenic microorganisms or other contaminants can be transported rapidly down-gradient to drinking-water wells serving cities and towns that may depend on the karst limestone aquifer as their primary water source. A documented example of contamination from wastewater in karst terrain in southeastern Minnesota resulted in 11 cases of typhoid fever and the death of one person. In this particular example, dye introduced into a sinkhole that was receiving partially treated sewage from a small village reappeared within 4 hours in the well where typhoid first occurred (Manduca, 2000). In the Black Hills of South Dakota in 1993, an infection of *Giardia* occurred from a well in the karstic Madison aquifer (Daly, 1993). Dye tracing investigations by the U.S. Geological Survey later demonstrated that surface water entering nearby swallow holes reached the well in a matter of hours (L. Putnam and A. Long, Rapid City, S. Dak., U.S. Geological Survey, oral commun., 2007).

Relatively long-term virus and bacteria survival rates further elevate the risk of drinking-water contamination in karst settings. For example, recent investigations in karst streams and springs in Arkansas by Davis and others (2005) documented survival rates of *E. coli* for the entire duration of their 75-day study, and from bacterial die-off rates they estimated a 135-day total survival period for this strain of bacteria. The threat of contamination in karst aquifers is further elevated during storm events when bacteria and virus levels in ground water and springs may increase by several orders of magnitude and may rapidly percolate through thin soils and into the underlying karst bedrock. Such increases may result from mobilization of viable pathogenic organisms that reside in stream and spring sediments, from surface runoff, and from drain-field effluent (Veni and others, 2001; Davis and others, 2005). Degradation of water quality from contaminants such as nitrate-nitrogen, toxic organic compounds, or other pollutants also is a significant concern in karst aquifers, possibly causing permanent damage or resulting in long-term contamination of an aquifer that could take years or decades to mitigate.

Several approaches have been developed in the attempt to numerically quantify aquifer sensitivity as an aid in the protection of ground-water resources, although most have not accurately addressed hydrologic features and characteristics unique to karst aquifers. One of the more frequently employed models is the DRASTIC method of Aller and others (1987), which is based on characteristics of the hydrogeologic setting, rating parameters such as aquifer medium and hydraulic conductivity, and weighted values for each of the identified hydrogeologic characteristics which collectively yield a sensitivity index map for the recharge area under investigation. This approach has been performed in a variety of hydrogeologic settings; however, there are no provisions for significant karst characteristics such as swallow holes, and results have not accurately reflected pollution potential values for recharge areas in karst terrain (Davis and others, 2000). Subsequent to development of the DRASTIC method, Davis and others (2000) published a modification of this approach known as KARSTIC which includes additional provisions for karst terrain and which can be used for general ground-water sensitivity characterizations needed for land management and planning decisions in this hydrogeologic setting. Miller (2005) employed an approach similar to KARSTIC in a study of susceptibility of the Madison aquifer on the eastern flank of the Black Hills, and Miller (2005) furthermore attempted to determine specific relationships between major structural features, hydrogeologic characteristics, stratigraphic information, and ground-water flow paths in this karst aquifer.

Another approach for quantification of aquifer sensitivity and protection of ground water in karst terrain was developed by the Swiss Agency for the Environment, Forests, and Landscape (1998) which is based on ground-water sensitivity maps of spring and well catchment areas. The method, titled EPIK, incorporates four hydrogeological parameters including epikarstic development, protective cover properties, infiltration conditions, and karstic network development. A protection index is calculated from weighted values of these four parameters allowing construction of a map delineating ground-water protection zones. Although useful for sensitivity characterization of karst aquifer recharge areas, these approaches do not specifically address or attempt to simulate contaminants that are released into the environment from specific sources such as on-site wastewater treatment systems; rather, they are used for general sensitivity analyses and protection of aquifer recharge areas from all types of contaminants.

A study of the Edwards aquifer in the Barton Springs area near Austin, Texas, attempted to quantify the amount of nitrogen that infiltrates into this karst limestone aquifer specifically from on-site wastewater treatment systems (Santos and Associates, 1995). A computer model titled Groundwater Loading Effects of Agricultural Management Systems (GLEAMS) was employed to calculate the movement of water and nutrients through plant systems and through soil within the vadose zone. To calculate water movement GLEAMS utilizes daily rainfall rates, surface runoff estimates, air temperatures, solar radiation, wind velocities, evapotranspiration estimates, plant growth rates, and storage within soil pores. To estimate nitrogen movement GLEAMS simulates vegetative uptake, sediment and runoff transport, biological fixation, ammonification, nitrification, volatilization, denitrification, soil storage, and deep percolation, as well as transformations between organic nitrogen, ammonia, nitrite-nitrogen, nitrate-nitrogen, and gases. There were 4,806 on-site wastewater treatment systems in the study area, and model results estimated that 2,550 to 17,000 pounds of nitrogen per year were infiltrating to the bedrock aquifer as a result of these systems. Dividing this estimated annual nitrogen load by the average annual volume of recharge to the aquifer resulted in a nitrogen concentration increase from 0.02 to 0.16 mg/L, although the authors noted that actual nitrogen concentrations would vary locally due to incomplete mixing in the aquifer (Santos and Associates, 1995).

A number of previous investigations have provided recommendations and protective measures for siting and constructing on-site wastewater treatment systems in karst aquifer recharge areas (Rahn and others, 1993; Santos and Associates, 1995; Zokaites, 1997; Veni and others, 2001). These recommendations have included the following:

- Reducing hydraulic loads through requirements for larger drain fields;
- Installation of low-flow plumbing fixtures;
- Routine inspection and maintenance of on-site wastewater treatment systems located in critical recharge areas;
- Public education to increase owner awareness of proper operation and maintenance;
- Increasing soil depth requirements below drain fields and remediation of existing systems with inadequate soils;
- Increasing residential lot sizes and spacing requirements;
- Reducing nitrogen loads;
- Requiring lined evapotranspiration systems or aerobic pretreatment;
- Requiring greater setback distances from karst features, springs, streams, and other surface-water bodies;
- Forming community systems;
- Establishment of local ordinances; and
- Offering incentives to upgrade or replace systems that may be deficient for a variety of reasons.

In a study focused on design and construction criteria for on-site wastewater treatment systems within the recharge areas of the Madison and Minnelusa aquifers near Rapid City, South Dakota, Rahn and others (1993) recommended: (1) rigid enforcement of existing regulations, (2) completion of a detailed engineering assessment conducted by a registered professional engineer to demonstrate proper conditions at each site, and (3) usage of septic tanks with two compartments on outcrop areas of the Pahasapa (Madison) Limestone. These authors also recommended connection of some residences to local municipal sewer lines and establishment of an aquifer protection program including design and siting specifications in sensitive areas.

RESULTS

Alluvial Deposits

Private Residence, Hill City, South Dakota

This study site consisted of a continuously occupied, family residence with an on-site wastewater treatment system that was approximately 3 years old at the time of the study. There were five residents living in the home, including two adults and three children. The on-site wastewater treatment system had no known functional problems, and it had never been pumped or serviced in any fashion. The general hydrogeologic setting at the site consisted of unconsolidated alluvial sediments overlying Precambrian lithologies consisting of thin-bedded to laminated quartz-biotite-garnet phyllite and schist (Ratte' and Wayland, 1969). Clasts within the alluvial deposit ranged from clay- to boulder-sized, and consisted of reworked Precambrian lithologies. Ground water occurred in the alluvial sediments, which are laterally confined by schist forming the walls of a narrow valley within which the residence is located. The soils underlying the property are classified as Cordeston loam, which typically extends to a depth of about 60 inches and is composed of loam with coarse fragments and sand in some areas. This soil is considered to have moderate limitations for on-site wastewater treatment system drain fields because of moderate permeability rates and shrink-swell potential (Ensz, 1990).

Topographically, the site occurs in a small, narrow valley comprising a minor, ephemeral tributary to Spring Creek which is located approximately 1,300 feet to the east. One on-site wastewater treatment system occurs up-gradient from this study site, located approximately 750 feet to the west within the same alluvial deposit that underlies the study site. No other on-site wastewater treatment systems occur within the local watershed up-gradient from this study site.

Figure 9 shows the configuration of the water table at the site, location of the on-site wastewater treatment system drain field, and location of the installed monitoring wells. Monitoring well R20-2002-10 represents background conditions up-gradient from the on-site wastewater treatment system drain field. Monitoring wells R20-2002-18 and R20-2002-11 were adjacent to and down-gradient from the drain field and were located to intercept potential contaminants migrating with the drain-field effluent. Ground-water depths were recorded before each water-quality sampling event (table 4), and ranged from about 14 to 23 feet below ground surface during the study. Water levels in the monitoring wells also fluctuated as much as 6 feet in elevation due to seasonal changes in climatic conditions. The water-table gradient shown in figure 9, from measurements taken on August 23, 2002, drops approximately 8 feet across a lateral distance of about 200 feet.

Selected results of analyses for samples from this study locality are summarized in table 5 as an aid to the reader. Results of all water-quality analyses for this site and all other sites in this investigation are presented in table 6. Results of laboratory analyses for *E. coli* bacteria are reported as the “most probable number” (MPN) of individual organisms per 100 milliters of water. Fecal coliform bacteria are measured as colonies per 100 milliliters of water. *E. coli* bacteria were detected in monitoring wells R20-2002-18 and R20-2002-11 during the spring sampling event, although only 1.0 MPN/100 milliliters was identified in the samples from each well. During the fall sampling event, 1.0 MPN/100 milliliters of *E. coli* and 4 colonies/100 milliliters of fecal coliform bacteria were detected in monitoring well R20-2002-11. Nitrate-nitrogen concentrations ranged from 0.1 to 0.5 mg/L in monitoring well R20-2002-10 which represents the background conditions of ground water entering the site, and nitrate-nitrogen concentrations reached a maximum of 1.4 mg/L in monitoring well R20-2002-18 immediately adjacent to the drain field. Nitrate-nitrogen concentrations also were slightly higher in all wells during the May sampling event when precipitation and ground-water levels were highest (app. A; tables 4, 6).

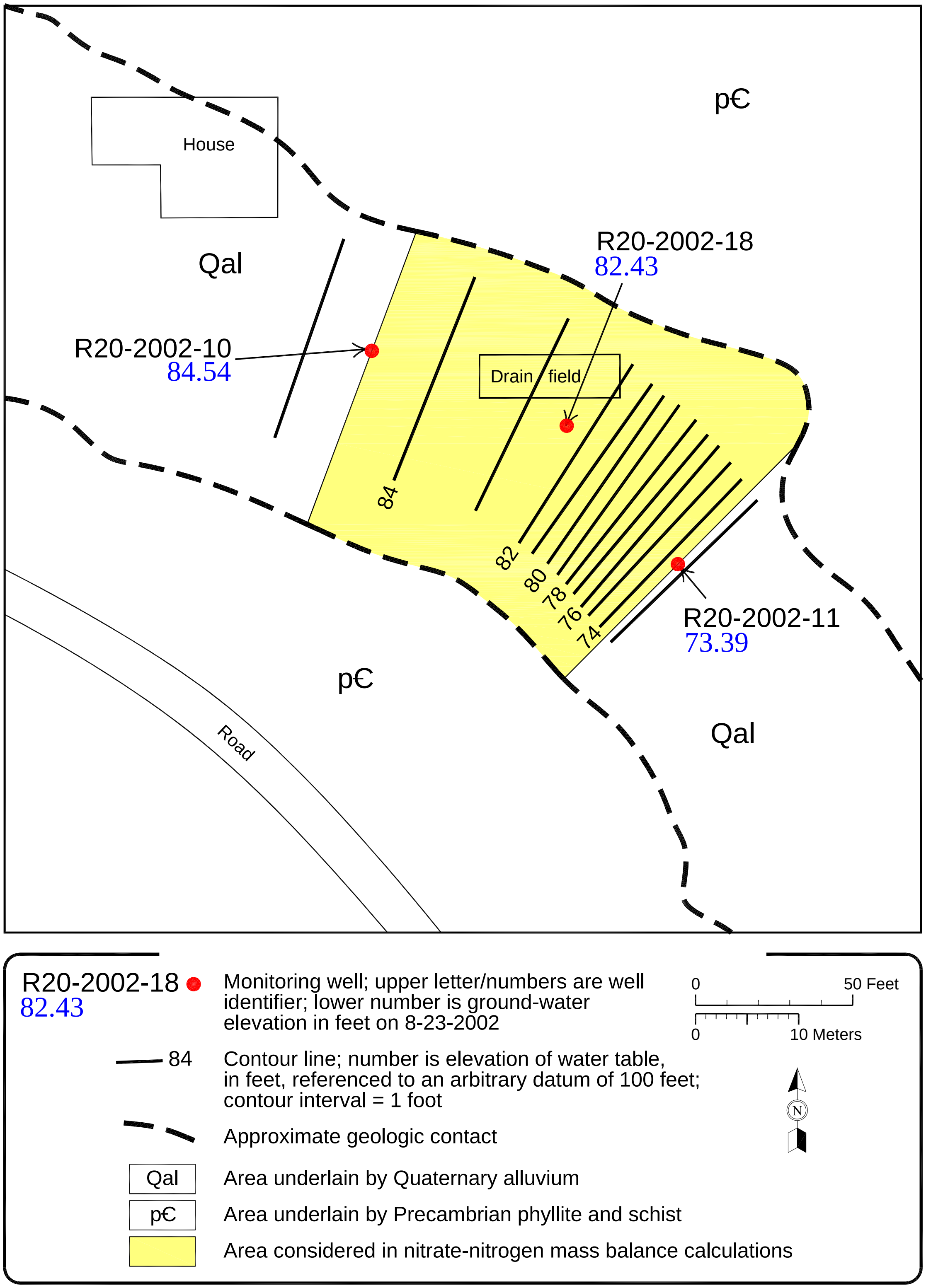


Figure 9. **Map showing ground-water elevations and locations of monitoring wells near an on-site wastewater treatment system drain field in an unconsolidated alluvial deposit near Hill City.**

Table 4. Water-level measurements and relative elevations of monitoring wells and other sample locations

Well/ water body	Date	Time	Relative casing-top elevation (ft)[1]	Depth to water from casing top (ft)	Depth to water from ground surface (ft)	Relative water-level elevation (ft)
Private residence, Hill City						
well R20-2002-10	07-24-2002	12:00 PM	101.54	16.85	16.85	84.69
	07-24-2002	2:00 PM	101.54	19.31	19.31	82.23
	08-23-2002	11:00 AM	101.54	17.00	17.00	84.54
	09-23-2002	3:00 PM	101.54	16.99	16.99	84.55
	09-24-2002	11:45 AM	101.54	16.98	16.98	84.56
	02-12-2003	3:15 PM	101.54	16.97	16.97	84.57
	02-13-2003	10:45 AM	101.54	17.00	17.00	84.54
	05-05-2003	11:00 AM	101.54	14.80	14.80	86.74
	05-06-2003	10:20 AM	101.54	14.78	14.78	86.76
	08-20-2003	11:55 AM	101.54	16.69	16.69	84.85
	08-21-2003	12:15 PM	101.54	16.69	16.69	84.85
well R20-2002-18	07-24-2002	2:00 PM	97.88	15.49	15.49	82.39
	08-23-2002	11:00 AM	97.88	15.45	15.45	82.43
	09-23-2002	3:15 PM	97.88	14.40	14.40	83.48
	09-24-2002	1:00 PM	97.88	15.41	15.41	82.47
	02-12-2003	3:30 PM	97.88	15.40	15.40	82.48
	02-13-2003	10:55 AM	97.88	15.44	15.44	82.44
	05-05-2003	11:00 AM	97.88	14.00	14.00	83.88
	05-06-2003	11:00 AM	97.88	14.03	14.03	83.85
	08-20-2003	12:00 PM	97.88	15.37	15.37	82.51
	08-21-2003	12:45 PM	97.88	15.37	15.37	82.51
well R20-2002-11	07-24-2002	2:00 PM	96.90	22.90	22.90	74.00
	08-23-2002	11:10 AM	96.90	23.51	23.51	73.39
	09-23-2002	4:00 PM	96.90	23.32	23.32	73.58
	09-24-2002	2:05 PM	96.90	23.35	23.35	73.55
	02-12-2003	3:45 PM	96.90	23.42	23.42	73.48
	02-13-2003	11:00 AM	96.90	23.44	23.44	73.46
	05-05-2003	11:00 AM	96.90	17.40	17.40	79.50
	05-06-2003	11:30 AM	96.90	17.07	17.07	79.83
	08-20-2003	12:05 PM	96.90	22.55	22.55	74.35
	08-21-2003	1:45 PM	96.90	22.53	22.53	74.37

Table 4 – continued

Well/ water body	Date	Time	Relative casing-top elevation (ft)[1]	Depth to water from casing top (ft)	Depth to water from ground surface (ft)	Relative water-level elevation (ft)
Private residence, Johnson Siding						
well R20-2002-21	07-25-2002	2:25 PM	93.36	4.95	4.95	88.41
	07-30-2002	2:40 PM	93.36	4.87	4.87	88.49
	08-15-2002	3:20 PM	93.36	4.70	4.70	88.66
	10-02-2002	11:00 AM	93.36	5.15	5.15	88.21
	10-03-2002	11:20 AM	93.36	5.27	5.27	88.09
	01-29-2003	3:15 PM	93.36	5.02	5.02	88.34
	01-30-2003	12:15 PM	93.36	5.15	5.15	88.21
	05-12-2003	11:00 AM	93.36	5.21	5.21	88.15
	05-13-2003	11:00 AM	93.36	5.19	5.19	88.17
	08-12-2003	2:50 PM	93.36	4.71	4.71	88.65
	08-13-2003	11:00 AM	93.36	4.72	4.72	88.64
well R20-2002-22	07-25-2002	2:35 PM	93.24	4.94	4.94	88.30
	07-30-2002	2:35 PM	93.24	4.88	4.88	88.36
	08-15-2002	3:25 PM	93.24	4.71	4.71	88.53
	10-02-2002	11:25 AM	93.24	5.16	5.16	88.08
	10-03-2002	12:55 PM	93.24	5.28	5.28	87.96
	01-29-2003	3:40 PM	93.24	5.09	5.09	88.15
	01-30-2003	12:45 PM	93.24	5.17	5.17	88.07
	05-12-2003	11:15 AM	93.24	5.24	5.24	88.00
	05-13-2003	11:30 AM	93.24	5.25	5.25	87.99
	08-12-2003	3:00 PM	93.24	4.76	4.76	88.48
	08-13-2003	11:10 AM	93.24	4.78	4.78	88.46
well R20-2002-20	07-25-2002	2:20 PM	99.65	12.70	12.70	86.95
	07-30-2002	2:45 PM	99.65	12.70	12.70	86.95
	08-15-2002	3:15 PM	99.65	15.54	15.54	84.11
	10-02-2002	10:30 AM	99.65	12.89	12.89	86.76
	10-03-2002	10:00 AM	99.65	12.90	12.90	86.75
	01-29-2003	2:45 PM	99.65	12.51	12.51	87.14
	01-30-2003	11:25 AM	99.65	12.55	12.55	87.10
	05-12-2003	10:45 AM	99.65	12.87	12.87	86.78
	05-13-2003	10:00 AM	99.65	12.88	12.88	86.77
	08-12-2003	2:35 PM	99.65	12.38	12.38	87.27
	08-13-2003	10:45 AM	99.65	12.41	12.41	87.24

Table 4 – continued

Well/ water body	Date	Time	Relative casing-top elevation (ft)[1]	Depth to water from casing top (ft)	Depth to water from ground surface (ft)	Relative water-level elevation (ft)
Rocky Knolls Golf Course, Custer						
well R20-2002-14	07-15-2002	2:00 PM	100.15	3.85	3.85	96.30
	08-23-2002	1:35 PM	100.15	3.95	3.95	96.20
	09-30-2002	3:10 PM	100.15	3.55	3.55	96.60
	10-01-2002	11:40 AM	100.15	3.04	3.04	97.11
	01-27-2003	3:00 PM	100.15	4.75	4.75	95.40
	01-28-2003	12:15 PM	100.15	4.75	4.75	95.40
	05-07-2003	12:00 PM	100.15	0.82	0.82	99.33
	05-08-2003	11:30 AM	100.15	0.81	0.81	99.34
	07-28-2003	11:15 AM	100.15	2.43	2.43	97.72
	07-29-2003	2:00 PM	100.15	2.40	2.40	97.75
well R20-2002-12	07-15-2002	1:45 PM	100.00	5.45	5.45	94.55
	08-23-2002	1:25 PM	100.00	4.04	4.04	95.96
	09-30-2002	2:40 PM	100.00	3.45	3.45	96.55
	10-01-2002	10:10 AM	100.00	3.72	3.72	96.28
	01-27-2003	3:30 PM	100.00	4.85	4.85	95.15
	01-28-2003	11:45 AM	100.00	4.95	4.95	95.05
	05-07-2003	11:45 AM	100.00	0.72	0.72	99.28
	05-08-2003	11:00 AM	100.00	0.83	0.83	99.17
	07-28-2003	11:10 AM	100.00	2.49	2.49	97.51
	07-29-2003	1:30 PM	100.00	2.49	2.49	97.51
well R20-2002-15	07-15-2002	2:30 PM	106.13	9.85	9.85	96.28
	08-23-2002	1:25 PM	106.13	10.27	10.27	95.86
	09-30-2002	3:25 PM	106.13	9.51	9.51	96.62
	10-01-2002	2:50 PM	106.13	9.56	9.56	96.57
	01-27-2003	4:00 PM	106.13	12.35	12.35	93.78
	01-28-2003	1:15 PM	106.13	12.35	12.35	93.78
	05-07-2003	12:15 PM	106.13	8.46	8.46	97.67
	05-08-2003	12:00 PM	106.13	8.41	8.41	97.72
	07-28-2003	11:00 AM	106.13	9.15	9.15	96.98
	07-29-2003	3:00 PM	106.13	9.21	9.21	96.92

Table 4 – continued

Well/ water body	Date	Time	Relative casing-top elevation (ft)[1]	Depth to water from casing top (ft)	Depth to water from ground surface (ft)	Relative water-level elevation (ft)
Hill City wastewater treatment lagoons						
well R20-2002-05	07-13-2002	10:30 AM	101.91	5.50	5.50	96.41
	07-14-2002	10:00 AM	101.91	5.75	5.75	96.16
	08-27-2002	12:10 PM	101.91	5.48	5.48	96.43
	09-11-2002	3:20 PM	101.91	5.39	5.39	96.52
	09-16-2002	5:24 PM	101.91	5.47	5.47	96.44
	09-17-2002	10:00 AM	101.91	5.45	5.45	96.46
	02-04-2003	12:40 PM	101.91	5.19	5.19	96.72
	02-05-2003	1:00 PM	101.91	6.83	6.83	95.08
	04-30-2003	1:00 PM	101.91	4.87	4.87	97.04
	05-01-2003	11:00 AM	101.91	5.23	5.23	96.68
	08-19-2003	1:10 PM	101.91	5.39	5.39	96.52
	08-20-2003	11:00 AM	101.91	5.48	5.48	96.43
	07-06-2005	10:15 AM	101.91	5.40	5.40	96.51
well R20-2002-04	07-13-2002	10:15 AM	100.00	6.60	3.60	93.40
	07-14-2002	9:10 AM	100.00	6.80	3.80	93.20
	08-27-2002	11:45 AM	100.00	6.64	3.64	93.36
	09-11-2002	3:20 PM	100.00	6.30	3.30	93.70
	09-16-2002	5:00 PM	100.00	6.19	3.19	93.81
	09-18-2002	2:50 PM	100.00	6.10	3.10	93.90
	09-19-2002	10:30 AM	100.00	5.81	2.81	94.19
	02-04-2003	1:10 PM	100.00	6.43	3.43	93.57
	02-05-2003	11:30 AM	100.00	6.40	3.40	93.60
	04-30-2003	1:30 PM	100.00	5.71	2.71	94.29
	05-01-2003	12:00 PM	100.00	5.28	2.28	94.72
	08-19-2003	12:25 PM	100.00	6.13	3.13	93.87
	08-20-2003	10:30 AM	100.00	6.04	3.04	93.96
	07-06-2005	11:05 AM	100.00	6.17	3.17	93.83

Table 4 – continued

Well/ water body	Date	Time	Relative casing-top elevation (ft)[1]	Depth to water from casing top (ft)	Depth to water from ground surface (ft)	Relative water-level elevation (ft)
Hill City wastewater treatment lagoons – continued						
well R20-2002-03	07-13-2002	10:00 AM	94.70	6.70	4.20	88.00
	07-14-2002	9:00 AM	94.70	6.75	4.25	87.95
	08-27-2002	11:30 AM	94.70	6.77	4.27	87.93
	09-11-2002	3:05 PM	94.70	6.48	3.98	88.22
	09-16-2002	4:25 PM	94.70	6.53	4.03	88.17
	09-18-2002	2:29 PM	94.70	6.54	4.04	88.16
	09-19-2002	1:00 PM	94.70	6.45	3.95	88.25
	02-03-2003	2:45 PM	94.70	6.48	3.98	88.22
	02-04-2003	12:10 PM	94.70	6.54	4.04	88.16
	04-28-2003	2:30 PM	94.70	5.90	3.40	88.80
	04-29-2003	1:30 PM	94.70	5.86	3.36	88.84
	08-18-2003	12:00 PM	94.70	6.60	4.10	88.10
	08-19-2003	12:15 PM	94.70	6.57	4.07	88.13
	07-06-2005	11:37 AM	94.70	6.56	4.06	88.14
well R20-2002-17	08-27-2002	11:00 AM	96.67	11.04	8.38	85.63
	09-11-2002	2:45 PM	96.67	11.15	8.49	85.52
	09-16-2002	3:45 PM	96.67	10.86	8.20	85.81
	09-18-2002	3:45 PM	96.67	10.82	8.16	85.85
	09-19-2002	1:50 PM	96.67	10.79	8.13	85.88
	02-04-2003	11:40 AM	96.67	12.70	10.04	83.97
	02-05-2003	10:45 AM	96.67	11.76	9.10	84.91
	04-28-2003	1:30 PM	96.67	10.23	7.57	86.44
	04-29-2003	11:15 AM	96.67	10.37	7.71	86.30
	08-18-2003	11:45 AM	96.67	11.26	8.60	85.41
	08-19-2003	11:30 AM	96.67	11.31	8.65	85.36
	07-06-2005	12:00 PM	96.67	11.03	8.37	85.64

Table 4 – continued

Well/ water body	Date	Time	Relative casing-top elevation (ft)[1]	Depth to water from casing top (ft)	Depth to water from ground surface (ft)	Relative water-level elevation (ft)
Hill City wastewater treatment lagoons – continued						
well R20-2002-02	07-11-2002	1:25 PM	86.64	10.71	8.63	75.93
	07-12-2002	11:56 AM	86.64	8.75	6.67	77.89
	07-13-2002	9:50 AM	86.64	8.90	6.82	77.74
	07-14-2002	8:50 AM	86.64	8.84	6.76	77.80
	08-27-2002	10:30 AM	86.64	8.92	6.84	77.72
	09-11-2002	2:30 PM	86.64	8.78	6.70	77.86
	09-16-2002	2:58 PM	86.64	8.76	6.68	77.88
	09-18-2002	3:15 PM	86.64	8.62	6.54	78.02
	09-23-2002	2:30 PM	86.64	8.70	6.62	77.94
	09-24-2002	10:00 AM	86.64	8.70	6.62	77.94
	02-03-2003	2:15 PM	86.64	9.08	7.00	77.56
	02-04-2003	11:00 AM	86.64	9.10	7.02	77.54
	04-28-2003	1:00 PM	86.64	8.31	6.23	78.33
	04-29-2003	10:00 AM	86.64	8.32	6.24	78.32
	08-18-2003	11:15 AM	86.64	8.81	6.73	77.83
	08-19-2003	11:00 AM	86.64	8.81	6.73	77.83
	07-06-2005	12:17 PM	86.64	8.67	6.59	77.97
Spring Creek near well R20-2002-05	07-24-2002	---	---	---	---	96.62
	07-06-2005	---	---	---	---	96.90
Spring Creek near well R20-2002-04	07-06-2005	---	---	---	---	92.46
Spring Creek near well R20-2002-03	07-24-2002	---	---	---	---	87.33
	07-06-2005	---	---	---	---	87.17
Spring Creek near well R20-2002-02	07-24-2002	---	---	---	---	77.37
	07-06-2005	---	---	---	---	77.41
south lagoon	07-06-2005	---	---	---	---	100.98
middle lagoon	07-06-2005	---	---	---	---	97.36
north lagoon	07-06-2005	---	---	---	---	88.88

[1] Casing top elevations were not surveyed relative to mean sea level. Rather, each project site (i.e., Private residence, Johnson Siding) had a datum established with an assumed elevation of 100 feet. Therefore, the water elevations between project sites cannot be directly correlated.

Table 5. Selected results of analyses for the residential on-site wastewater treatment system at Hill City

Monitoring well	Constituent	Concentration
R20-2002-10	Total phosphorus	10.0 milligrams per liter
R20-2002-18	*E. coli*	1.0/100 milliliters[1]
	Nitrate-nitrogen	1.4 milligrams per liter
R20-2002-11	*E. coli*	1.0/100 milliliters[1]
	Fecal coliform	4.0/100 milliliters[2]

[1] /100 milliliters = most probable number per 100 milliliters

[2] /100 milliliters = colonies per 100 milliliters

Concentrations for total phosphorus from the site varied from 0.28 to 10.0 mg/L which was the greatest concentration for total phosphorus from any source that was sampled during the entire project (table 6). Total phosphorus concentrations were greatest in monitoring wells R20-2002-10 and R20-2002-18 which are up-gradient from and adjacent to the drain field, respectively, and lowest in monitoring well R20-2002-11 which is near the down-gradient edge of the property. According to the landowner, the lawn around the residence was never fertilized; however, it could be possible that other factors may have affected total phosphorus concentrations. Other analyzed water-quality parameters at the site such as chloride, ammonia, and conductivity did not appear to display anomalous or significant trends or concentrations, although conductivity concentrations were slightly lower in all samples collected during May 2003 when precipitation and ground-water levels were relatively higher. Ammonia concentrations were slightly greater in all samples collected in February 2003 when precipitation levels were low. Caffeine was not detected at the site (table 6).

Private Residence, Johnson Siding, South Dakota

This site consisted of a continuously occupied family residence with an on-site wastewater treatment system that was approximately 30 years old at the time of the study. The residence was occupied by three adult inhabitants. The on-site wastewater treatment system had no known functional problems, and according to the landowner it had been pumped and inspected approximately every 3 years for the life of the system. The general hydrogeologic setting at the site consists of unconsolidated alluvial sediments overlying Precambrian quartzite, siliceous schist, and minor chert (DeWitt and others, 1989). Clasts within the alluvial deposit range from clay- to boulder-sized, and consist of reworked Precambrian lithologies. The alluvial sediments

Table 6. Results of water-sample analyses

Site name and sample source	Location	Date collected	Time collected	Nitrate plus nitrite as N (mg/L)[1]	Total Kjeldahl nitrogen (mg/L)	Ammonia (mg/L)	Total phosphorous (mg/L)	Dissolved phosphorous (mg/L)	Chloride (mg/L)	*E. coli* (/100 ml)[2]	Fecal coliform (/100 ml)[3]	Caffeine (µg/L)[4]	pH[5]	Temperature (degrees Celsius)[5]	Conductivity (µs)[6]
Private residence, Hill City															
well R20-2002-10	NE¼NE¼NW¼NE¼ sec. 36, T. 1 S., R. 4 E.	09-24-02	12:00 PM	<0.1	<0.32	<0.02	0.598	0.051	6	<1	<2	<0.700	8.03	11.2	319
		02-13-03	10:45 AM	0.1	0.44	0.23	6.6	0.036	5	<1	<2	---	7.16	7.7	370
		05-06-03	12:30 PM	0.5	0.47	0.13	10	0.048	5	<1	<10	---	6.87	7.5	304
		08-21-03	12:15 PM	0.4	<0.11	<0.02	1.02	0.048	4	<1	<10	---	6.78	9.8	324
well R20-2002-18	NE¼NE¼NW¼NE¼ sec. 36, T. 1 S., R. 4 E.	09-24-02	1:10 PM	0.1	0.44	0.11	3.91	0.093	6	<1	<2	<0.700	8.23	12.2	377
		02-13-03	12:00 PM	0.4	0.42	0.25	2.2	0.031	5	<1	<2	---	6.98	7.8	363
		05-06-03	11:15 AM	1.4	0.42	<0.02	1.07	0.058	6	1.0	<10	---	6.47	6.9	288
		08-21-03	12:45 PM	0.5	0.16	0.18	4.5	0.053	5	<1	<10	---	6.66	10.6	331
well R20-2002-11	NE¼NE¼NW¼NE¼ sec. 36, T. 1 S., R. 4 E.	09-24-02	2:15 PM	0.7	<0.32	<0.02	0.5	0.043	5	1.0	4	<0.700	6.54	10.3	347
		02-13-03	12:30 PM	0.5	0.44	0.17	1.66	0.041	6	<1	<2	---	6.76	8.6	350
		05-06-03	11:45 AM	1.1	0.16	<0.02	0.282	0.055	5	1.0	<10	---	6.69	7.2	281
		08-21-03	1:45 PM	0.5	<0.11	<0.02	1.23	0.05	4	<1	<10	---	6.76	11.3	327
Private residence, Johnson Siding															
well R20-2002-21	NE¼NW¼SW¼NW¼ sec. 05, T. 1 N., R. 6 E.	10-03-02	11:30 AM	0.2	<0.32	0.06	0.998	0.012	<3	<1	<10	<0.700	7.38	11.9	375
		01-30-03	12:30 PM	0.1	0.22	0.07	1.21	0.008	4	<1	<2	---	7.50	4.2	378
		05-13-03	11:00 AM	0.1	<0.11	<0.02	0.928	<0.002	3	<1	<10	---	7.72	6.7	397
		08-13-03	11:45 AM	0.1	0.15	0.04	1.07	0.017	<3	<1	<10	---	7.10	14.2	376
well R20-2002-22	NE¼NW¼SW¼NW¼ sec. 05, T. 1 N., R. 6 E.	10-03-02	12:35 PM	0.2	0.35	0.02	1.52	0.011	3	<1	<10	<0.700	7.44	11.6	368
		10-03-02[7]	1:15 PM	0.2	<0.32	0.05	0.728	0.014	<3	1.0	<10	<0.700	7.22	11.3	368
		01-30-03	1:00 PM	0.1	0.47	0.15	4.05	0.008	3	<1	<2	---	7.65	3.8	381
		01-30-03[8]	1:30 PM	<0.1	<0.10	<0.02	0.004	0.003	<3	<1	<2	---	7.09	13.2	1.6
		05-13-03	11:30 AM	0.1	0.27	<0.02	1.71	<0.002	4	<1	<10	---	7.68	6.4	414
		05-13-03[8]	12:00 PM	<0.1	<0.11	<0.02	<0.002	<0.002	<3	<1	<10	---	6.28	17.6	1.2
		08-13-03	12:15 PM	<0.1	0.58	0.07	2.31	0.011	3	<1	<10	---	7.45	13.9	377
		08-13-03[7]	12:30 PM	<0.1	0.92	0.09	2.61	0.015	<3	<1	<10	---	7.45	13.9	377
well R20-2002-20	NE¼NW¼SW¼NW¼ sec. 05, T. 1 N., R. 6 E.	10-03-02	10:35 AM	0.1	0.32	0.18	0.164	0.012	7	<1	<10	<0.700	8.83	11.8	295
		01-30-03	12:00 PM	0.9	0.19	0.18	1.26	0.004	4	<1	<2	---	7.09	8.8	389
		05-13-03	10:15 AM	1.3	<0.11	<0.02	1.16	<0.002	5	<1	<10	---	7.47	8.2	422
		08-13-03	11:20 AM	1.1	0.3	0.09	1.38	0.015	4	<1	<10	---	7.39	11.2	421

Table 6 – continued

Site name and sample source	Location	Date collected	Time collected	Nitrate plus nitrite as N (mg/L)[1]	Total Kjeldahl nitrogen (mg/L)	Ammonia (mg/L)	Total phosphorous (mg/L)	Dissolved phosphorous (mg/L)	Chloride (mg/L)	*E. coli* (/100 ml)[2]	Fecal coliform (/100 ml)[3]	Caffeine (µg/L)[4]	pH[5]	Temperature (degrees Celsius)[5]	Conductivity (µs)[6]
Rocky Knolls Golf Course, Custer															
well R20-2002-14	SE¼NW¼NE¼SE¼ sec. 27, T. 3 S., R. 4 E.	10-01-02	12:05 PM	0.4	0.36	<0.02	0.578	0.038	31	<1	<10	<0.700	7.00	11.6	585
		01-28-03	12:15 PM	<0.1	0.63	0.08	1.81	0.017	39	<1	<2	---	6.98	6.0	542
		05-08-03	11:30 AM	0.6	0.66	0.05	2.3	0.043	24	<1	<10	---	6.77	5.6	546
		07-29-03	2:00 PM	<0.1	0.56	0.04	0.491	0.026	25	5.2	4	---	6.87	17.2	582
well R20-2002-12	SE¼NW¼NE¼SE¼ sec. 27, T. 3 S., R. 4 E.	10-01-02	10:45 AM	<0.1	0.53	<0.02	0.167	0.042	23	<1	<10	<0.700	7.39	11.6	686
		01-28-03	11:45 AM	<0.1	0.24	<0.02	0.083	0.042	34	<1	<2	---	6.96	6.7	602
		05-08-03	11:15 AM	0.2	0.84	0.07	0.446	0.096	24	<1	<10	---	7.06	6.0	749
		07-29-03	1:30 PM	<0.1	0.6	<0.02	0.369	0.119	15	<1	<2	---	7.20	15.9	611
well R20-2002-15	SE¼NW¼NE¼SE¼ sec. 27, T. 3 S., R. 4 E.	10-01-02	2:50 PM	2.1	0.75	0.08	0.844	0.026	179	<1	<10	<0.700	6.72	9.8	1043
		01-28-03	1:15 PM	4.2	0.74	0.08	1.27	0.019	158	<1	<2	---	6.91	6.7	1070
		05-08-03	12:15 PM	4.0	0.59	<0.02	1.35	0.026	129	<1	<2	---	6.79	5.9	987
		07-29-03	3:00 PM	4.2	0.55	0.03	0.584	0.027	187	<1	<2	---	6.65	11.8	1007
Sources of recharge to the Madison aquifer															
Bear Butte Creek at Galena	SE¼NE¼NW¼SE¼ sec. 4, T. 4 N., R. 4 E.	01-14-03	1:15 PM	0.3	<0.32	<0.02	0.008	0.007	21	1.0	<2	<0.700	6.95	2.2	644
		04-15-03	11:30 AM	<0.1	<0.11	<0.02	0.039	0.011	9	<1	<2	---	8.02	6.9	259
		07-22-03	1:45 PM	0.1	0.12	<0.02	0.022	0.015	16	687.0	470	---	8.16	24.8	372
		10-22-03	11:15 AM	4.8	<0.11	<0.02	0.004	0.004	17	20.1	10	---	8.01	7.6	1237
		10-22-03[7]	11:15 AM	4.8	<0.11	<0.02	0.005	0.004	16	25.3	10	---	8.01	7.6	1237
Boxelder Creek at Norris Peak Road	SE¼NE¼NW¼SE¼ sec. 17, T. 2 N., R. 6 E.	01-16-03	2:00 PM	0.2	<0.32	<0.02	0.003	0.005	8	2.0	2	<0.700	7.74	2.2	405
		04-16-03	12:30 PM	<0.1	<0.11	<0.02	0.014	0.012	5	<1	<10	---	8.16	9.5	311
		07-23-03	1:00 PM	<0.1	<0.11	<0.02	0.012	0.01	11	6.3	10	---	8.64	22.5	323
		10-16-03	12:30 PM	<0.1	<0.11	<0.02	0.008	0.032	8	2.0	<10	---	8.09	8.3	373
Rapid Creek at Dark Canyon	SW¼SW¼SW¼NW¼ sec. 13, T. 1 N., R. 6 E.	01-21-03	1:40 PM	0.1	<0.32	0.03	<0.002	0.008	3	<1	<2	<0.700	7.65	0.1	331
		04-22-03	11:00 AM	0.1	<0.11	<0.02	0.008	0.008	5	<1	<2	---	8.35	11.0	372
		07-24-03[9]	10:45 AM	<0.1	<0.11	<0.02	0.007	0.014	5	1.0	<10	---	8.50	14.9	368
		07-30-03[7]	2:00 PM	---	---	---	---	---	---	3.1	<10	---	8.60	15.1	364
		10-23-03	2:45 PM	<0.1	<0.11	<0.02	0.004	0.007	4	<1	<10	---	8.35	11.3	376
		10-23-03[8]	2:45 PM	<0.1	<0.11	<0.02	0.002	0.005	<3	<1	<10	---	5.91	20.5	2.3
Spring Creek at Stratobowl	NE¼SW¼SW¼NE¼ sec. 12, T. 1 S., R. 6 E.	01-21-03	11:30 AM	0.2	<0.32	<0.02	0.007	0.006	16	1.0	2	<0.700	7.03	1.0	366
		04-24-03	2:15 PM	<0.1	0.12	<0.02	0.019	0.019	17	1.0	<10	---	7.67	9.7	310
		08-26-03	11:15 AM	<0.1	<0.11	<0.02	0.023	0.018	18	6.3	10	---	8.48	21.1	321
		10-30-03	12:00 PM	<0.1	0.12	<0.02	0.01	0.009	18	2.0	10	---	8.09	4.8	330

Table 6 – continued

Site name and sample source	Location	Date collected	Time collected	Nitrate plus nitrite as N (mg/L)[1]	Total Kjeldahl nitrogen (mg/L)	Ammo-nia (mg/L)	Total phos-phorous (mg/L)	Dissolved phos-phorous (mg/L)	Chloride (mg/L)	*E. coli* (/100 ml)[2]	Fecal coliform (/100 ml)[3]	Caffeine (µg/L)[4]	pH[5]	Temp-erature (degrees Celsius)[5]	Conduc-tivity (µs)[6]
Water supplies derived from the Madison aquifer															
Boulder Park public water supply	NW¼NE¼NW¼NW¼ sec. 14, T. 5 N., R. 4 E.	01-16-03	10:15 AM	0.4	<0.32	<0.02	0.008	0.016	5	<1	<2	<0.700	7.21	9.3	398
		04-15-03	9:45 AM	0.4	<0.11	<0.02	0.026	0.02	8	1.0	<2	---	7.42	9.9	350
		07-23-03	11:00 AM	0.4	<0.11	<0.02	0.018	0.023	8	3.1	<10	---	7.65	11.7	377
		10-22-03	10:00 AM	0.3	<0.11	<0.02	0.016	0.019	6	1.0	<10	---	7.54	11.0	380
Cavalry Trails public water supply	NE¼NE¼SE¼SE¼ sec. 15, T. 2 N., R. 6 E.	01-22-03	10:30 AM	0.3	<0.32	<0.02	0.012	0.019	< 3	<1	<2	<0.700	7.36	10.1	346
		04-21-03	10:15 AM	0.4	<0.11	<0.02	0.014	0.015	3	<1	<10	---	7.55	11.0	336
		07-24-03[9]	1:30 PM	0.3	<0.11	<0.02	0.015	0.02	3	<1	<10	---	7.71	12.6	324
		08-28-03[7]	1:30 PM	---	---	---	---	---	---	<1	<2	---	7.76	13.1	335
		10-16-03	10:15 AM	0.3	<0.11	<0.02	0.015	0.042	3	<1	<10	---	7.40	9.8	334
Chapel Lane public water supply	SW¼SE¼SE¼SE¼ sec. 8, T. 1 N., R. 7 E.	01-13-03	2:40 PM	0.4	<0.32	<0.02	0.012	0.016	7	<1	<2	<0.700	7.44	13.9	348
		04-23-03	11:00 AM	0.4	<0.11	<0.02	0.02	0.021	8	<1	<2	---	7.83	13.6	347
		07-30-03	1:15 PM	0.4	<0.11	<0.02	0.012	0.036	7	<1	<10	---	7.78	15.5	347
		11-04-03[10]	2:30 PM	0.4	<0.11	<0.02	0.014	0.017	8	<1	<1	---	7.62	12.9	354
City Spring at Wilderness Park	SE¼SE¼SE¼NE¼ sec. 32, T. 2 N., R. 7 E.	01-27-03	11:25 AM	0.4	<0.10	<0.02	0.014	0.018	6	<1	<2	<0.700	7.37	10.7	385
		04-21-03	11:15 AM	0.5	<0.11	<0.02	0.011	0.012	10	<1	<10	---	7.38	11.3	437
		08-14-03	1:00 PM	<0.1	0.28	<0.02	0.055	0.036	10	20.1	110	---	7.86	21.9	426
Copper Oaks public water supply	NW¼NE¼SW¼SW¼ sec. 5, T. 1 S., R. 7 E.	01-21-03	10:10 AM	0.4	<0.32	<0.02	0.044	0.008	11	<1	<2	<0.700	7.05	8.9	348
		04-28-03	9:15 AM	0.1	<0.11	<0.02	0.096	0.021	18	6.3	<10	---	7.78	10.5	330
		08-26-03	1:30 PM	0.5	<0.11	<0.02	0.018	0.016	16	<1	<10	---	7.55	13.8	360
		10-30-03	1:30 PM	0.4	<0.11	<0.02	0.027	0.017	13	<1	<10	---	7.42	9.8	350
Highland Hills public water supply	SW¼NE¼NE¼SE¼ sec. 32, T. 1 N., R. 7 E.	01-23-03	2:30 PM	0.4	<0.32	<0.02	0.017	0.024	8	<1	<2	<0.700	6.85	10.0	370
		04-30-03	10:30 AM	0.3	<0.11	<0.02	0.027	0.039	10	<1	<10	---	7.51	11.9	351
		08-27-03	1:45 PM	0.4	<0.11	<0.02	0.016	0.019	10	<1	<10	---	7.55	14.9	347
		11-03-03	2:00 PM	0.4	<0.11	<0.02	0.02	0.012	18	<1	<10	---	7.41	11.3	387
Jackson Spring public water supply	SE¼SE¼NE¼SW¼ sec. 8, T. 1 N., R. 7 E.	01-22-03	1:15 PM	0.3	<0.32	<0.02	0.009	0.015	5	<1	<2	<0.700	7.44	10.6	376
		04-22-03	2:00 PM	0.4	<0.11	<0.02	0.011	0.014	5	<1	<2	---	7.50	13.8	366
		07-30-03	11:00 AM	0.4	<0.11	<0.02	0.011	0.012	6	<1	<10	---	7.62	13.5	365
		10-27-03	11:30 AM	0.4	<0.11	<0.02	0.011	0.01	5	<1	<10	---	7.50	12.6	367

Table 6 – continued

Site name and sample source	Location	Date collected	Time collected	Nitrate plus nitrite as N (mg/L)[1]	Total Kjeldahl nitrogen (mg/L)	Ammonia (mg/L)	Total phosphorous (mg/L)	Dissolved phosphorous (mg/L)	Chloride (mg/L)	*E. coli* (/100 ml)[2]	Fecal coliform (/100 ml)[3]	Caffeine (µg/L)[4]	pH[5]	Temperature (degrees Celsius)[5]	Conductivity (µs)[6]
Water supplies derived from the Madison aquifer – continued															
Rapid City #6 public water supply	NE¼SE¼SE¼NE¼ sec. 32, T. 2 N., R. 7 E.	01-27-03	10:30 AM	<0.1	<0.10	<0.02	0.014	0.012	5	<1	<2	<0.700	8.28	16.2	254
		01-27-03[7]	10:45 AM	<0.1	<0.10	<0.02	0.013	0.018	3	<1	<2	<0.700	8.26	16.2	251
		04-21-03	1:15 PM	<0.1	<0.11	<0.02	0.009	0.007	<3	<1	<10	---	8.10	10.5	314
		04-21-03[7]	1:30 PM	<0.1	<0.11	<0.02	0.008	0.01	<3	<1	<10	---	8.10	10.5	314
		08-14-03	1:30 PM	0.2	<0.11	<0.02	0.007	0.032	3	<1	<10	---	7.82	15.1	338
		08-14-03[8]	2:00 PM	<0.1	<0.11	<0.02	<0.002	0.003	<3	<1	<10	---	6.57	29.8	1.3
		10-21-03	1:15 PM	0.2	<0.11	<0.02	0.017	0.012	6	<1	<10	---	7.72	18.8	350
Rapid City #9 public water supply	NW¼SE¼SW¼NW¼ sec. 10, T. 1 N., R. 7 E.	01-23-03	10:00 AM	0.4	<0.32	<0.02	0.008	0.009	6	<1	<2	<0.700	7.55	13.3	359
		04-23-03	1:45 PM	0.3	<0.11	<0.02	0.014	0.011	7	<1	<2	---	7.71	15.6	365
		08-14-03	11:00 AM	0.4	<0.11	<0.02	0.01	0.015	6	<1	<10	---	7.97	17.8	363
		10-27-03	1:45 PM	0.4	<0.11	<0.02	0.012	0.011	5	<1	<10	---	7.58	16.0	366
Sturgis public water supply	NE¼NE¼SW¼NW¼ sec. 9, T. 5 N., R. 5 E.	01-14-03	11:05 AM	1.2	<0.32	<0.02	0.016	0.013	5	<1	<2	<0.700	7.26	12.0	595
		04-16-03	9:30 AM	1.2	<0.11	<0.02	0.015	0.015	6	<1	<10	---	7.32	12.4	601
		07-22-03	12:00 PM	1.1	0.12	<0.02	0.015	0.014	4	<1	<10	---	7.40	14.0	574
		10-23-03	10:30 AM	1.2	<0.11	<0.02	0.014	0.015	4	<1	<10	---	7.30	12.7	590
Hill City wastewater treatment lagoons															
well R20-2002-05	NE¼SW¼NE¼SE¼ sec. 30, T. 1 S., R. 5 E.	09-17-02	12:00 PM	<0.1	0.48	<0.02	0.872	0.074	17	<1	2	<0.700	8.19	15.0	373
		02-05-03	1:15 PM	<0.1	0.37	0.02	0.255	0.029	19	<1	<2	---	7.73	5.0	450
		05-01-03	11:00 AM	<0.1	0.64	0.06	0.096	0.039	22	<1	<10	---	8.07	5.1	487
		08-20-03	11:10 AM	<0.1	0.37	0.09	0.107	0.038	20	<1	<10	---	7.85	15.7	498
well R20-2002-04	SW¼NW¼NW¼SW¼ sec. 29, T. 1 S., R. 5 E.	09-19-02	10:40 AM	<0.1	3.98	2.82	0.382	0.034	33	1.0	<10	<0.700	7.81	12.2	691
		02-05-03	11:30 AM	<0.1	2.87	2.79	0.711	0.03	30	<1	<2	---	7.73	4.8	686
		05-01-03	12:00 PM	<0.1	3.57	2.49	0.137	0.03	28	<1	<10	---	7.61	5.4	646
		08-20-03	10:30 AM	<0.1	3.31	3.09	0.165	0.047	29	<1	<10	---	7.60	14.3	653
well R20-2002-03	SE¼NW¼NW¼ SW¼ sec. 29, T. 1 S., R. 5 E.	09-19-02	1:00 PM	0.1	3.58	2.47	0.36	0.05	80	<1	<10	<0.700	7.29	13.9	1091
		02-04-03	12:15 PM	2.0	5.13	3.42	0.184	0.017	86	<1	<2	---	7.07	4.2	1103
		04-29-03	1:30 PM	1.7	4.18	3.29	0.054	0.017	80	<1	<10	---	7.27	5.0	1066
		08-19-03	12:15 PM	0.3	4.51	3.59	0.179	0.03	82	<1	<10	---	7.06	13.8	1156

Table 6 – continued

Site name and sample source	Location	Date collected	Time collected	Nitrate plus nitrite as N (mg/L)[1]	Total Kjeldahl nitrogen (mg/L)	Ammonia (mg/L)	Total phosphorous (mg/L)	Dissolved phosphorous (mg/L)	Chloride (mg/L)	*E. coli* (/100 ml)[2]	Fecal coliform (/100 ml)[3]	Caffeine (µg/L)[4]	pH[5]	Temperature (degrees Celsius)[5]	Conductivity (µs)[6]
Hill City wastewater treatment lagoons – continued															
well R20-2002-17	SW¼SE¼SW¼NW¼ sec. 29, T. 1 S., R. 5 E.	09-19-02	2:10 PM	1.6	5.54	3.78	1.89	0.102	82	<1	<10	<0.700	6.92	11.4	1002
		02-05-03	10:45 AM	2.6	12.6	10	0.562	0.026	84	<1	<2	---	7.12	8.2	998
		04-29-03	11:15 AM	3.3	3.19	2.05	0.148	0.077	83	<1	<10	---	7.03	6.9	1068
		08-19-03	11:30 AM	10.1	6.01	5.32	0.432	0.058	89	<1	<10	---	7.15	12.3	1016
		08-19-03[7]	11:45 AM	9.3	6.32	5.79	1.02	0.038	89	<1	<10	---	7.15	12.3	1016
well R20-2002-02	NE¼SW¼SW¼NW¼ sec. 29, T. 1 S., R. 5 E.	09-24-02	10:15 AM	<0.1	7.16	5.66	1.96	0.114	77	<1	<2	<0.700	7.26	15.0	948
		02-04-03	11:00 AM	<0.1		3.27	0.904	0.02	80	<1	<2	---	7.26	7.8	924
		04-29-03	10:30 AM	<0.1	5.06	3.86	1.58	0.024	78	<1	<10	---	7.28	6.7	863
		08-19-03	11:00 AM	<0.1	6.28	5.33	1.05	0.062	84	<1	<10	---	7.22	15.2	952
Lagoon #1	SE¼NE¼NE¼SE¼ sec. 30, T. 1 S., R. 5 E.	05-20-03	10:45 AM	0.8	34.1	10.7	4.76	3.49	80	>2420	2,300,000	5.51	8.18	10.8	784
Lagoon #2	NW¼NW¼NW¼SW¼ sec. 29, T. 1 S., R. 5 E.	05-20-03	11:30 AM	<0.1	12.8	7.77	2.71	2.08	74	21.1	24	<0.700	8.72	14.0	660
		05-20-03[7]	11:30 AM	<0.1	14.4	7.72	2.71	2.06	75	48.1	32	---	8.72	14.0	660
Lagoon #3	SE¼SW¼SW¼NW¼ sec. 29, T. 1 S., R. 5 E.	05-20-03	12:15 PM	<0.1	12.7	5.52	2.57	1.93	71	18.7	340	<0.700	8.67	14.7	623

[1] mg/L = milligrams per liter

[2] /100 ml = most probable number per 100 milliliters

[3] /100 ml = colonies per 100 milliliters

[4] µg/L = micrograms per liter

[5] Determined in the field using unfiltered samples

[6] µs = microsiemens; determined in the field using unfiltered samples

[7] Duplicate or confirmation sample

[8] Blank sample for purposes of quality assurance and quality control

[9] Questionable laboratory results for *E. coli* and fecal coliform

[10] Questionable laboratory results for fecal coliform

are the result of erosion and deposition associated with Rapid Creek, which is a large perennial stream immediately adjacent to the property. The soils underlying the property are classified as the Hilger-Virkula complex which is typically cobbly loam to silty clay loam to a depth of about 60 inches. This soil complex is considered to have severe limitations for on-site wastewater treatment system drain fields because of steep slopes, cobbles and boulders, and slow permeability rates in some areas (Ensz, 1990). Streamflow in Rapid Creek at this location is significantly affected by releases from Pactola Dam which is upstream from the site and results in higher flows during summer months when downstream irrigators require greater quantities of water (app. B).

Topographically, this site occurs on the Rapid Creek flood plain, immediately adjacent to Rapid Creek which drains a large portion of the central Black Hills uplift. Hundreds of on-site wastewater treatment systems exist within the Rapid Creek watershed up-gradient from this study site, including the community of Johnson Siding which is located approximately 0.5 mile to the northwest and consists of over 100 residences served by on-site wastewater treatment systems. Immediately up-gradient from the study location, neighboring on-site wastewater treatment systems are located at distances of approximately 200 feet, 330 feet, and 400 feet within the same alluvial deposit that underlies this study site.

Figure 10 shows the locations of the on-site wastewater treatment system drain field, the installed monitoring wells, the house, Rapid Creek, and the configuration of the local water table on July 30, 2002. It was originally anticipated that monitoring well R20-2002-20 would provide water samples representing background conditions at the site, and that monitoring wells R20-2002-21 and R20-2002-22 would sample ground-water effluent migrating away from the drain field. However, in this locality and under the hydrologic conditions portrayed in figure 10, the site was characterized as a losing stream situation and the water-table gradient slopes toward R20-2002-20. Therefore, monitoring wells R20-2002-21 and R20-2002-22 represent the up-gradient, background conditions and monitoring well R20-2002-20 provided the water-quality samples down-gradient from the drain field. Monitoring wells R20-2002-21 and R20-2002-22 also may have been located in a position to intercept constituents entering the ground water from surface water in Rapid Creek. Ground water occurred in the alluvial sediments at depths ranging from 4.70 feet below ground surface in monitoring well R20-2002-21 to 15.54 feet below ground surface in monitoring well R20-2002-20, and water levels fluctuated up to about 3 feet during the study (table 4). The water-table gradient shown in figure 10, from measurements taken on July 30, 2002, drops approximately 1.5 feet across a lateral distance of about 100 feet.

Results of all water-quality analyses from this site are presented in table 6, and selected results of analyses from the site are summarized in table 7. Data for this study locality show that nitrate-nitrogen concentrations ranged from a minimum of 0.1 mg/L in all three monitoring wells to a maximum of 1.3 mg/L in well R20-2002-20 which is the down-gradient sample location. Furthermore, well R20-2002-20 exhibited the three greatest nitrate-nitrogen concentrations at the site, although all concentrations at the site were well below the U.S. Environmental Protection Agency maximum contaminant level of 10 mg/L. It also is interesting to note that monitoring well R20-2002-20 is a lateral distance of approximately 70 feet from the edge of the on-site wastewater treatment system drain field.

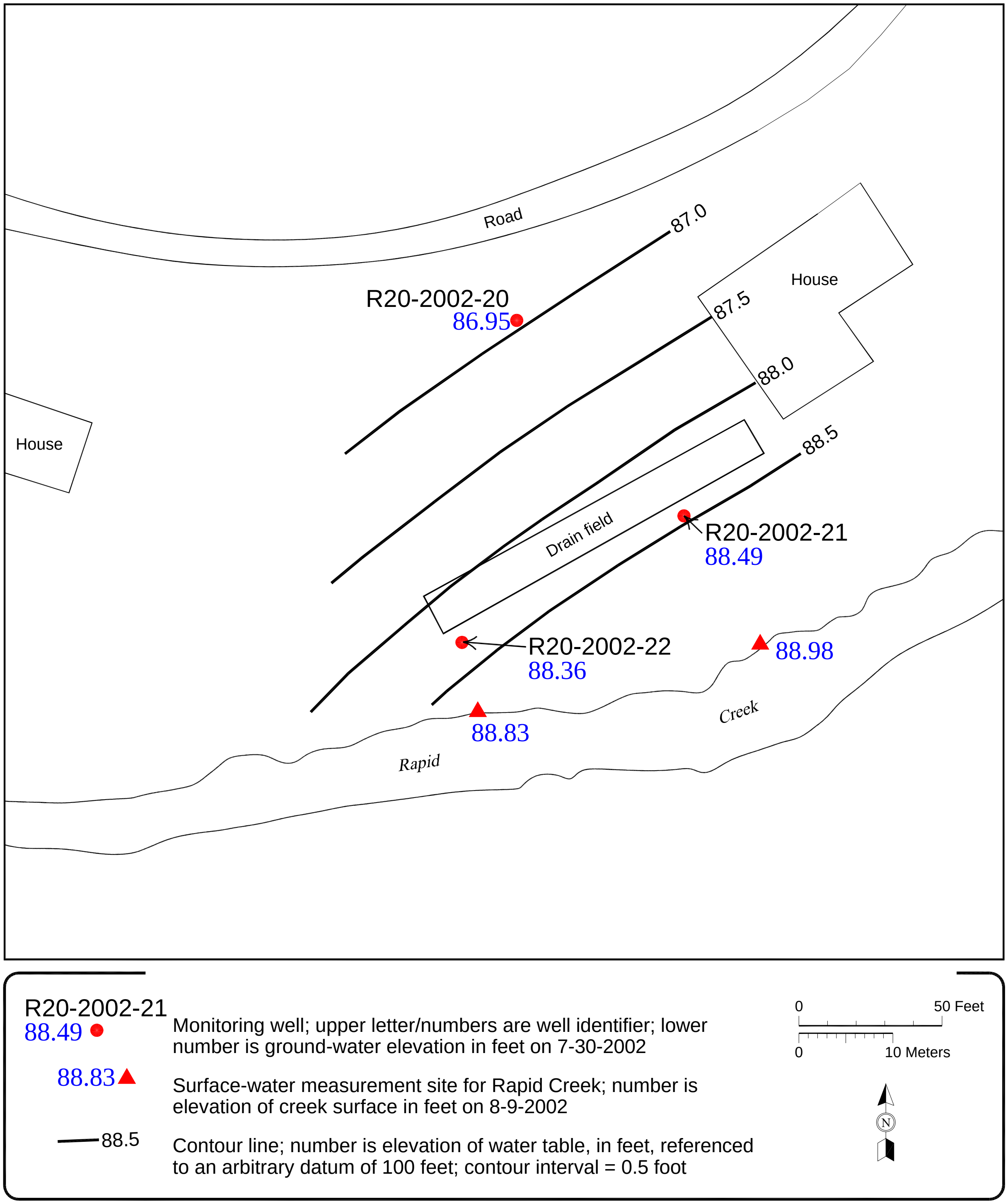


Figure 10. Map showing ground-water elevations and locations of monitoring wells near an on-site wastewater treatment system drain field in an unconsolidated alluvial deposit near Johnson Siding.

Table 7. Selected results of analyses for the residential on-site wastewater treatment system at Johnson Siding

Monitoring well	Constituent	Concentration
R20-2002-22	Total phosphorus	4.05 milligrams per liter
	E. coli	1.0/100 milliliters[1]
R20-2002-20	Nitrate-nitrogen	1.3 milligrams per liter
	Ammonia	0.18 milligrams per liter
	Chloride	7 milligrams per liter
	Conductivity	422 microsiemens

[1] /100 milliliters = most probable number per 100 milliliters

Concentrations of ammonia also indicate an increase across the site ranging from less than 0.02 mg/L in all monitoring wells to 0.18 mg/L in two samples from monitoring well R20-2002-20. The greatest chloride concentration (7 mg/L) and the greatest conductivity reading (422 microsiemens) occurred in a sample collected from monitoring well R20-2002-20. The greatest conductivity concentrations from all three monitoring wells were recorded in May 2003 when precipitation was relatively high and streamflow was low (apps. A, B).

Total phosphorus concentrations at the site varied widely from less than 0.002 mg/L (the laboratory detection limit) to a maximum of 4.05 mg/L, and both of these concentrations came from monitoring well R20-2002-22 which is up-gradient of the drain field. Total phosphorus also appears to be slightly elevated in the January and August sampling events when ground-water levels were relatively high (table 4). Dissolved phosphorus ranged from less than 0.002 mg/L (the laboratory detection limit) in all three monitoring wells at the site to 0.017 mg/L in monitoring well R20-2002-21. The greatest total Kjeldahl nitrogen concentration at the site was 0.92 mg/L in monitoring well R20-2002-22 on August 13, 2003. Samples from all three monitoring wells at the site were less than the laboratory detection limit of 0.11 mg/L for total Kjeldahl nitrogen on May 13, 2003, when streamflow was low and precipitation was high (apps. A, B).

E. coli was detected in one only sample from the entire site (well R20-2002-22). When considered together with the water-table gradient, the total phosphorus, and the total Kjeldahl nitrogen, the *E. coli* results from monitoring well R20-2002-22 may indicate an influence from surface water in Rapid Creek rather than from the on-site wastewater treatment system drain field which is located down-gradient from this well. Caffeine was not detected at this site, and no detections of fecal coliform bacteria were observed in samples analyzed for that parameter (table 6).

Rocky Knolls Golf Course, Custer, South Dakota

The club house at the Rocky Knolls Golf Course in Custer, South Dakota, has an on-site wastewater treatment system that has been in operation for approximately 20 years. This on-site wastewater treatment system is used on a seasonal basis, usually from the months of May through October, and it had been pumped and inspected approximately every 3 years and had no known functional problems. The general hydrogeologic setting of the site consists of unconsolidated alluvial sediments overlying Precambrian lithologies of thick-bedded, tan, quartzose schist with sparse garnet and sillimanite (Redden and others, 2001). The alluvial material was derived from erosion and deposition associated with French Creek, and it is composed of reworked Precambrian clasts ranging from clay to boulders in size. The soils underlying the property are classified as Buska-Virkula loams which are typically loam to silty clay loam to a depth of about 40 to 60 inches below ground surface. These soils are considered to have moderate to severe limitations for on-site wastewater treatment system drain fields because of steep slopes and moderately slow permeability (Ensz, 1990).

Topographically, this site occurs on the French Creek flood plain at the edge of a small lake created by a dam on French Creek. The French Creek watershed drains a significant portion of the south-central Black Hills uplift, and hundreds of on-site wastewater treatment systems exist within this watershed up-gradient from this study site. The nearest neighboring on-site wastewater treatment system up-gradient from this study site is approximately 1,000 feet to the west, and there are about 70 on-site wastewater treatment systems within approximately 2,000 feet of the site, although many of these on-site wastewater treatment systems are installed in soils overlying fractured crystalline bedrock rather than unconsolidated alluvial sediments.

Figure 11 shows the locations of the on-site wastewater treatment system drain field, the club house and parking area, the lake, the installed monitoring-well network, and the configuration of the water table at the site. Monitoring wells had to be installed as quickly and discreetly as possible to minimize disruption to the golf course business; therefore, it was necessary to select monitoring well locations before site-specific water-level information was available. Monitoring well R20-2002-15 was installed approximately 150 feet southeast of the drain field to establish background conditions, and monitoring wells R20-2002-12 and R20-2002-14 were installed between the drain field and the lake in an effort to intercept effluent that could be migrating toward the lake. Three separate attempts were made to install an additional monitoring well northeast of the drain field in the event that the water-table gradient was to the northeast; however, due to repeated auger refusal and equipment breakage, it was not possible to install a monitoring well in that location.

Water-level elevations ranged from 0.72 foot below ground surface in monitoring well R20-2002-12 to 12.35 feet below ground surface in monitoring well R20-2002-15 (table 4), and water levels fluctuated as much as 4 feet during the study. The water-table gradient shown in figure 11, from measurements taken on August 23, 2002, drops approximately 1.5 feet across a

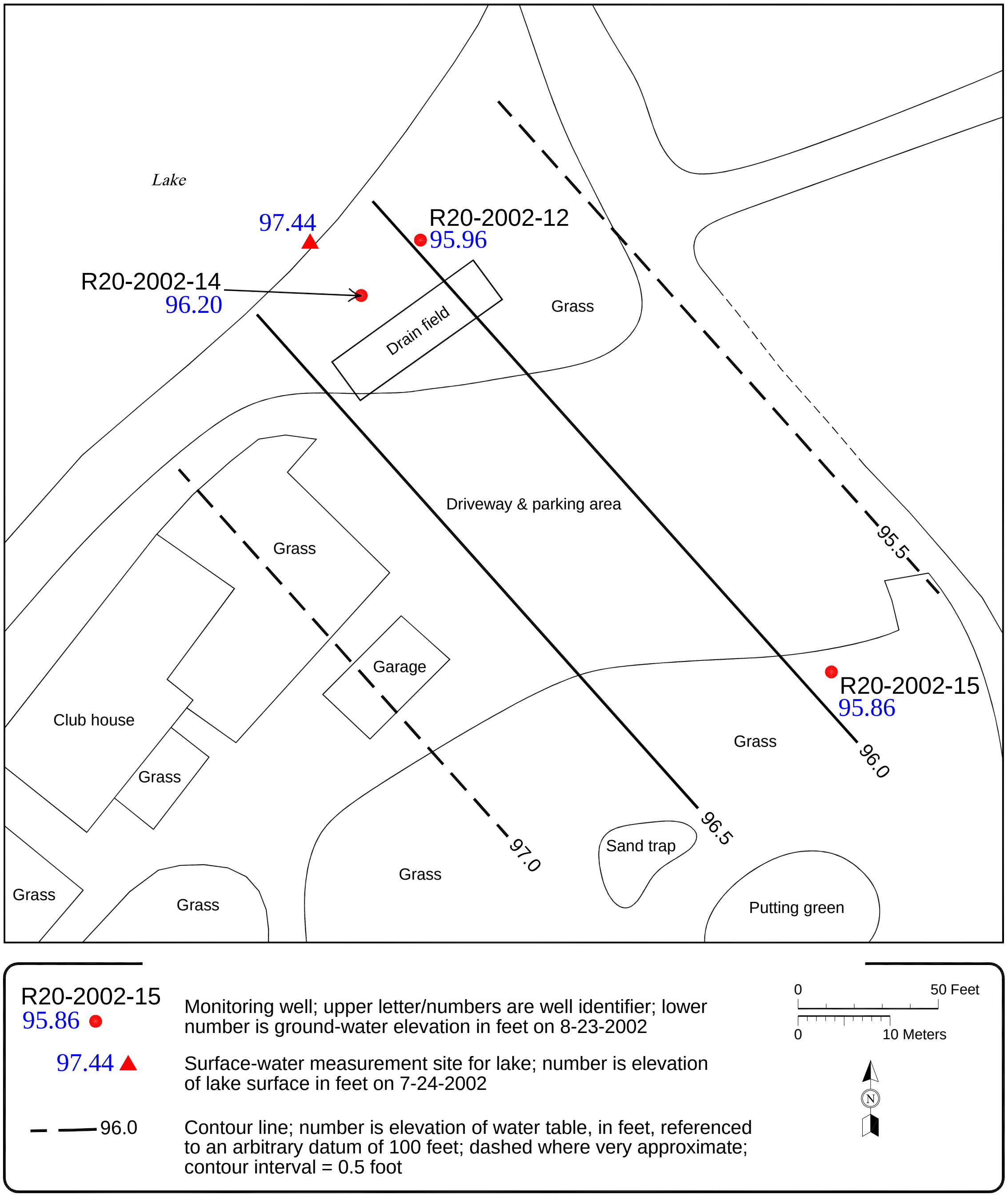


Figure 11. Map showing ground-water elevations and locations of monitoring wells near an on-site wastewater treatment system drain field in an unconsolidated alluvial deposit near Custer.

lateral distance of about 150 feet. Ground-water elevation contours shown in figure 11 indicate that the water-table gradient at this site slopes toward the northeast where monitoring-well installation was unsuccessful due to subsurface conditions. Therefore, there is no monitoring well in the down-gradient direction at this site.

Results of all water-quality analyses from this site are presented in table 6, and selected results of analyses from the site are summarized in table 8. Data for this study locality show that nitrate-nitrogen concentrations ranged from less than 0.1 mg/L (non-detect) in monitoring wells R20-2002-12 and R20-2002-14 to a maximum of 4.2 mg/L in monitoring well R20-2002-15. Nitrate-nitrogen concentrations from monitoring wells R20-2002-12 and R20-2002-14 were less than 1.0 mg/L for all four sample events, but nitrate-nitrogen concentrations from monitoring well R20-2002-15 ranged from 2.1 to 4.2 mg/L in the same four sample events. Furthermore, water samples from monitoring well R20-2002-15 contained 5 to 10 times as much chloride and nearly twice the conductivity values of samples from monitoring wells R20-2002-12 and R20-2002-14. An exception to this trend was the detection of both *E. coli* and fecal coliform bacteria from monitoring well R20-2002-14. These detections of bacteria occurred in July 2003 when precipitation was low and streamflow was declining (apps. A, B). Because of the proximity of monitoring well R20-2002-14 to French Creek (fig. 11), it may be possible that surface water in the creek was the source of the bacteria. It is also possible that elevated nitrate-nitrogen and chloride concentrations and elevated conductivity values from monitoring well R20-2002-15 were caused by recycled municipal wastewater which is used to irrigate the golf course and/or fertilizer that is applied to the golf course.

Table 8. Selected results of analyses for the on-site wastewater treatment system at the Rocky Knolls Golf Course

Monitoring well	Constituent	Concentration
R20-2002-15	Nitrate-nitrogen	4.2 milligrams per liter
	Chloride	187 milligrams per liter
	Conductivity	1,070 microsiemens
R20-2002-14	Fecal coliform	4.0/100 milliliters[1]
	E. coli	5.2/100 milliliters[2]
	Total phosphorus	2.3 milligrams per liter
R20-2002-12	Dissolved phosphorus	0.119 milligrams per liter

[1] /100 milliliters = colonies per 100 milliliters
[2] /100 milliliters = most probable number per 100 milliliters

Similar to the previously described sites, total phosphorus concentrations were variable and ranged from a minimum of 0.083 mg/L to a maximum of 2.3 mg/L. The greatest concentrations of total phosphorus from all three monitoring wells occurred in May 2003 when both ground-water levels and streamflow in the adjacent French Creek were relatively high, possibly acting as flushing mechanisms, mobilizing phosphorus in the shallow subsurface. Total Kjeldahl nitrogen concentrations ranged from 0.24 to 0.84 mg/L, ammonia concentrations ranged from less than 0.02 (non-detect) to 0.08 mg/L, and dissolved phosphorus concentrations were between 0.017 and 0.119 mg/L. Caffeine was not detected in any of the three monitoring wells at the site (table 6).

Karst Limestone

The Pahasapa Limestone, known as the Madison aquifer in the subsurface, is the largest source of ground water in the Black Hills area. Most of the cities, towns, and communities around the flanks of the Black Hills obtain water for drinking and other purposes either from wells into the Madison aquifer or from artesian springs originating from this aquifer. In the Black Hills the Pahasapa (Madison) Limestone consists of white, gray, and tan, fine- to medium-grained limestone and dolomite containing brown to gray chert. It ranges in thickness from 300 to 630 feet, generally thinning from north to south across the Black Hills (Martin and others, 2004). The Madison aquifer is karstic in nature, and it is characterized by well-developed solution features including caverns, collapse breccia, swallow holes, and other features derived from dissolution of the carbonate rock which composes the aquifer. Although significant dolomite occurs within the Pahasapa (Madison) Limestone, for the purposes of simplicity and clarity for the reader, this hydrogeologic setting is herein referred to as karst limestone.

Thousands of on-site wastewater treatment systems exist both up-gradient from and within the recharge area for the karstic Madison aquifer in the vicinity of this investigation. Therefore, the approach taken in this study with respect to the karst limestone hydrogeologic setting was to sample some of the recharge sources to the Madison aquifer and partially document the quality of water entering the aquifer. Then, these data were compared with water-quality analyses of water that has traveled through the karst limestone environment either to a deep well serving a public-water supply system on the flanks of the Black Hills or to water that was discharged as artesian spring water on the flanks of the Black Hills.

Figure 12 shows the distribution of the karst limestone recharge area (outcrop of Madison Group), the up-gradient sites of recharge waters that were sampled (surface-water sample site), and the down-gradient locations of sampled public-water supplies and artesian springs (Madison aquifer sample site). Measures were taken at each public-water supply system to obtain raw water samples, unaffected by treatment of any type. Table 6 provides a complete list of the names and locations of these sampled surface-water locations, public water-supply systems, and artesian springs, as well as all of the water-quality information that resulted from sample collection and analyses from these sources of water to and from the Madison aquifer. Table 9 summarizes selected results of analyses from these same sites.

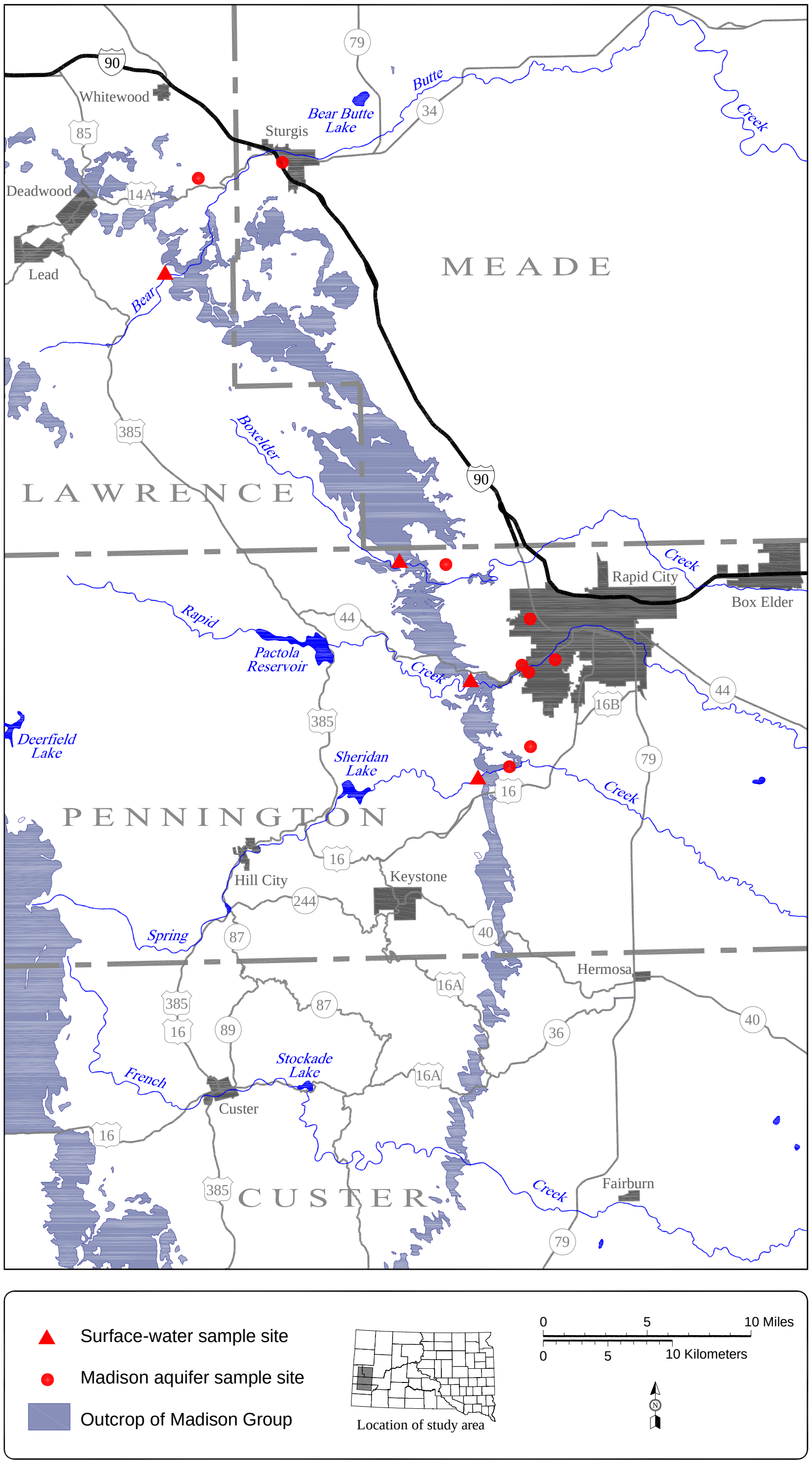


Figure 12. Map showing locations of water-sample collection points for evaluation of the karst limestone hydrogeologic setting.

Table 9. Selected results of analyses for sources of recharge to the Madison aquifer and public water-supply wells and artesian springs discharging from the Madison aquifer

Sources of recharge		
Creek	**Constituent**	**Concentration**
Bear Butte Creek	Nitrate-nitrogen	4.8 milligrams per liter
	E. coli	687.0/100 milliliters[1]
	Fecal coliform	470/100 milliliters[2]
	Conductivity	1,237 microsiemens
Boxelder Creek	*E. coli*	6.3/100 milliliters[1]
	Fecal coliform	10/100 milliliters[2]
Spring Creek	*E. coli*	6.3/100 milliliters[1]
	Fecal coliform	10/100 milliliters[2]
Rapid Creek	*E. coli*	3.1/100 milliliters[1]
Discharge points		
Water supply or spring	**Constituent**	**Concentration**
Sturgis well #2	Nitrate-nitrogen	1.2 milligrams per liter
	Conductivity	601 microsiemens
Copper Oaks	*E. coli*	6.3/100 milliliters[1]
Boulder Park	*E. coli*	3.1/100 milliliters[1]
City Spring	*E. coli*	20.1/100 milliliters[1]
	Fecal coliform	110/100 milliliters[2]

[1] /100 milliliters = most probable number per 100 milliliters
[2] /100 milliliters = colonies per 100 milliliters

Bear Butte Creek, Boxelder Creek, Rapid Creek, and Spring Creek are perennial streams that provide continuous recharge to the Madison aquifer (Hortness and Driscoll, 1998), and water samples from each of these streams were analyzed for the same list of parameters as samples from the other sites in this study to evaluate for possible effects from on-site wastewater treatment systems (table 6). Bear Butte Creek in particular had several interesting water-quality parameter results, including nitrate-nitrogen concentrations ranging up to 4.8 mg/L, *E. coli* results of 687.0 MPN per 100 milliliters, fecal coliform bacteria counts of 470 colonies per 100 milliliters, a maximum conductivity value of 1,237 microsiemens, and chloride concentrations ranging from 9 to 21 mg/L. Most of these parameter concentrations occurred in the July and October 2003 sampling events when precipitation and streamflow were low (apps. A, B). Boxelder Creek, Rapid Creek, and Spring Creek had lower quantities of *E. coli*, and fecal coliform bacteria were also detected in Boxelder Creek and Spring Creek. Detection of bacteria in Boxelder Creek, along with the greatest chloride concentration recorded for Boxelder Creek (11 mg/L), occurred in July 2003 under declining streamflow conditions and with low precipitation levels. Water samples from Spring Creek had chloride concentrations ranging only from 16 to 18 mg/L which did not appear to fluctuate with climatic conditions or streamflow. The only detection of bacteria in Rapid Creek occurred in July 2003 when precipitation was low, but streamflow was elevated due to irrigation releases from Pactola Dam. It is not unusual to find fecal coliform bacteria within streams such as these because of wildlife or livestock that contain these types of organisms within their digestive tracts; however, DNA ribotyping of fecal coliform bacteria from Spring Creek surface water has yielded human signatures in some cases (Schwickerath, 2004).

Nitrate-nitrogen concentrations and conductivity values in water samples from Bear Butte Creek are relatively greater than those from the other sampled streams and may indicate the influence of treated water that is discharged into up-gradient tributaries from the former Gilt Edge Mine site. Nitrate-nitrogen is released as a breakdown product of cyanide that was used in ore processing and from blasting agents used in mining. Water-quality data from the South Dakota Department of Environment and Natural Resources (2006a) show that samples of effluent from the water-treatment plant at the mine site had a nitrate-nitrogen concentration of 17.3 mg/L in September 2003 which was 1 month prior to the concentration of 4.8 mg/L that was recorded down-gradient in Bear Butte Creek for this study. Elevated conductivity values also were documented in water discharged from the mine site, and was measured at 3,310 microsiemens in October 2003 which is the same month that a value of 1,237 microsiemens was measured in Bear Butte Creek for this study. Sulfate concentrations also are known to be elevated in discharge waters from the mine site, and sodium hydroxide and calcium hydroxide have been used periodically to raise pH levels in water discharged from the mine site (South Dakota Department of Environment and Natural Resources, 2006a). These constituents all may contribute to elevated conductivity values for Bear Butte Creek.

Sample results for the selected public water-supply systems and artesian springs on the flanks of the Black Hills show that nitrate-nitrogen concentrations ranged from less than 0.1 (non-detect) to a maximum of 1.2 mg/L (table 6); however, 8 of the 10 public water supply systems and artesian springs that were sampled had nitrate-nitrogen concentrations between 0.1 and 0.5 mg/L. An exception to this general trend was shown by Rapid City municipal well #6 which is a Madison aquifer well and which was noticeably lower in nitrate-nitrogen concentrations, exhibiting no nitrate-nitrogen concentration greater than 0.2 mg/L. Another interesting exception to the trend was the Sturgis public water-supply well which also produces water from the Madison aquifer and which had consistently higher nitrate-nitrogen concentrations with three samples having 1.2 mg/L.

E. coli bacteria were detected in the Boulder Park water supply system in three of the four sampling events (table 9). Detections for both fecal coliform bacteria and *E. coli* occurred in water samples from City Spring in August 2003 when the spring had nearly ceased flowing; however, this site is influenced by waterfowl and other wildlife. The same sample from City Spring in which bacteria were detected also was relatively lower in nitrate-nitrogen concentration than other samples from City Spring. Copper Oaks and Highland Hills public water supply systems exhibited slightly elevated chloride concentrations relative to the other sampled sources, which may reflect the influence of recharge water originating from Spring Creek (table 6). *E coli* bacteria and a relatively low nitrate-nitrogen concentration were present in a water sample collected from the Copper Oaks public water-supply system in April 2003 (apps. A, B). The sample was collected under relatively high precipitation and streamflow conditions.

Conductivity values were between 251 and 437 microsiemens for all recharge sources, public water-supply wells, and artesian springs that were sampled with the notable exception of Bear Butte Creek and the Sturgis public water-supply well. Conductivity values in Bear Butte Creek fluctuated from 259 microsiemens in April 2003 to 1,237 microsiemens in October 2003 and the Sturgis public water-supply well varied from 574 microsiemens in July 2003 to 601 microsiemens in April 2003 (table 6). Exact correlation of conductivity values of water samples from the Sturgis well to surface-water samples collected up-gradient along Bear Butte Creek is complicated by ground-water travel times and ground-water mixing in the Madison aquifer. However, the data suggest that elevated conductivity values in Bear Butte Creek may be affecting conductivity values in the Sturgis public water-supply well.

Rapid City well #9, in southwestern Rapid City, exhibited concentrations of nitrate-nitrogen, chloride, and conductivity that consistently exceeded those of Rapid City well #6 which is located in northwestern Rapid City (table 6). The difference in ground-water quality between these two wells may reflect the influence of fast ground-water flowpaths identified by Long and others (2006) on the eastern flank of the Black Hills uplift. These authors used chlorofluorocarbons, tritium, and specific conductance to estimate ground-water age and residence time, and from this information the location and orientation of probable fast flowpaths within the karstic limestone were identified. These tracers indicated that the fast flowpaths originate where streams sink into the recharge area of the Madison aquifer. Nitrogen and phosphorus concentrations also were elevated along the identified fast flowpaths, and it is possible that Rapid City well #9 is influenced to a greater extent by one of the fast flowpaths than Rapid City well #6 which may explain the difference in ground-water quality found in these two wells.

Total Kjeldahl nitrogen was detected twice in Spring Creek at a concentration of 0.12 mg/L and once in Bear Butte Creek at 0.12 mg/L. Total Kjeldahl nitrogen also was detected once at 0.12 mg/L in the Sturgis public water-supply well and once at City Spring at 0.28 mg/L (table 6). Total phosphorus and dissolved phosphorus concentrations were consistently low, with no concentration for either parameter exceeding 0.02 mg/L in any sample. Ammonia was below the detection limit in all sampled sources (table 6).

Municipal Wastewater Treatment Lagoons, Hill City, South Dakota

The wastewater treatment lagoons at Hill City have been in operation since it was constructed in 1972. There are a series of three non-aerated evaporation lagoons that are intended to provide total retention of flow with no direct discharge to Spring Creek. This wastewater treatment system serves about 860 permanent residents which results in a flow of approximately 101,000 gallons per day entering the lagoons (McLaughlin Water Engineers, Ltd., 2000). Much higher flow rates are experienced during summer months as a result of increased population from tourism and recreation.

Original documents and diagrams on file with the South Dakota Department of Environment and Natural Resources do not mention or show that the lagoons were lined. Communication with personnel on the South Dakota Department of Environment and Natural Resources staff at the time of construction indicates that the lagoons were constructed in native sediment, there were no synthetic or clay liners emplaced, and no compaction was performed (Gary Stephenson, Rapid City, S. Dak., oral commun., 2006). Historical documents on file with the South Dakota Department of Environment and Natural Resources also indicate that the lagoons have never overflowed, and no discharge to Spring Creek has been documented (South Dakota Department of Environment and Natural Resources, 2006b). Some sediment accumulation may have occurred in the lagoons since 1972; however, McLaughlin Water Engineers, Ltd. (2000) calculated that 38,000,000 gal/yr were discharged into underlying sediment as recently as the year 2000; therefore, significant sealing of the lagoon floors probably has not occurred. Vegetative growth in the lagoons also may enhance leakage where roots have penetrated the lagoon floors. It should be noted that the city of Hill City is currently building a new wastewater treatment plant and that the use of this lagoon system will be discontinued in the future.

The three lagoons at the site are located on alluvial sediments in the floor of the valley occupied by Spring Creek. Spring Creek drains a large area in the central Black Hills, and hundreds of on-site wastewater treatment systems exist within this watershed up-gradient of this site. Numerous residential on-site wastewater treatment systems exist along Spring Creek in the Hill City area both up-gradient and down-gradient from the study site. There are five on-site wastewater treatment systems within approximately 600 feet of the northern, eastern, and southern perimeters of the lagoon system, the nearest of which is located approximately 260 feet to the east within the same alluvial deposit as the study site. This information is provided to better characterize the site and is not intended to imply effects of these systems on ground-water quality.

The alluvial deposit underlying the site consists of heterogeneous clay- to boulder-size clasts of reworked Precambrian lithologies. The alluvial deposit blankets the valley floor along Spring Creek and was derived from erosion and deposition associated with Spring Creek. Precambrian bedrock underlies the alluvium and consists primarily of thin-bedded to laminated quartz-biotite-garnet schist (Ratte' and Wayland, 1969). Exposures of schist near the lagoon system show well-developed vertical fractures. The soils at the site are classified as Cordeston-Marshbrook loams which are typical on flood plains in the central Black Hills. These soils are composed of an upper friable loam and a lower gravelly or sandy loam extending down to about 60 inches below land surface. These soils are considered to have moderate to severe limitations for on-site wastewater treatment system drain fields because of restricted permeability, seasonal high water tables, and the potential for flooding (Ensz, 1990).

Five monitoring wells were installed to evaluate the potential effects of this series of wastewater lagoons on the local ground-water quality. Figures 13 and 14 collectively show the locations of the wastewater lagoons, the five monitoring wells, Spring Creek, and the water-table elevations measured on July 7, 2005. Monitoring well R20-2002-05 was installed approximately 600 feet to the west of lagoon #1 to serve as a background monitoring point for ground water migrating into the area. Monitoring wells R20-2002-04, R20-2002-03, R20-2002-17, and R20-2002-02 were installed around the perimeter of the lagoon network in an attempt to characterize the effects of the lagoon system on the shallow ground water in the alluvium along Spring Creek. Measured water levels in the vicinity of the wastewater lagoons varied from a high of 2.28 feet below ground surface in monitoring well R20-2002-04 to a low of 10.04 feet below ground surface in monitoring well R20-2002-17, and water levels fluctuated approximately 2.5 feet during the study (table 4). The water-table gradient shown in figure 13 drops approximately 22 feet over a lateral distance of about 2,500 feet, and the water-level contours indicate that Spring Creek changes from a losing stream up-gradient of the lagoon system to a gaining stream in the vicinity of the lagoons.

Results of all water-quality analyses from this site are presented in table 6, and selected results of analyses from the site are summarized in table 10. Sample results from the monitoring-well network show interesting trends with regard to concentrations of several parameters including nitrate-nitrogen, total Kjeldahl nitrogen, ammonia, and dissolved phosphorus (table 6). These parameters are at lower concentrations in samples from monitoring well R20-2002-05, which represents background conditions, and they are at progressively higher concentrations in monitoring wells R20-2002-04 and R20-2002-03 until reaching their highest concentrations in monitoring well R20-2002-17. These same parameters then decrease in concentration in monitoring well R20-2002-02, which is the farthest down-gradient point that was sampled, just before both surface water and ground water flow away from the wastewater treatment lagoons. For example, nitrate-nitrogen was not detected in monitoring wells R20-2002-05 and R20-2002-04; however, nitrate-nitrogen concentrations are 0.3 mg/L in monitoring well R20-2002-03 and reach a concentration of 10.1 mg/L in monitoring well R20-2002-17 (fig. 14). In monitoring well R20-2002-02, nitrate-nitrogen was not detected. Similarly, all concentrations of total Kjeldahl nitrogen were less than 1.0 mg/L in the background monitoring well, progressively increase within monitoring wells R20-2002-04 and R20-2002-03, peak at a high of 12.6 mg/L in monitoring well R20-2002-17, then decrease to between about 5.0 and 7.0 mg/L in monitoring well R20-2002-02. Ammonia concentrations follow a virtually identical trend beginning with 0.09 mg/L in the background well, climbing to a high concentration of 5.79 mg/L in monitoring well R20-2002-17, then decreasing in monitoring well R20-2002-02 (fig. 14). Conductivity values also increase from approximately 450 microsiemens in monitoring well R20-2002-05 to about 650 microsiemens in monitoring well R20-2002-04, reach approximately 1,100 microsiemens in monitoring well R20-2002-03, decrease to about 1,000 microsiemens in monitoring well R20-2002-17, then decline to approximately 900 microsiemens in monitoring well R20-2002-02. Concentrations for chloride show a slightly different trend across the site, with concentrations of approximately 20 mg/L in samples from monitoring well R20-2002-05, about 30 mg/L in monitoring well R20-2002-04, approximately 80 mg/L in monitoring well

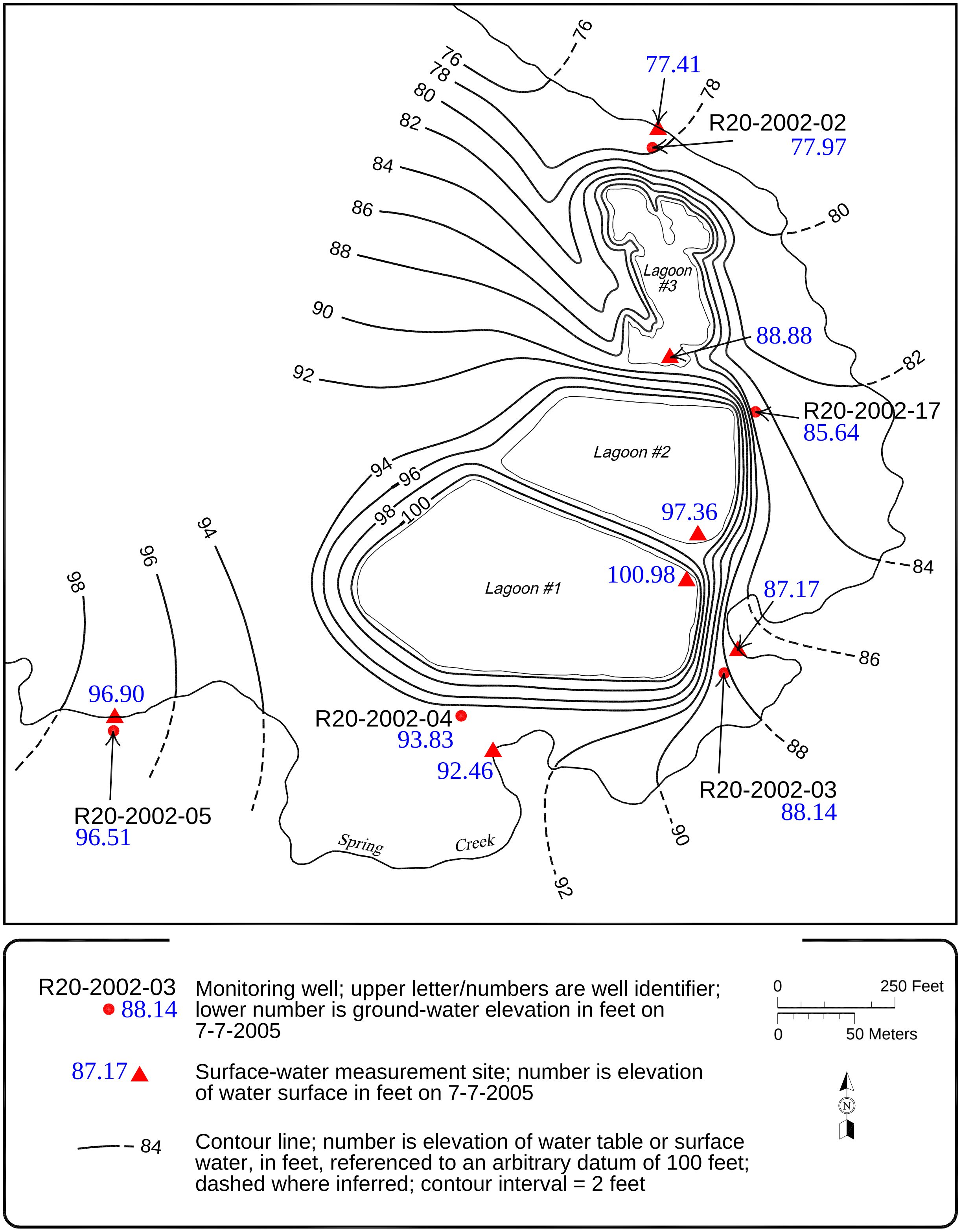
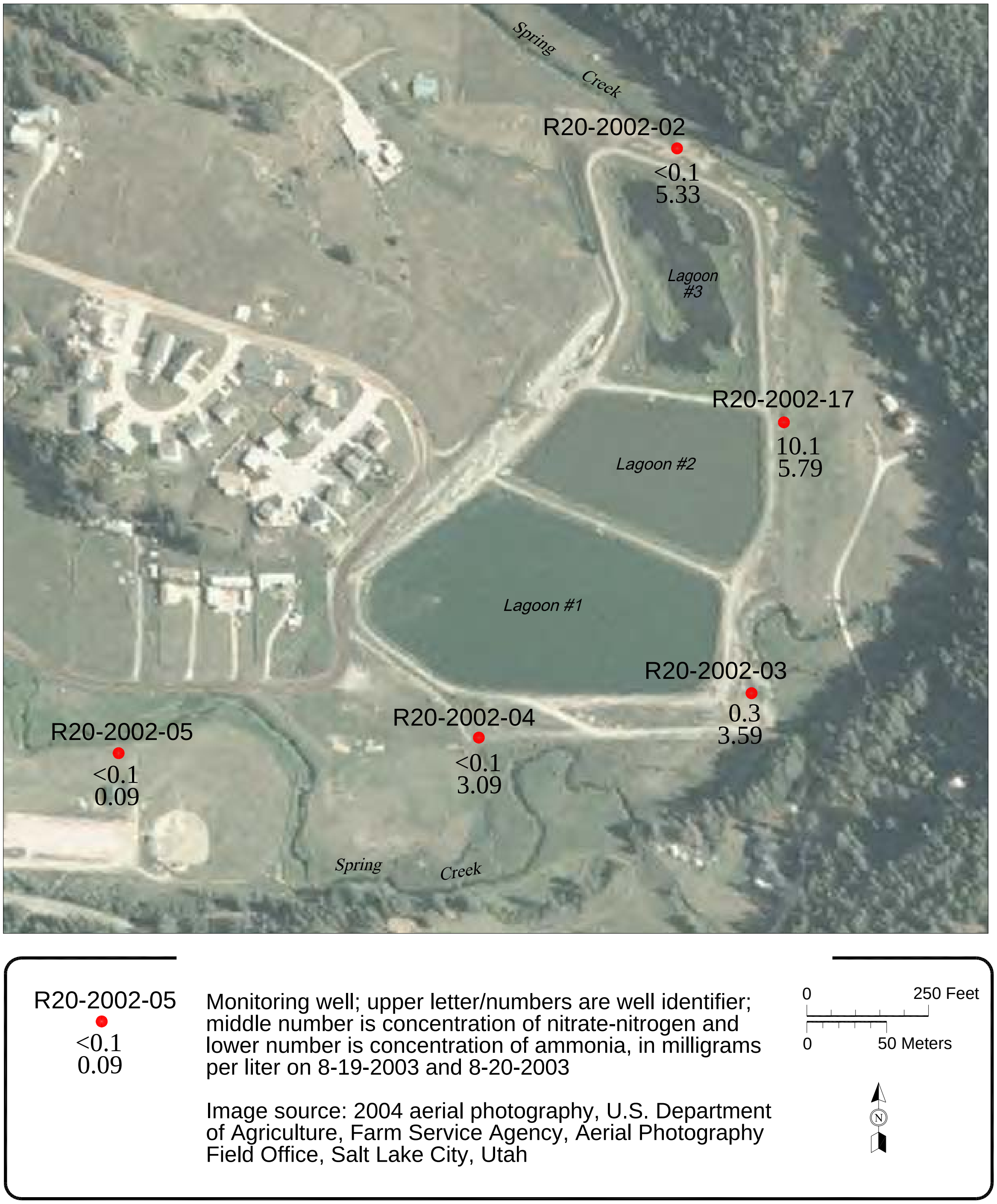


Figure 13. **Map showing ground-water elevations and locations of monitoring wells near the Hill City wastewater treatment lagoons.**



Monitoring well; upper letter/numbers are well identifier; middle number is concentration of nitrate-nitrogen and lower number is concentration of ammonia, in milligrams per liter on 8-19-2003 and 8-20-2003

Image source: 2004 aerial photography, U.S. Department of Agriculture, Farm Service Agency, Aerial Photography Field Office, Salt Lake City, Utah

Figure 14. Map showing concentrations of nitrate-nitrogen and ammonia in monitoring wells near the Hill City wastewater treatment lagoons.

Table 10. Selected results of analyses for the Hill City wastewater treatment lagoons

Monitoring well	Constituent	Concentration
R20-2002-05	All constituents	Background levels or non-detect
R20-2002-04	*E. coli*	1.0/100 milliliters[1]
	Total Kjeldahl nitrogen	3.98 milligrams per liter
	Ammonia	3.09 milligrams per liter
	Chloride	33 milligrams per liter
	Conductivity	691 microsiemens
R20-2002-03	Nitrate-nitrogen	2.0 milligrams per liter
	Total Kjeldahl nitrogen	5.13 milligrams per liter
	Ammonia	3.59 milligrams per liter
	Chloride	86 milligrams per liter
	Conductivity	1,156 microsiemens
R20-2002-17	Nitrate-nitrogen	10.1 milligrams per liter
	Total Kjeldahl nitrogen	12.6 milligrams per liter
	Ammonia	10.0 milligrams per liter
	Total phosphorus	1.89 milligrams per liter
	Chloride	89 milligrams per liter
	Conductivity	1,068 microsiemens
R20-2002-02	Nitrate-nitrogen	<0.1 milligrams per liter
	Total Kjeldahl nitrogen	7.16 milligrams per liter
	Ammonia	5.66 milligrams per liter
	Total phosphorus	1.96 milligrams per liter
	Chloride	84 milligrams per liter
	Conductivity	952 microsiemens

[1] /100 milliliters = most probable number per 100 milliliters

R20-2002-03, and remaining at about 80 mg/L in monitoring wells R20-2002-17 and R20-2002-02 (table 6). These results may illustrate that chloride is generally non-reactive in soil, migrating readily with ground water, and showing no attenuation across the site.

In contrast, total and dissolved phosphorus concentrations are more variable and do not appear to follow a trend across the site. *E. coli* bacteria were detected in only one sample, from monitoring well R20-2002-04, and fecal coliform bacteria also were detected in only one sample, from monitoring well R20-2002-05 which is up-gradient from the lagoon site and may be affected by surface water in Spring Creek (fig. 13). Caffeine was not detected in any ground-water samples from this study location.

Consideration of the results of water-quality analyses with respect to climatic conditions at the site is informative, particularly with respect to monitoring well R20-2002-17, which contained notably greater total Kjeldahl nitrogen and ammonia concentrations in February 2003 when precipitation, ground-water, and surface-water levels all were relatively low (table 4; apps. A, B). This same monitoring well showed significantly greater nitrate-nitrogen concentrations in August 2003 when streamflow was low but precipitation was high. Samples from monitoring well R20-2002-03 were slightly higher in nitrate-nitrogen and total Kjeldahl nitrogen in February 2003 when precipitation and surface-water levels were low, and samples from monitoring well R20-2002-02 were relatively higher in total Kjeldahl nitrogen, ammonia, total phosphorus, and dissolved phosphorus in September 2002 when precipitation was higher but streamflow was low. Overall, samples parameters at the Hill City wastewater treatment lagoons showed greater concentrations during low streamflow conditions and show direct correlation with precipitation.

Water samples obtained directly from the three wastewater lagoons were analyzed for the same parameters as the samples from the monitoring-well network in order to provide information on the original concentration of the potential contaminants (table 6). Nitrate-nitrogen concentrations from the lagoons were lower than in samples from two of the monitoring wells, ranging from 0.8 mg/L in lagoon #1 to less than 0.1 mg/L (non-detect) in lagoons #2 and #3. However, total Kjeldahl nitrogen, dissolved phosphorus, and total phosphorus were significantly higher in lagoon waters than in ground-water samples. *E. coli* and fecal coliform bacteria were significantly more abundant within the wastewater lagoons, and caffeine was detected at 5.51 micrograms per liter in lagoon #1. Chloride concentrations and conductivity values did not vary significantly between samples from the lagoons and those from the monitoring-well network.

Nitrate-nitrogen concentrations are below laboratory detection limits in lagoon #2; however, in monitoring well R20-2002-17 located less than 50 feet down-gradient from lagoon #2 (fig. 14), nitrate-nitrogen concentrations are as high as 10.1 mg/L. This could reflect nitrification processes occurring beneath the lagoons as ammonia in lagoon water is converted to nitrate-nitrogen at infiltrative surfaces and within alluvial sediments underlying the lagoons. Concentrations of total Kjeldahl nitrogen, the sum of organic nitrogen and ammonia, also are greater in lagoon water than in samples from monitoring wells (table 6) which may provide further evidence of conversion of ammonia from lagoon water to nitrate-nitrogen in ground water.

NITRATE-NITROGEN ANALYSIS

Nitrogen Mass-Balance Models

Numerous studies of water-quality effects of on-site wastewater treatment systems have concluded that generation of nitrate-nitrogen is one of the most critical sources of concern (Pruel, 1966; Bouma and others, 1972; Walker and others, 1973; Viraraghavan and Warnock, 1976; Peavy and Groves, 1978; Andreoli and others, 1979; Peavy and Brawner, 1979; Starr and Sawhney, 1980; Cogger and Carlile, 1984; Uebler, 1984; Bauman and Schafer, 1985; Robertson and others, 1989; Tinker, 1991; Hantzsche and Finnemore, 1992; Anderson and others, 1994; Shaw and Turyk, 1994; U.S. Environmental Protection Agency, 2002b; Taylor, 2003). These studies have found that in many cases, particularly involving high-density installations of on-site wastewater treatment systems, nitrate-nitrogen concentrations in ground water exceed the federal drinking water standard of 10 mg/L, causing adverse effects to both ground-water and surface-water resources. Furthermore, regulations and requirements for on-site wastewater treatment systems usually are written for individual sites, and the problem of persistent, cumulative effects from large numbers of on-site wastewater treatment systems in concentrated areas is not addressed in most regulations (Bauman and Schafer, 1985; Hantzsche and Finnemore, 1992).

Nitrate-nitrogen that reaches ground water usually moves with little retardation or denitrification, and reduction of nitrate-nitrogen concentrations in ground water is primarily through dispersion and by dilution from precipitation recharge (U.S. Environmental Protection Agency, 2002b; Taylor, 2003). Therefore, many of the techniques that have been developed to estimate cumulative water-quality effects of effluent from on-site wastewater treatment systems have incorporated nitrogen mass-balance models that attempt to mathematically quantify the mass of background nitrogen entering the boundaries of the study area, nitrogen contributions from within the study area, nitrogen reduction from dilution and other factors, and the mass of nitrogen leaving the study area with migrating ground water. Simplifying assumptions usually are made pertaining to dilution, conversion of ammonium to nitrate-nitrogen, dispersion, denitrification, ground water, precipitation and precipitation recharge, mixing of effluent with ground water, and other factors particular to individual study areas. Most authors also caution that the mass-balance approach is more accurate for predicting nitrate-nitrogen concentrations from large-scale, residential developments with long-term, steady-state conditions, and that the mass-balance approach is not intended to accurately predict nitrate-nitrogen concentrations at a particular point down-gradient from an individual lot or on-site wastewater treatment system (Wehrmann, 1984; Frimpter and others, 1990; Tinker, 1991; Hantzsche and Finnemore, 1992; Taylor, 2003).

Principle variations of the different nitrogen mass-balance approaches consist of the method of calculation of nitrogen concentrations and the volume of ground water entering system boundaries. The most simplified techniques consider only nitrate-nitrogen contributions from on-site wastewater treatment systems in the study area, and that nitrate-nitrogen reduction occurs only as a result of recharge associated with effluent and infiltrating precipitation. More comprehensive methods consider additional sources of nitrate-nitrogen such as fertilizer, storm-water runoff, agricultural practices, animal waste, soil, bedrock, precipitation, and up-gradient ground water. Some models consider the effects of lateral ground-water flow from up-gradient areas, or they may include nitrate-nitrogen reductions from well withdrawals, plant uptake, mineralization, or denitrification. Various models have been designed for different applications, incorporating a variety of combinations of nitrogen input and reduction mechanisms, and many investigations use more than one model to compare results and assist in sensitivity analyses (Bauman and Schafer, 1985; Tinker, 1991; Hantzsche and Finnemore, 1992; Long, 1995; Santos and Associates, 1995; Taylor, 2003).

Application of Nitrogen Models

Nitrogen mass-balance models in the published literature usually are accompanied by qualifications that limit the validity of the model to long-term, area-wide analyses of numerous on-site wastewater treatment systems, and they usually are intended as screening tools to identify areas of potential concern that may warrant more detailed investigation prior to intense development (Bauman and Schafer, 1985; Tinker, 1991; Hantzsche and Finnemore, 1992; Taylor, 2003). However, some of the on-site wastewater treatment systems studied as part of this investigation are located in hydrogeologic settings that are well suited for approximate quantification of ground-water flow, precipitation recharge, and nitrogen loading rates, and it is informative to apply nitrogen mass-balance models to a selected site to compare observed nitrate-nitrogen concentrations with those predicted by the models.

The residential on-site wastewater treatment system located in Hill City, South Dakota, was selected for this analysis because of the hydrogeologic characteristics of the site, and because of advantageous monitoring well placement that provided data regarding background nitrate-nitrogen concentrations entering the system and concentrations of nitrate-nitrogen in ground water exiting the system at the down-gradient boundary of the site (fig. 9). The alluvial deposit under investigation at that locality is relatively limited, with a width of about 100 feet across the site, so definition of model boundaries was more accurate than at other sites in larger deposits with less well-defined hydrogeologic contacts and less monitoring-well control. Nitrogen mass-balance models that were selected for this analysis were those of Hantzsche and Finnemore (1992), which are based primarily on recharge from wastewater and precipitation over a defined gross area, and Bauman and Schafer (1985) which includes a lateral ground-water flow component.

Table 11 shows calculation of the nitrogen mass-balance model of Hantzsche and Finnemore (1992) for the study site at the private residence in Hill City, South Dakota. Location of monitoring wells, geologic contacts, and model boundaries are shown on figure 9, and nitrate-nitrogen concentrations of samples collected from the site are given in table 6. The model predicts the resultant average concentration (n_r) of nitrate-nitrogen that enters ground water as recharge from wastewater and precipitation, calculated over a 1-year time period. The discharge of effluent into the soil averaged over the gross area (I) is based on a wastewater flow of 150 gallons per day per dwelling unit averaged over the 12,000 square feet area included in the analysis. The legal boundaries of the lot are larger than 12,000 square feet; however, only the area underlain by the actual alluvial aquifer was included in the analysis because the model is intended to predict the mass of nitrogen and the quantity of water affecting the ground water in the alluvial aquifer underlying the site. Hantzsche and Finnemore (1992) considered typical

Table 11. Calculation of nitrogen mass balance using the method of Hantzsche and Finnemore

Nitrogen Mass Balance

Expression for the average concentration of nitrate-nitrogen from on-site wastewater treatment system effluent and precipitation is given by:

$$n_r = In_w(1 - d) + Rn_b \div (I + R)$$

Parameters

n_r: Average concentration of nitrate-nitrogen in precipitation and effluent
I: Volume rate of effluent entering soil, averaged over the gross area
n_w: Nitrate-nitrogen concentration in effluent
d: Nitrate-nitrogen lost from denitrification
R: Precipitation recharge
n_b: Background nitrate-nitrogen from precipitation recharge
Area = 12,000 ft^2

Input Data

I = (150 gal/day) (365 day/yr) (0.13368 ft^3/gal) (12 in/ft) ÷ (12,000 ft^2) = 7.3 in/yr
n_w = 40 mg/L
d = 0
R = 2 in/yr
n_b = 1 mg/L

Results

n_r = (7.3 in/yr) (40 mg/L) (1 – 0) + (2 in/yr) (1 mg/L) ÷ (7.3 in/yr + 2 in/yr) = 31.6 mg/L

See Hantzsche and Finnemore (1992) for a description of the calculation method used in this table

effluent nitrate-nitrogen concentrations (n_w) to be 40 mg/L in their calculations, and therefore 40 mg/L was used in this analysis. Nitrogen losses from denitrification (d) also can be included in the equation, although conditions conducive to denitrification are thought to occur only rarely (Taylor, 2003), and no denitrification was assumed in this analysis. The average recharge rate of precipitation (R) for the study area is from Driscoll and others (2002), and background nitrate-nitrogen from precipitation is assumed to be 1.0 mg/L (Hantzsche and Finnemore, 1992). A number of simplifying assumptions are made in this mass-balance approach including uniform, complete mixing of wastewater and precipitation over the entire study area, and full conversion of ammonium to nitrate-nitrogen. This model also ignores dispersion, lateral flow, and mixing with ground-water flow entering the study area from up-gradient areas.

With the input values shown in table 11, the nitrogen mass-balance model of Hantzsche and Finnemore (1992) predicts an average nitrate-nitrogen concentration of 31.6 mg/L from the combined effects of the on-site wastewater treatment system at the site and from precipitation. Observed nitrate-nitrogen concentrations were up to 1.4 mg/L in monitoring well R20-2002-18 immediately adjacent to the drain field, and up to 1.1 mg/L in monitoring well R20-2002-11 at the down-gradient boundary of the analysis (table 6; fig. 9). In this analysis, the nitrate-nitrogen concentration predicted by the model was significantly higher than the observed concentrations in the monitoring wells. One reason for this discrepancy could be the absence of consideration of lateral ground-water flow in the mass-balance equation. It also is possible that input variables such as the volume rate of effluent entering the soil at the site and nitrate-nitrogen concentration in effluent, or assumptions such as complete mixing of wastewater and precipitation, are not representative of true conditions, and if adjusted, could result in predicted nitrate-nitrogen concentrations that are more similar to observed data. For example, the level of treatment that the effluent received in the on-site wastewater treatment system and in the subsurface environment at this site could have resulted in lower actual nitrate-nitrogen concentrations than were input into the model. Monitoring wells installed at the site also could have missed a narrow plume of elevated nitrate-nitrogen concentration, or heterogeneity within the underlying alluvial deposit could have affected the flow path of wastewater effluent.

A second nitrogen mass-balance model, published by Bauman and Schafer (1985), was applied to the same on-site wastewater treatment system using the same aquifer boundaries and dimensions (fig. 9) to compare the results of the different approaches and to gain insights into the sensitivity of these models to different parameters. Table 12 shows input parameters and calculations used to quantify the water budget and nitrogen budget for the study area and the resulting nitrate-nitrogen concentration predicted for ground water exiting the system at the down-gradient boundary using the method of Bauman and Schafer (1985).

The mass-balance model of Bauman and Schafer (1985) includes lateral ground-water flow entering from the up-gradient boundary (W_g) calculated with Darcy's law. Hydraulic conductivity (K) was estimated as 1 x 10^{-3} centimeters per second from hydraulic conductivity rates published by Freeze and Cherry (1979) for unconsolidated sand and silt. An average aquifer thickness (b) was estimated as 10 feet from monitoring well data (app. C), and aquifer width (w) was estimated as 100 feet from the lateral extent of alluvial deposits at the site (fig. 9). Hydraulic gradient (dh/dl) was obtained from ground-water elevations in monitoring wells R20-2002-10 and R20-2002-11 (table 4). A concentration of 0.5 mg/L was entered for background nitrate-nitrogen concentration in ground water (N_g) which was the highest concentration recorded at the up-gradient boundary of the study area in monitoring well R20-2002-10. As in the previous model, the average recharge rate of precipitation ® for the study area is from Driscoll and others (2002), and background nitrate-nitrogen from precipitation is assumed to be 1.0 mg/L. The discharge of effluent from the on-site wastewater treatment system at the site is estimated as 170 liters per day per person (Bauman and Schafer, 1985) for five persons for a period of 1 year. Bauman and Schafer (1985) cited a nitrate-nitrogen concentration level of 62 mg/L as an average concentration in on-site wastewater treatment system effluent from numerous previous studies, and therefore 62 mg/L is used in this analysis as well. In comparison, the U.S. Environmental Protection Agency reports concentrations of nitrate-nitrogen in on-site wastewater treatment system effluent ranging from 40 to 100 mg/L followed by 10 to 20 percent removal at depths of

Table 12. Calculation of nitrogen mass balance using the method of Bauman and Schafer

Parameters and Input Data

W_g: Ground water entering up-gradient boundary, determined from Darcy's law

$$W_g = Kb(dh/dl)w$$

Hydraulic conductivity (K) = $(1.0 \times 10^{-3}$ cm/sec) (1 ft/30.48 cm) = 3.3×10^{-5} ft/sec
Aquifer mixing thickness (b) = 10 ft
Hydraulic gradient (dh/dl) = (84.54 ft – 73.39 ft) ÷ 120 ft = 0.093
Aquifer width (w) = 100 ft

N_g: Background nitrate-nitrogen in ground water = 0.5 mg/L
W_r: Precipitation recharge = 2 in/yr
N_r: Nitrate-nitrogen in precipitation recharge = 1.0 mg/L
W_e: On-site wastewater treatment system effluent
N_e: Nitrate-nitrogen in effluent = 62 mg/L
Area = 12,000 ft^2

Water Budget

W_g = $(3.3 \times 10^{-5}$ ft/sec) (10 ft) (0.093) (100 ft) = 3.1×10^{-3} ft^3/sec
= $(3.1 \times 10^{-3}$ ft^3/sec) (60 sec/min) (60 min/hr) (24 hr/day) (365 day/yr) = 98,000 ft^3/yr

W_r = (2.0 in/yr) (12,000 ft^2) (1 ft/12 in) = 2,000 ft^3/yr

W_e = (170 L/day/person) (5 persons) (365 day/yr) (0.03531467 ft^3/L) = 11,000 ft^3/yr

Total water input = $W_t = W_g + W_r + W_e$

W_t = 98,000 ft^3/yr + 2,000 ft^3/yr + 11,000 ft^3/yr = 111,000 ft^3/yr

Nitrogen Budget

N_g = (98,000 ft^3/yr) (0.5 mg/L) (28.3168 L/ft^3) = 1,388,000 mg/yr

N_r = (2,000 ft^3/yr) (1.0 mg/L) (28.3168 L/ft^3) = 57,000 mg/yr

N_e = (11,000 ft^3/yr) (62 mg/L) (28.3168 L/ft^3) = 19,312,000 mg/yr

Total nitrogen input = $N_t = N_g + N_r + N_e$

N_t = 1,388,000 mg/yr + 57,000 mg/yr + 19,312,000 mg/yr = 20,757,000 mg/yr

Nitrogen in Ground Water Exiting System

$N_t \div W_t$ = (20,757,000 mg/yr) ÷ (111,000 ft^3/yr) = 187.0 mg/ft^3
= (187.0 mg/ft^3) (0.03531467 ft^3/L) = 6.6 mg/L

See Bauman and Schafer (1985) for a description of the calculation method used in this table

3 to 5 feet (U.S. Environmental Protection Agency, 2002b). The area, A, under consideration is defined by the lateral extent of the alluvial aquifer and the up-gradient and down-gradient locations of monitoring wells at the site (fig. 9).

Included in the approach of Bauman and Schafer (1985) are assumptions that all nitrogen inputs are converted to nitrate-nitrogen, that there are no losses of nitrogen from well withdrawals, and that the only output from the system is through the down-gradient boundary of the study area. Complete mixing of effluent and ground water also is assumed, and nitrogen inputs from fertilizer or other sources are not considered because the intent of the analysis is to evaluate the effects from on-site wastewater treatment systems alone.

Using the input values shown in table 12, the nitrogen mass-balance model of Bauman and Schafer (1985) predicts that ground water at the down-gradient boundary of the study area would have a nitrate-nitrogen concentration of 6.6 mg/L as a result of nitrogen contributions from the on-site wastewater treatment system, from up-gradient ground water, and from precipitation. This predicted concentration is more similar to observed maximum concentrations in down-gradient monitoring wells, which were 1.4 mg/L and 1.1 mg/L in monitoring wells R20-2002-18 and R20-2002-11, respectively, than was predicted by the method of Hantzsche and Finnemore (1992). The reason for the greater accuracy of the Bauman and Schafer (1985) model can probably be attributed to the inclusion of ground-water flow from the up-gradient boundary of the study area, which more accurately reflects the water budget of the site, and therefore more accurately simulates the amount of dilution of nitrate-nitrogen from the sources of recharge.

Results from both of the above methods are sensitive to variations of input parameters such as nitrate-nitrogen concentration of effluent and precipitation recharge in the modeled area. Predicted nitrate-nitrogen concentrations with the method of Bauman and Schafer (1985) also vary inversely with hydraulic conductivity rates and gradient. The method of Hantzsche and Finnemore (1992) also is sensitive to estimated denitrification rates which decrease the predicted nitrate-nitrogen concentration as the rate of denitrification is increased in the equation.

A common goal or challenge associated with nitrogen mass-balance modeling efforts is to determine acceptable densities of on-site wastewater treatment systems for a proposed development within a specific geographic area. As part of their evaluation of nitrate-nitrogen effects from on-site wastewater treatment systems, Hantzsche and Finnemore (1992) derived an equation that estimates the critical minimum gross acreage per developed lot, A, for a defined area with a predetermined resultant average ground-water nitrate-nitrogen concentration, n_r, such as the drinking water standard of 10 mg/L (table 13). Using the same input data that were included in the mass-balance model of Hantzsche and Finnemore (1992) in table 11, and setting 10 mg/L as the resultant average nitrate-nitrogen concentration in ground water, the estimated minimum acreage per lot was calculated as 3.36 acres/dwelling unit for the alluvial aquifer at this site (table 13). The estimated minimum acreage calculated for this site is based on numerous assumptions and should not be viewed as accurate or definitive, nor should it be applied to other sites in similar or different hydrogeologic settings.

Table 13. Calculation of critical minimum gross acreage per developed lot using the method of Hantzsche and Finnemore

Critical Minimum Gross Acreage Per Developed Lot

A = Critical minimum gross acreage per developed lot

$A = 0.01344\ W\ [n_w - dn_w - n_r] \div R\ (n_r - n_b)$

Input Data

0.01344 = Conversion factor having units: acre inch day dwelling units $yr^{-1}\ gal^{-1}$
W = Average daily wastewater flow per dwelling unit = 150 gal/day
n_w = 40 mg/L
d = 0
n_r = 10 mg/L
R = 2 in/yr
n_b = 1 mg/L

Results

A = (0.01344) (150 gal/day) [40 mg/L – (0) (40 mg/L) – 10 mg/L] ÷ (2 in/yr) (10 mg/L – 1 mg/L)
A = 3.36 acres/dwelling unit

See Hantzsche and Finnemore (1992) for a description of the calculation method used in this table

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Growth and Development Concerns

One of the concerns with respect to on-site wastewater treatment systems is the potential cumulative effect on ground-water and surface-water quality from nitrate-nitrogen contributed by large-scale, dense, residential developments. Densely spaced on-site wastewater treatment systems can result in the accumulation of soluble contaminants in ground water, and under certain conditions, can affect water-table elevations, soil treatment of contaminants, performance of individual systems, and can cause discharge to surface waters (Bauman and Schafer, 1985; Tinker, 1991; Hantzsche and Finnemore, 1992; Taylor, 2003). For example, an investigation by Tinker (1991) found nitrate-nitrogen concentrations up to 21.6 mg/L in samples from indoor and outdoor faucets in five subdivisions in Wisconsin served by relatively shallow wells (less than 120 feet) in glacial outwash and river terrace deposits. In that study, mean lot sizes within the five subdivisions ranged from 0.5 to 1.3 acres and the study locations were situated in areas with negligible up-gradient agricultural activity or other known sources of nitrate-nitrogen other than lawn fertilizer. Tinker (1991) also statistically correlated the number of on-site wastewater treatment systems up-gradient of each monitoring well using modeled contaminant plumes. Significance levels derived from Student's t-test exceeded 90 to 99.9 percent for correlations between nitrate-nitrogen at sampled wells and the number of on-site wastewater treatment systems in the calculated up-gradient area, indicating that on-site wastewater treatment systems were a source of nitrate-nitrogen in the sampled wells. Tinker (1991) also investigated fertilizer application rates in the five subdivisions and calculated that some areas received more nitrate-nitrogen from lawn fertilizer while other areas received more nitrate-nitrogen from on-site wastewater treatment systems. No information on the condition of the on-site wastewater treatment systems was reported.

Another study by Hantzsche and Finnemore (1992) evaluated three densely developed areas in California supplied by shallow sandy aquifers and found area-wide nitrate-nitrogen concentrations up to 64.9 mg/L in ground-water monitoring wells and local water-supply wells. Mean lot sizes within the three subdivisions in that study ranged from 0.33 to 0.66 acres, and the density of on-site wastewater treatment systems per acre ranged from 1.3 to 3.0 dwelling units per acre. These authors considered that nitrate-nitrogen loading from fertilizer was approximately 0.37 to 1.1 mg/L and would be substantially accounted for in the assumed background nitrate-nitrogen concentration of 0.5 to 1.0 mg/L. No information on the condition of the on-site wastewater treatment systems was reported.

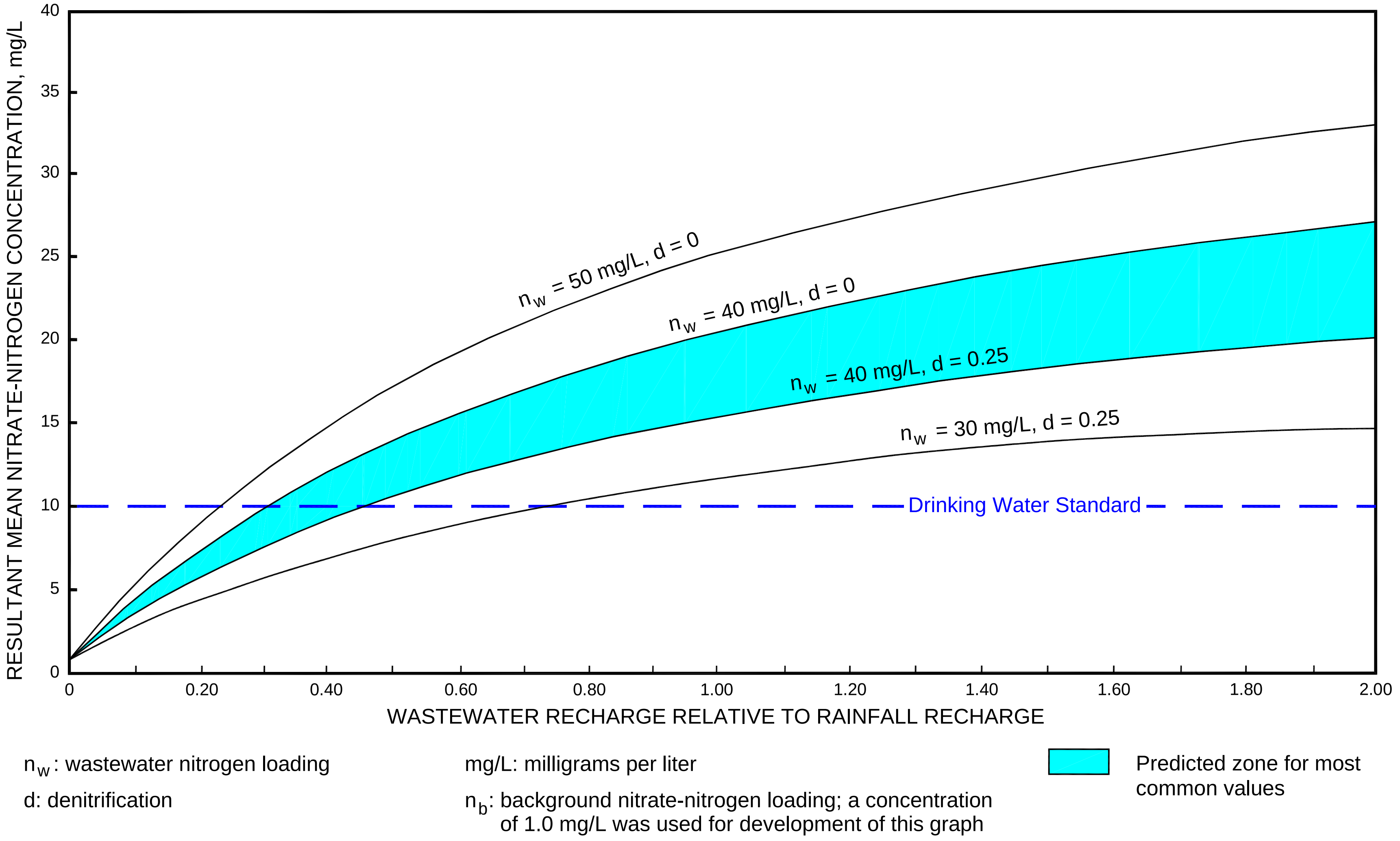
In the Black Hills area, Bad Moccasin (1986) studied ground water beneath a development of approximately 200 homes on 1-acre lots along Rapid Creek east of Rapid City, South Dakota, and reported nitrate-nitrogen concentrations up to 10.4 mg/L from residential drinking-water wells. The homes were served by shallow wells approximately 15 to 30 feet deep that were installed into a shallow alluvial aquifer consisting of sand, gravel, and clay approximately 15 to 30 feet thick. The homes also were served by on-site wastewater treatment systems although no information was provided regarding the exact locations of the drain fields with respect to the individual water wells. Bad Moccasin (1986) also noted that the distance from the ground surface to the static ground-water water level ranged from 3.4 to 11.5 feet and that some of the systems did not meet the 4 feet of vertical separation distance between the bottom of the drain field trenches and the water table as required by state regulations. A subsequent investigation in the same development (Mott and others, 2004) found similar nitrate-nitrogen concentrations and also noted a marked increase in nitrate-nitrogen, chloride, and coliform bacteria in ground water within the developed area in comparison to background levels of these parameters on the edges of the development. Mott and others (2004) further reported that the regions of highest nitrate-nitrogen, chloride, and bacterial content correlated with the high densities of on-site wastewater treatment systems rather than with the locations of large animal pastures within the development. A similar investigation of effects on shallow ground water from on-site wastewater treatment systems in the community of Piedmont, South Dakota, analyzed ground water from 428 private and public drinking-water wells in a variety of unconfined and confined aquifers ranging from 10 to 2,000 feet in depth (Bartlett & West Engineers, Inc., 1998). Results of water-quality analyses from this study indicated that fecal coliform bacteria occurred in 4 percent of the sampled wells, total coliform bacteria were present in 28 percent of sampled wells, and nitrate-nitrogen concentrations were greater than or equal to 5.0 mg/L in approximately 13 percent of the wells that were included in the investigation (Bartlett & West Engineers, Inc., 1998).

Several other investigations of residential areas served by on-site wastewater treatment systems located on alluvial deposits along Rapid Creek east of Rapid City have reported nitrate-nitrogen concentrations in ground water as high as 19.3 mg/L (Coker, 1981; Hafi, 1983; Musa, 1984; Rahn and Davis, 1986). Also in the Black Hills area, Rahn (2006) reported a trend of increasing nitrate-nitrogen background concentrations over a period of time from 1993 to 2004 in water-supply wells in the karstic Madison aquifer serving Rapid City, South Dakota (fig. 3). Possible sources for the increase in nitrate-nitrogen concentrations that Rahn (2006) identified included streamflow recharging the Madison aquifer, on-site wastewater treatment systems up-gradient from municipal wells, and fertilizers from homes and agricultural areas. Rahn (2006) suggested that commercial agriculture and mining probably were not the primary source of the increased nitrate-nitrogen concentrations because there are no feedlots, very few farm lands, and very little mining with explosives in the source-water areas for these wells. Long and others (2006) also reported that population growth and development on and up-gradient from aquifer recharge areas in recent years may have influenced plumes of elevated nitrate-nitrogen concentrations that were identified within or near conduit flow paths in the Madison aquifer on the eastern flank of the Black Hills.

One of the greatest challenges associated with nitrate-nitrogen discharged from on-site wastewater treatment systems is prediction of the future effects to ground-water and surface-water resources from newly proposed high-density residential developments. Hantzsche and Finnemore (1992) presented a mass-balance equation (table 11) that yields the average concentration of nitrate-nitrogen in ground water resulting from large numbers of on-site wastewater treatment systems within a defined geographic area. Using this equation, these authors developed curves of predicted values for resultant nitrate-nitrogen concentrations as a function of effluent quality, denitrification, and the ratio of wastewater recharge relative to precipitation recharge (fig. 15).

The graphical solutions shown in figure 15 were obtained from solving the mass-balance equation in table 11 with typical ranges of variables. The resulting curves in figure 15 can be used by others as an aid in estimating appropriate input values for the equation and in identifying situations of potential concern. Simplifying assumptions pertaining to nitrogen sources, mixing of effluent and precipitation, and other factors are incorporated into this mathematical model; however, comparisons of field results from actual study areas to predicted concentrations indicated that observed concentrations fell within the envelope defined by the curves of predicted values (Hantzsche and Finnemore, 1992).

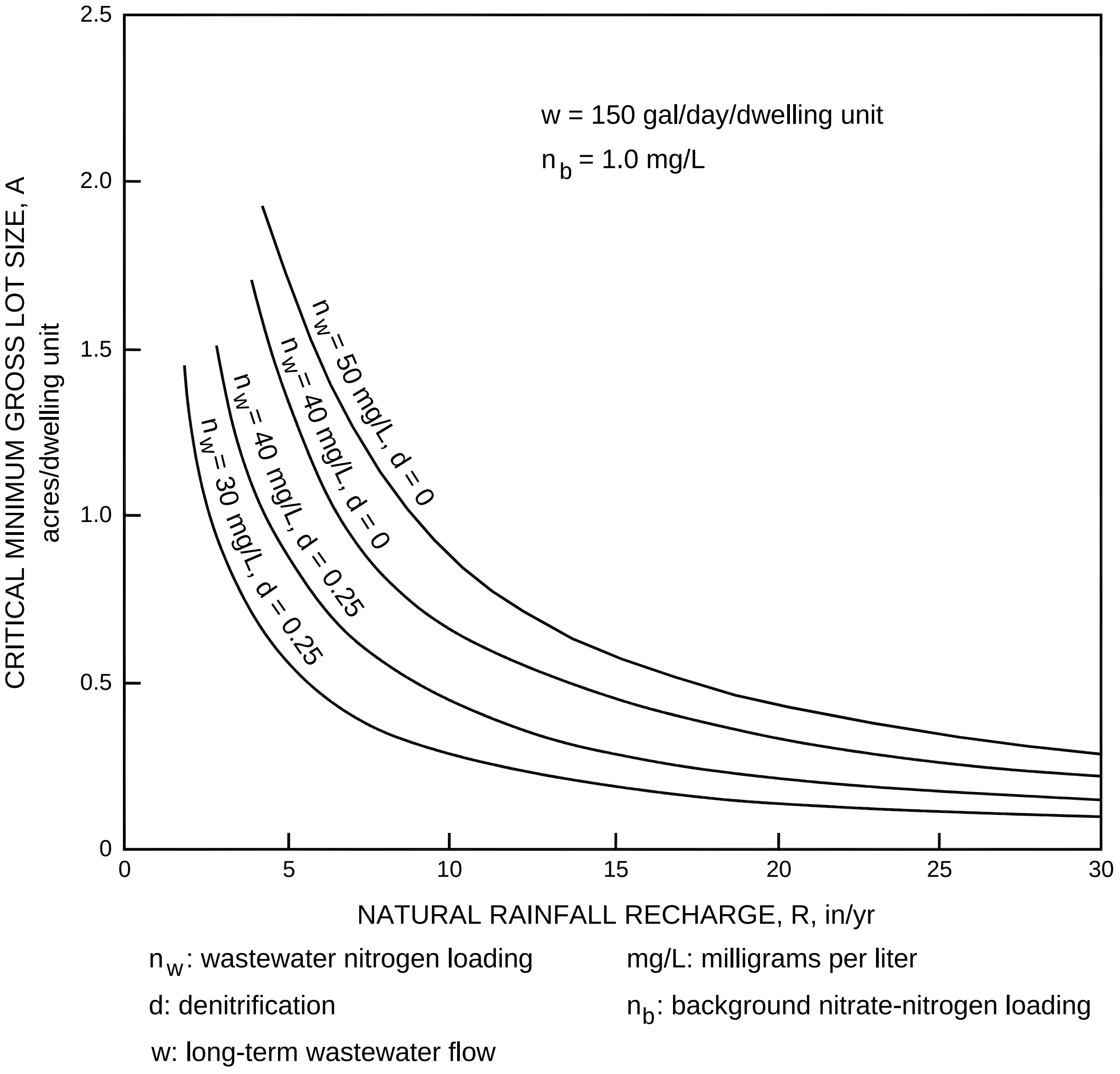
The graph in figure 15 shows the high level of sensitivity of the equation to variations in input values for wastewater nitrogen concentration (n_w) and the rate of denitrification (d) that is assumed. The graph also shows the strong influence of precipitation recharge on resultant values, and the data indicate that the risk of potential problems with nitrate-nitrogen contamination of ground water is greatest in areas of low precipitation and high development density (Hantzsche and Finnemore, 1992). Another nitrogen mass-balance model presented by Bauman and Schafer (1985) incorporates lateral ground-water flow (table 12), and analysis of model results by these authors using a range of variables also indicated that arid and semi-arid regions could be at greater risk to contamination. Bauman and Schafer (1985) also reported that the variables most responsible for affecting the resultant predicted nitrate-nitrogen concentration were hydraulic conductivity, ground-water gradient, precipitation recharge, residential density, and wastewater nitrate-nitrogen concentration.



Adapted from Hantzche and Finnemore (1992)

Figure 15. Graph showing resultant ground-water nitrate-nitrogen concentration as a function of effluent quality, denitrification, and wastewater recharge relative to rainfall recharge.

Hantzsche and Finnemore (1992) derived a second equation (table 13) that allows estimation of the critical minimum gross acreage per developed lot based on a predetermined desired average resultant nitrate-nitrogen concentration such as the drinking water standard of 10 mg/L (Hantzsche and Finnemore, 1992). Figure 16 is a graph of critical minimum gross acreage per lot with respect to precipitation recharge for a range of values for denitrification and for nitrate-nitrogen concentration in wastewater. The resultant plotted curves indicate that the influence of rainfall recharge is particularly significant, and consequently, larger lot sizes may be appropriate in arid regions.



Adapted from Hantzche and Finnemore (1992)

Figure 16. Graph showing influence of effluent quality, denitrification, and rainfall recharge on critical minimum lot size.

SUMMARY AND CONCLUSIONS

The primary purpose of this investigation was to provide information regarding the potential effects of wastewater treatment systems on ground-water quality within specific hydrogeologic settings in the Black Hills area. Alluvial aquifers and karst limestone aquifers were selected as the focus of investigation because of their importance to local drinking-water supplies, their sensitivity to contamination, and the continuing proliferation of on-site wastewater treatment systems in areas underlain by these aquifers. An effort was made to investigate properly maintained on-site wastewater treatment systems with no known functional problems. Wastewater treatment lagoons for the city of Hill City, South Dakota, occur on unconsolidated alluvial deposits and also were included in this investigation. The study approach in these alluvial hydrogeologic settings included installation and sampling of monitoring wells at strategic locations around the on-site wastewater treatment systems and wastewater treatment lagoons to evaluate water-quality effects on shallow ground water. Water-quality analyses also were performed on surface water serving as recharge to karst limestone aquifers and on ground water discharged from these same limestone aquifers down-gradient from developed areas.

This investigation was not an exhaustive characterization of the effects of on-site wastewater treatment systems on water quality within these hydrogeologic settings, and the results herein should not be considered as conclusive or statistically valid. Many more analyses within these and other hydrogeologic settings could be completed to more fully characterize the effects of wastewater treatment systems in an area such as the Black Hills. Furthermore, only a limited list of indicator parameters was evaluated in this study. Analysis of a more comprehensive suite of potential contaminants including viruses, toxic organic compounds, and other constituents could reveal significant additional information regarding the effects of on-site wastewater treatment systems on ground-water quality. Geologic heterogeneity, variable ground-water flow rates, climatic variations, anisotropic flow regimes, mixing ratios of different age waters, and lag times between recharge and discharge in karst settings further complicate accurate interpretations of actual effects.

Three individual on-site wastewater treatment systems located within alluvial aquifers overlying Precambrian bedrock units were evaluated. A continuously occupied residential site near Hill City, South Dakota, was located in an ephemeral stream valley with no nearby perennial streamflow. Another continuously occupied residential site near the community of Johnson Siding, South Dakota, was located immediately adjacent to Rapid Creek, which is a perennial stream. A third study site at the Rocky Knolls Golf Course in Custer, South Dakota, experienced seasonal usage and was located adjacent to a permanent impoundment on French Creek.

Results of ground-water quality analyses at the Hill City residential study site included nitrate-nitrogen concentrations from less than 0.1 to 1.4 mg/L, detections of *E. coli* and fecal coliform bacteria, and total phosphorus concentrations from 0.28 to 10 mg/L. Analysis of a sample from a monitoring well located approximately 6 feet down-gradient from the edge of the drain field showed the greatest nitrate-nitrogen concentration and one detection of *E. coli* bacteria. Analysis of a sample from a monitoring well located approximately 60 feet down-gradient from the drain field showed one detection of fecal coliform bacteria and two detections of *E. coli* bacteria. The greatest total phosphorus concentration occurred in a sample from a monitoring well up-gradient from the drain field and may have been affected by factors other than the on-site wastewater treatment system on the property.

Results of ground-water quality analyses at the Johnson Siding study site included nitrate-nitrogen concentrations from less than 0.1 to 1.3 mg/L, ammonia concentrations from less than 0.02 to 0.18 mg/L, chloride concentrations from less than 3 to 7 mg/L, and conductivity values from 295 to 422 microsiemens. These sample results all were from a monitoring well located approximately 70 feet down-gradient from the drain field, and they were greater in concentration or numeric value than sample results from the up-gradient (background) monitoring wells at the site. *E. coli* bacteria were detected in one ground-water sample at this study site; however, it was from an up-gradient monitoring well adjacent to Rapid Creek. A maximum total phosphorus concentration of 4.05 mg/L was recorded from analysis of a sample from a monitoring well up-gradient from the drain field; however, this sample could have been affected by factors other than the on-site wastewater treatment system at the site.

Results of ground-water quality analyses at the Rocky Knolls Golf Course in Custer, South Dakota, included nitrate-nitrogen concentrations from less than 0.1 to 4.2 mg/L, chloride concentrations from 15 to 187 mg/L, total phosphorus concentrations from 0.08 to 2.3 mg/L, dissolved phosphorus concentrations from 0.017 to 0.119 mg/L, conductivity values from 542 to 1,070 microsiemens, and detections of both fecal coliform and *E. coli* bacteria. Maximum concentrations and values for nitrate-nitrogen, chloride, and conductivity all occurred in an up-gradient monitoring well immediately adjacent to the golf course and may reflect the effects of recycled municipal wastewater which is used to irrigate the golf course and/or fertilizer that is applied to the golf course. The maximum total phosphorus and dissolved phosphorus concentrations and all detections of bacteria occurred in two monitoring wells that were near a surface-water body (impoundment on French Creek) which could have influenced these results.

Detections of bacteria in ground water at these three study sites were obtained in very close proximity to the individual on-site wastewater treatment system drain fields in areas where drinking-water wells could not be legally installed due to set-back requirements. None of the other parameters that were analyzed near individual on-site wastewater treatment systems exceeded drinking-water standards established by the U.S. Environmental Protection Agency (1994, 2002b) and the South Dakota Department of Environment and Natural Resources (2003).

Surface-water samples were analyzed from Bear Butte Creek, Boxelder Creek, Rapid Creek, and Spring Creek just upstream from streamflow loss zones to the Madison aquifer. Results of surface-water quality analyses for these streams included nitrate-nitrogen concentrations from less than 0.1 to 4.8 mg/L, conductivity values from 259 to 1,237 microsiemens, detections of *E. coli* bacteria in all four streams, and detections of fecal coliform bacteria in all streams except Rapid Creek. It is not unusual to find *E. coli* and fecal coliform bacteria within streams such as these because of wildlife and livestock that contain these types of organisms within their digestive tracts; however, a concurrent study involving DNA ribotyping of fecal coliform bacteria from surface water in Spring Creek yielded human signatures for 35 percent of samples in which bacteria were detected (Schwickerath, 2004). Nitrate-nitrogen concentrations and conductivity values were substantially greater in Bear Butte Creek than in other sampled streams which probably reflect the effects of treated water that is discharged into upstream tributaries from the former Gilt Edge Mine site. Nitrate-nitrogen concentrations from Boxelder Creek, Rapid Creek, and Spring Creek were 0.2 mg/L or less in all samples, and conductivity values ranged from 310 to 405 microsiemens for these three streams. Chloride concentrations were up to 21 mg/L and 18 mg/L in Bear Butte Creek and Spring Creek, respectively, and chloride levels were less than or equal to 11 mg/L in Boxelder Creek and less than or equal to 5 mg/L in Rapid Creek.

Ground water within limestone aquifers and spring flow from these aquifers were sampled from eight public water-supply systems and two artesian springs on the eastern flank of the Black Hills. Nitrate-nitrogen concentrations were 0.5 mg/L or less for these sources with the exception of the Sturgis public water-supply well which exhibited three nitrate-nitrogen concentrations of 1.2 mg/L. The Sturgis public water-supply well also yielded a maximum conductivity value of 601 microsiemens which was the highest value recorded from any source of water from the Madison aquifer in this study. *E. coli* bacteria were detected in the Boulder Park water-supply system in three water samples and in one water sample from the Copper Oaks water-supply system. The Copper Oaks and Highland Hills public water-supply systems both exhibited chloride concentrations up to 18 mg/L, possibly reflecting the influence of surface water from Spring Creek which recharges the limestone aquifer in the vicinity of these wells.

A ground-water monitoring network consisting of five monitoring wells was installed to investigate subsurface conditions around the wastewater treatment lagoons at Hill City. These lagoons are estimated to leak millions of gallons of water annually. Results of water-quality analyses from the monitoring-well network included nitrate-nitrogen concentrations from less than 0.1 to 10.1 mg/L, total Kjeldahl nitrogen concentrations from 0.37 to 12.6 mg/L, ammonia concentrations from less than 0.02 to 10.0 mg/L, total phosphorus concentrations from 0.054 to 1.96 mg/L, and chloride concentrations from 17 to 89 mg/L. Conductivity values ranged from 373 to 1,156 microsiemens, *E. coli* bacteria were detected in one monitoring well near lagoon #1, and fecal coliform bacteria were detected in a background monitoring well up-gradient of the wastewater treatment lagoons. Concentrations of nitrate-nitrogen, ammonia, dissolved phosphorus, and total Kjeldahl nitrogen become progressively elevated from south to north across the site, reaching their greatest concentrations along the eastern boundaries of lagoons #1 and #2. Concentrations of these constituents then decrease by varying amounts along the northern, down-gradient boundary of the site. The monitoring well up-gradient of the wastewater treatment lagoons is located approximately 40 feet from Spring Creek and it is possible that Spring Creek is the source of the fecal coliform bacteria detected in this well.

Analyses of wastewater directly from the lagoons indicated that total phosphorus, total Kjeldahl nitrogen, and dissolved phosphorus were more concentrated in the lagoons than in the adjacent ground water and that conductivity values and chloride concentrations within the lagoons were similar to those for ground-water samples from the monitoring-well network. Concentrations of ammonia in lagoon waters ranged from 10.7 mg/L in lagoon #1 to 5.5 mg/L in lagoon #3 in comparison with ground-water ammonia concentrations which were all below 5.8 mg/L except one sample near the eastern edge of lagoon #2 that was 10.0 mg/L. Nitrate-nitrogen concentrations in lagoon waters were less than or equal to 0.8 mg/L; however, in ground-water samples adjacent to the lagoons, nitrate-nitrogen concentrations ranged from less than 0.1 to 10.1

mg/L. The discrepancy between nitrate-nitrogen concentrations in lagoon waters and ground water probably reflects conversion of ammonia to nitrate-nitrogen at infiltrative surfaces and within alluvial sediments beneath the lagoons. Although caffeine was detected within the wastewater lagoons, it was not detected in ground-water samples from around the lagoon system. Water-level measurements showed that Spring Creek changed from a losing stream up-gradient of the lagoon system to a gaining stream in the vicinity of the lagoons, illustrating the effect of water from the lagoons as it recharges the local ground-water system and elevates the local water table.

Information resulting from a total maximum daily load assessment of Spring Creek that was in progress concurrently with this investigation has documented that fecal coliform bacteria from human sources were present in surface water in Spring Creek upstream and downstream from the Hill City wastewater treatment lagoons in every monthly sample collected during that study, which occurred from May 2002 through July 2003 (Schwickerath, 2004) and overlapped with the collection of ground-water quality samples from this investigation. Schwickerath (2004) also reported that there was no significant change in bacteria concentrations in Spring Creek either upstream or downstream from the sewage lagoons over the course of the investigation during which time streamflow varied from approximately 0.6 to 50 cubic feet per second. The occurrence of human fecal coliform bacteria upstream from the sewage lagoons shows that sources of human waste other than the sewage lagoon system also are affecting surface water in Spring Creek. These other sources of human fecal coliform bacteria may include poorly functioning or leaking on-site wastewater treatment systems, “outhouses” or other types of pit privies that are no longer allowed to be constructed but could still be in use from past decades when they were allowed, various types of “alternative” systems that are being used for domestic sewage disposal, or even direct piping of raw sewage into Spring Creek surface water.

Swanson (2004) evaluated nutrients in surface water in Spring Creek upstream and downstream from the Hill City wastewater treatment lagoons also as part of the total maximum daily load assessment for Spring Creek. Swanson (2004) reported nitrate-nitrogen concentrations from 0.025 to 0.39 mg/L, ammonia concentrations from 0.025 to 0.09 mg/L, total Kjeldahl nitrogen concentrations from 0.025 to 0.98 mg/L, and total phosphorus concentrations from 0.011 to 0.17 mg/L from samples collected upstream from the Hill City wastewater treatment lagoons. Downstream from the Hill City wastewater treatment lagoons, nitrate-nitrogen concentrations were 0.025 to 0.57 mg/L, ammonia concentrations were 0.025 to 0.51 mg/L, total Kjeldahl nitrogen concentrations were 0.025 to 2.7 mg/L, and total phosphorus concentrations were 0.02 to 1.8 mg/L. Swanson (2004) concluded from these and other data that the primary source of phosphorus to Spring Creek was from stormwater runoff in the vicinity of Hill City and that the lagoons were not a significant source of phosphorus to Spring Creek. However, a computer model of nutrient loading in Spring Creek indicated that the nitrogen load, including nitrate-nitrogen, total Kjeldahl nitrogen, and ammonia, increased by 24 percent in the vicinity of the sewage lagoons at the Hill City wastewater treatment lagoons (Swanson, 2004). Ground-water samples from the monitoring-well network around the lagoons also were elevated in concentrations of nitrate-nitrogen, total Kjeldahl nitrogen, and ammonia. These data together with nutrient data for Spring Creek presented by Swanson (2004) indicate that the Hill City wastewater treatment lagoons are a source of nitrogen compounds that affect ground water and surface water down-gradient of the lagoons.

Nitrate-nitrogen migrates readily with ground water, usually with little retardation or denitrification, and it is a source of concern with respect to ground-water quality effects from on-site wastewater treatment systems. Therefore, nitrogen mass-balance models were applied to the study site at the residence in Hill City, South Dakota, to estimate the resultant average concentration of nitrate-nitrogen in ground water at the site as a result of effluent from the on-site wastewater treatment system. The mass-balance model presented by Hantzsche and Finnemore (1992) primarily considers nitrogen inputs from wastewater and precipitation, and this model predicted an average nitrate-nitrogen concentration of 31.6 mg/L for ground water exiting the residential site at Hill City. The nitrogen mass-balance model published by Bauman and Schafer (1985) includes lateral ground-water flow in the calculation of nitrate-nitrogen concentration, and this model predicted a nitrate-nitrogen concentration of 6.6 mg/L in ground water exiting the study site which more closely approximated the maximum nitrate-nitrogen concentration of 1.4 mg/L that was observed in ground-water samples from monitoring wells down-gradient from the drain field at this site. The critical minimum acreage per developed lot for a hydrogeologic setting such as this also was calculated as 3.36 acres per dwelling unit using a resultant average ground-water nitrate-nitrogen concentration of 10 mg/L for the area defined in the analysis. The estimated minimum acreage calculated for this site is based on numerous assumptions and should not be viewed as accurate or definitive, nor should it be applied to other sites in similar or different hydrogeologic settings.

Climatic conditions affected water-quality parameters in several respects. Ground-water samples from monitoring wells around on-site wastewater treatment systems generally showed maximum nitrate-nitrogen concentrations during the spring when precipitation and streamflow were highest; however, ammonia concentrations and conductivity values were usually greatest under low streamflow and precipitation conditions. Detections of bacteria were recorded under a variety of precipitation and streamflow conditions in ground-water samples around on-site wastewater treatment systems. Surface-water samples from streams providing recharge to the Madison aquifer generally exhibited elevated parameter concentrations and values when precipitation and streamflow were low; however, sources of discharge from the Madison aquifer did not clearly reflect the effect of climatic variations on water quality. Ground-water samples from monitoring wells at the Hill City wastewater treatment lagoons also exhibited elevated parameter concentrations during low streamflow conditions; however, sample concentrations and values showed no direct correlation with precipitation. Collection of water-quality samples on a quarterly, seasonal schedule allowed evaluation of potential effects under varying climatic conditions; however, it is possible that short-term fluctuations in water-quality parameters could have been undetected, particularly with respect to surface water. More frequent sampling would provide more complete characterization of water-quality effects resulting from rapid climatic changes such as storm events.

Future Research

As suburban and residential development continues, the number of on-site wastewater treatment systems will increase. The result will be an increasing need for information pertaining to many aspects of on-site wastewater treatment systems and the effects of these systems on water quality and the health of the public. Although much is known about processes within on-site wastewater treatment systems, many of the complex relationships between treatment, hydraulic processes, and the factors that control their behavior are not well understood. Research needs range in scale from evaluation of the performance and effects of individual on-site wastewater treatment systems, to analysis of effects from dense concentrations of on-site wastewater treatment systems, to more broad investigations at the watershed and geologic setting scale.

Research needs at the level of the individual on-site wastewater treatment system include better understanding of physical, chemical, and biological processes at infiltrative surfaces in drain fields and within the underlying unsaturated zone. More reliable performance indicators, modeling tools, monitoring techniques, and testing methods for individual on-site wastewater treatment systems also need to be developed. Micro-scale analysis of the fate, transport, and attenuation of pathogens, nutrients, and other potential contaminants under unsaturated and saturated conditions is important to more fully characterize effects on ground-water quality.

One of the more difficult challenges related to evaluation of ground-water quality effects from on-site wastewater treatment systems lies in obtaining representative ground-water samples from the unsaturated zone beneath drain-field infiltrative surfaces. Soil water pressures are less than atmospheric pressure in the unsaturated zone, and standard monitoring wells do not produce water under that condition. This is particularly significant in evaluations of karst units such as the Madison aquifer in the Black Hills where the unsaturated zone can be hundreds of feet thick and is characterized by anisotropic flow. Therefore, specialized equipment and unique approaches may be needed in the attempt to quantify effects on water quality and water quantity from on-site wastewater treatment systems in the unsaturated zone. Lysimeters use negative pressure to draw water from the soil in unsaturated situations and they can be installed to depths exceeding 55 feet depending on the geologic media (Weight and Sonderegger, 2001). Further research using lysimeters in unsaturated sediments beneath on-site wastewater treatment systems could yield additional information regarding contaminant migration to underlying ground water, although erratic flow rates, small sample size, and other difficulties present additional challenges. Angled drill holes also might be used to install lysimeters or other sample devices directly beneath drain fields without disturbing drain-field processes.

Further investigation of the water-quality effects from dense concentrations of on-site wastewater treatment systems is of importance and often is challenging. Future research on this subject will likely range from traditional approaches including analysis of contaminants such as nitrogen and bacteria, to evaluation for viruses and other pathogenic microorganisms, to investigation of a host of potential “emerging contaminants” such as preservatives, detergents, disinfectants, pharmaceuticals, fire retardants, hormones, polycyclic aromatic hydrocarbons, solvents, pesticides, plasticizers, and many others. Additional efforts also should be made to identify methods with which to successfully target, evaluate, and characterize the actual effects to water quality from dense concentrations of on-site wastewater treatment systems within specific hydrogeologic settings.

At scales encompassing watersheds and aquifer recharge areas, there is a need for information regarding the collective effects of on-site wastewater treatment systems on ground-water and surface-water quality. Reliable information at this scale is necessary for questions pertaining to minimum lot sizes, determination of minimum separation distances, discrimination of pathogens and nutrients discharged to receiving waters, and for providing information for total maximum daily load studies, modeling, and water management decisions. Better risk categorization models and indexing approaches to classify and characterize aquifer vulnerability to contamination from on-site wastewater treatment systems and other contaminant sources also are needed at the scale of watersheds and aquifer-recharge areas. Soil thickness and composition are important factors in treatment of effluent from on-site wastewater treatment systems. More accurate information regarding soil thickness would assist land management and use decisions. Variations in soil composition affect a variety of wastewater treatment processes and thus, additional information on soil composition would enhance the understanding of the potential effects of on-site wastewater treatment systems on water quality.

Aquifer Vulnerability Concerns

Karst limestone aquifers and shallow alluvial aquifers are two of the primary sources of drinking water in the Black Hills, and vulnerability of these aquifers to contamination is of significant concern. Previous investigations in the Black Hills and other areas have documented ground-water quality effects from on-site wastewater treatment systems in both of these hydrogeologic settings. Evidence has included violation of drinking-water maximum contaminant levels, illnesses resulting from pathogenic microorganisms, rising background concentrations of nitrate-nitrogen in ground waters, and other undesirable effects to ground-water quality. Numerous studies have documented effects from nitrate-nitrogen, fecal coliform bacteria, and other contaminants in shallow ground water beneath and down-gradient from densely spaced residential developments. Estimates of failure rates for existing on-site wastewater treatment systems in the United States range from 10 to 20 percent, and if these rates are assumed to be valid for the Black Hills area, there could be approximately 1,000 to 2,000 on-site wastewater treatment systems currently failing in the central Black Hills, which is the recharge area for the primary local aquifers. Studies of ground-water flow rates in the Madison aquifer using dyes and other tracers have demonstrated travel times of hours or days from streamflow loss zones to down-gradient drinking water wells, clearly demonstrating that this aquifer is highly sensitive and is characterized by conduit flow in some areas. In addition, residential development continues in the Black Hills area, contributing increasing quantities of wastewater within these sensitive aquifer recharge areas. Collectively, these issues present difficult challenges to protection of water quality in the Black Hills.

Nitrate-nitrogen from on-site wastewater treatment systems and other sources is of particular concern in both shallow alluvial and karst hydrogeologic settings. Numerous investigations in the Black Hills and other areas have documented nitrate-nitrogen concentrations in excess of drinking-water standards in ground water beneath densely spaced residential areas on alluvial aquifers or other unconfined aquifers. These investigations also have shown that the risk of contamination from nitrate-nitrogen is greater in areas of low precipitation. Because nitrate-nitrogen is not readily attenuated in the subsurface, there also is a risk of cumulative effects on water quality as numbers of on-site wastewater treatment systems increase within watersheds and aquifer-recharge areas. Pathogenic microorganisms from on-site wastewater treatment systems also pose a potential public health risk if wastewater contaminates water resources. This risk is further heightened in areas such as the Black Hills that are characterized by karst features including interconnected ground water and surface water and solution enhanced fractures, caverns, and conduits which may allow extremely rapid ground-water flow velocities in an anisotropic subsurface environment.

Most state regulations for on-site wastewater treatment systems are based on design and installation specifications, lateral and vertical minimum separation distances, percolation tests, wastewater flow rates, and other factors; however, they generally do not address issues such as aquifer sensitivity or cumulative effects from increased housing density. Aquifer sensitivity, or susceptibility, is defined as the measure of ease with which water enters and moves through an aquifer, and characterization of aquifer sensitivity incorporates intrinsic aquifer features such as rock type, transmissivity rates, fractures, karst features, and other factors that affect the ability of water and potential contaminants to enter and move through an aquifer. Karst limestone aquifers and shallow alluvial aquifers possess characteristics that can cause them to be categorized as sensitive. Therefore, additional protective measures may be advisable for sensitive aquifers in the Black Hills and similar areas to protect the quality of water resources as suburban development introduces increasing risks of contamination.

Given the evidence of the effects of on-site wastewater treatment systems on surface-water and ground-water quality as shown in previous studies in the Black Hills, an inspection program to evaluate the condition and functionality of existing on-site wastewater treatment systems could be an important component in the management and mitigation of effects of on-site wastewater treatment systems. Increasing minimum lot sizes, soil depth requirements, and lateral separation distances from karst features and surface waters also are measures that could be considered in protecting drinking-water supplies in hydrogeologic settings such as karst limestone and shallow alluvial deposits. Nitrogen and pathogenic microorganisms contributed by on-site wastewater treatment systems are specific concerns, and targeting of these particular contaminants for reduction may be advisable for adequate water-quality protection. More advanced wastewater treatment through means such as evapotranspiration systems or aerobic pretreatment, and reduction of hydraulic loads through larger drain fields and low-flow plumbing fixtures also could provide further protection to water resources.

Aquifer-recharge areas in the Black Hills and similar areas often extend beyond jurisdictional limits such as city, county, or state boundaries, and cooperation between these entities will be necessary for the comprehensive planning and watershed-based approach that are required to achieve appropriate protection of water resources in these settings. Continuing suburban and residential development within sensitive aquifer-recharge areas heightens the need for cooperative water-resource protection efforts in these areas. Therefore, education of policy-makers and homeowners regarding the potential effects and the proper location, design, and maintenance of on-site wastewater treatment systems is crucial to public understanding and acceptance of the need for better planning and growth management in the Black Hills and other developing areas.

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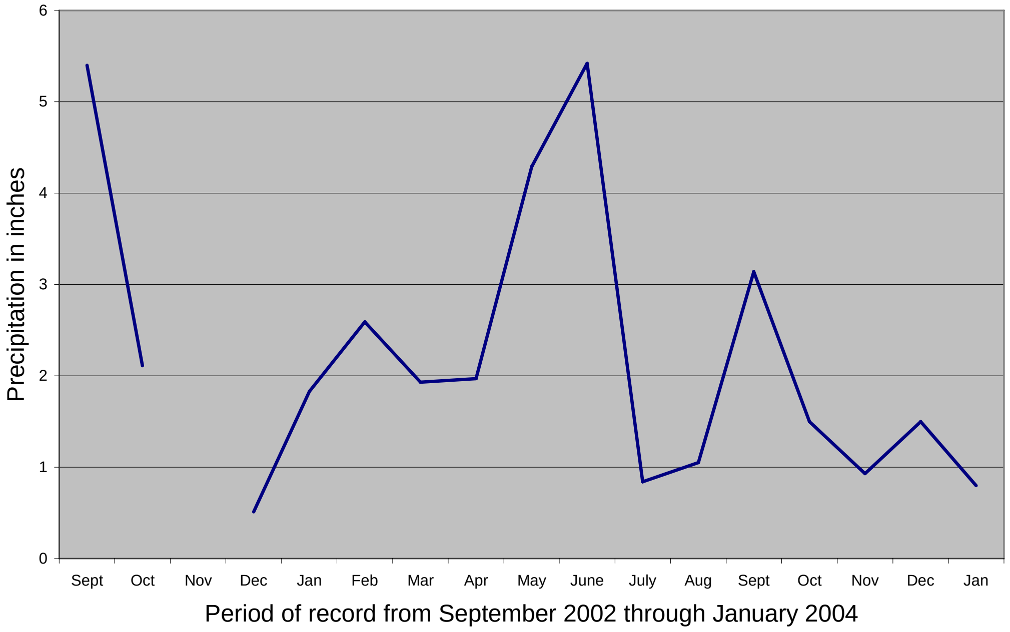
APPENDIX A

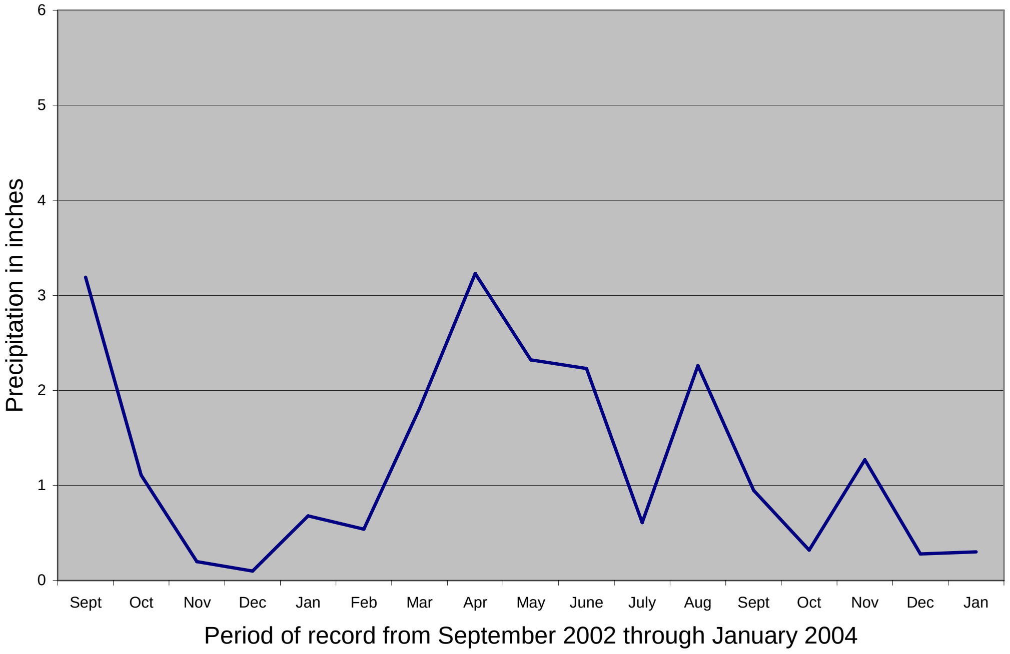
PRECIPITATION DATA

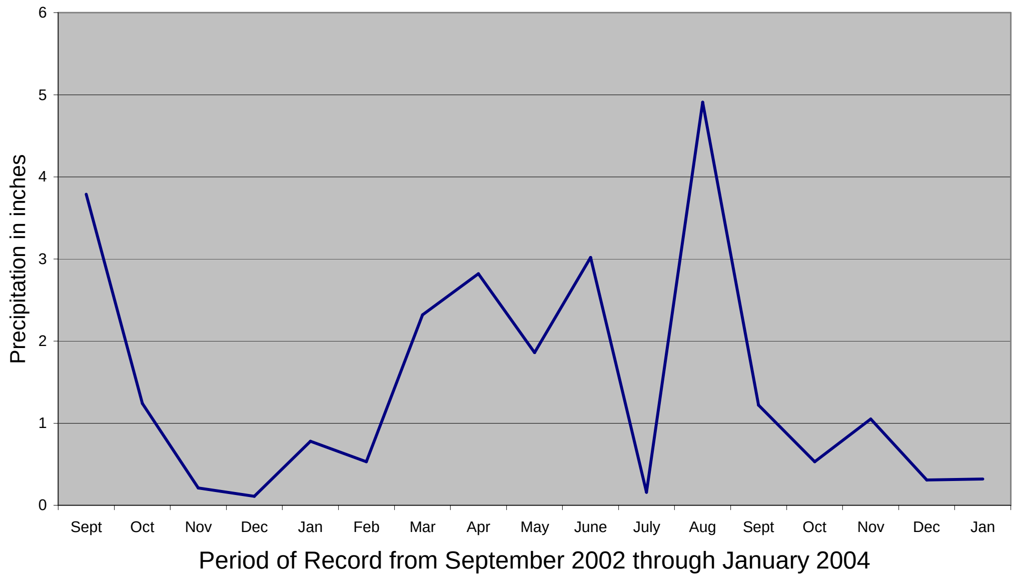
Information presented in this appendix was obtained from http://www.ncdc.noaa.gov/oa/ncdc.html.

Precipitation Data from National Oceanic and Atmospheric Administration
ID no. 394834, Lead, South Dakota

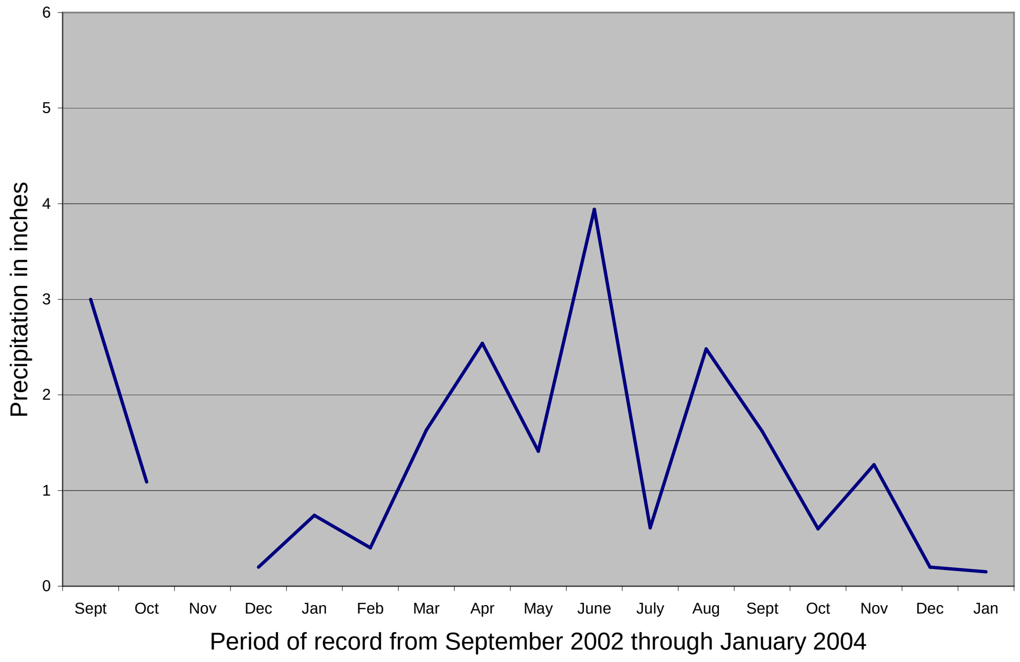


Precipitation Data from National Oceanic and Atmospheric Administration
ID no. 396427, Pactola Dam, South Dakota





Precipitation Data from National Oceanic and Atmospheric Administration
ID no. 393868, Hill City, South Dakota



Precipitation Data from National Oceanic and Atmospheric Administration
ID no. 392087, Custer, South Dakota

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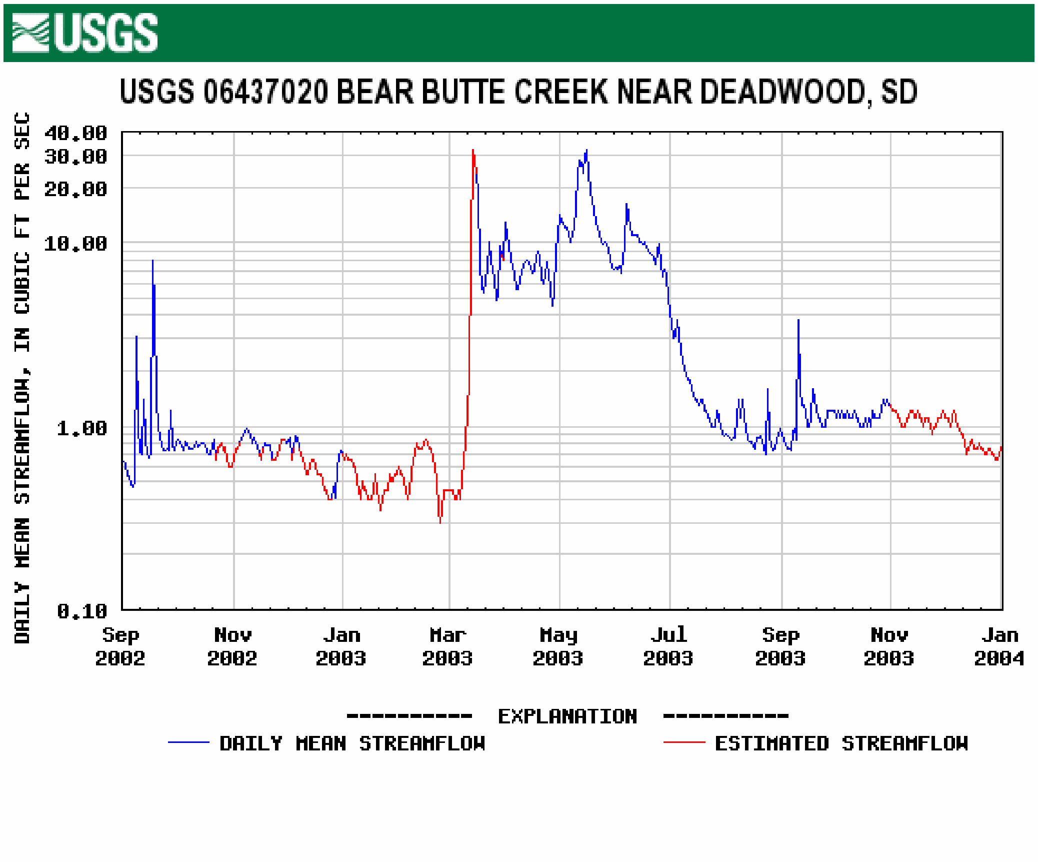
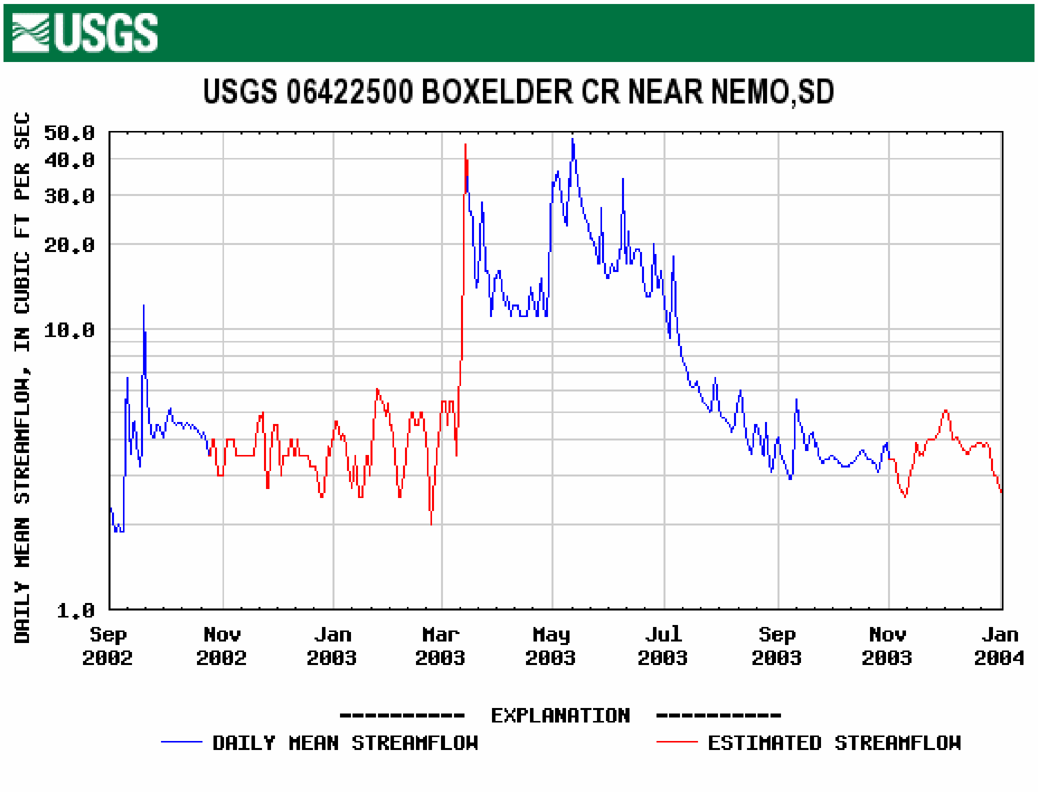
APPENDIX B

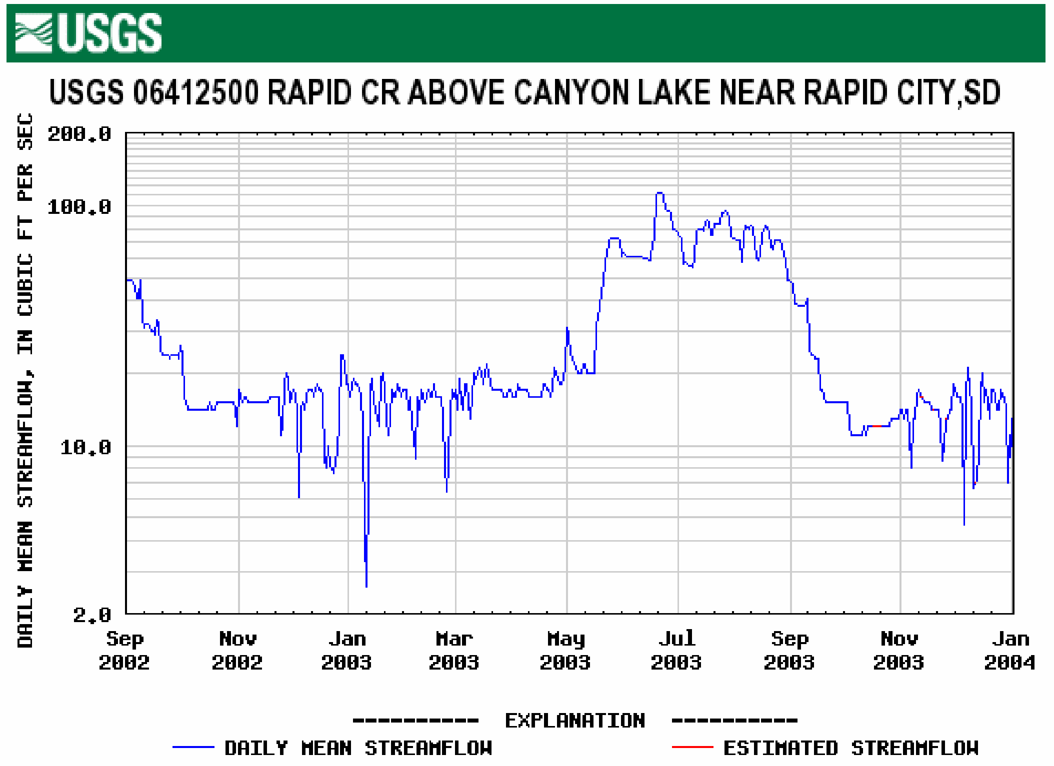
STREAMFLOW DATA

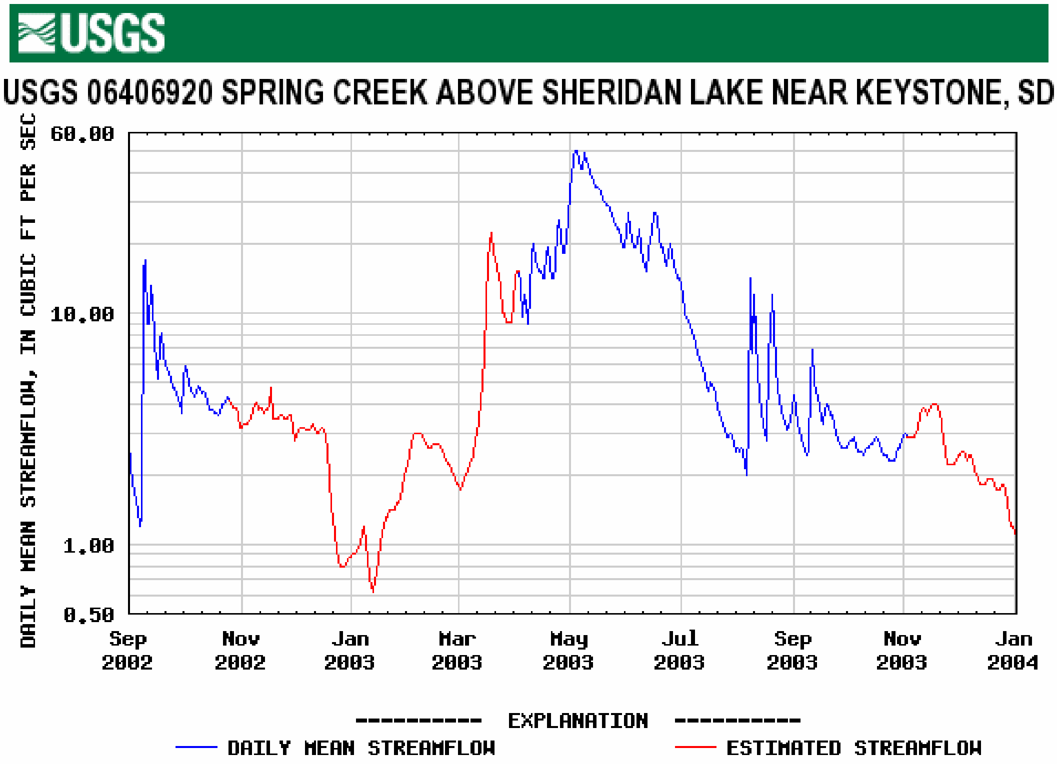
Information presented in this appendix was obtained from the following web site.

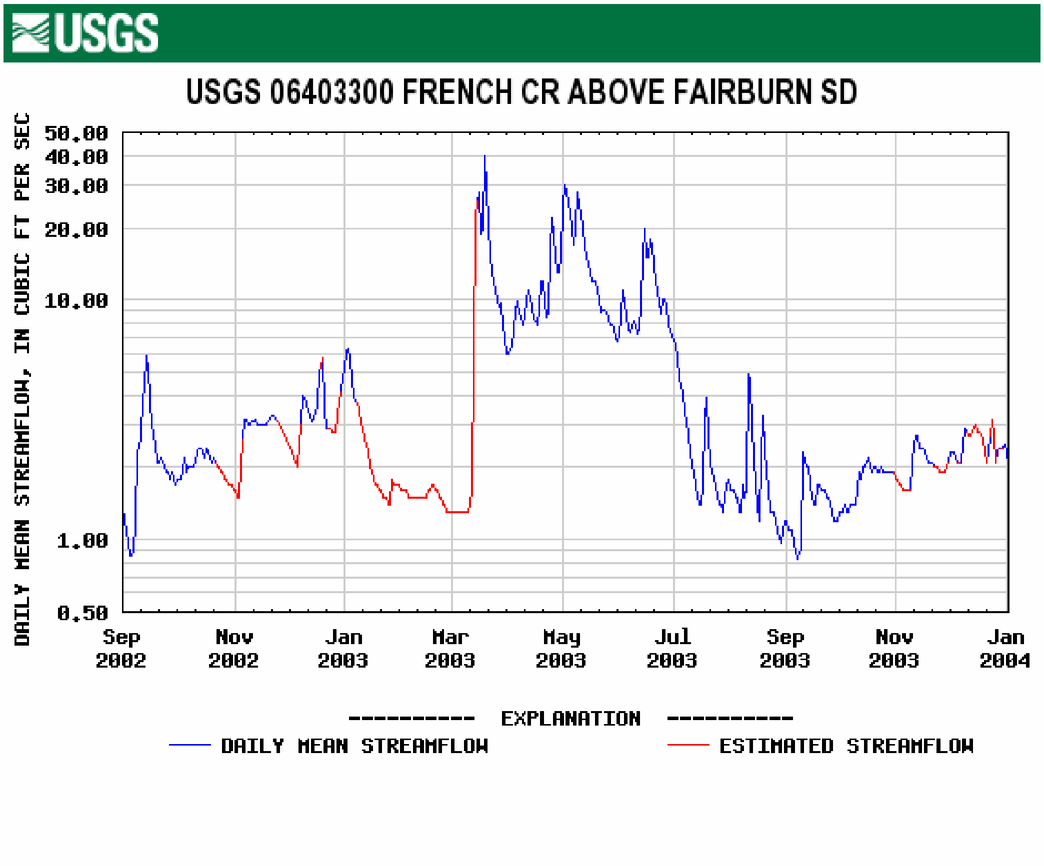
http://waterdata.usgs.gov/nwis


USGS
USGS 06437020 BEAR BUTTE CREEK NEAR DEADWOOD, SD
DAILY MEAN STREAMFLOW, IN CUBIC FT PER SEC
40.00
30.00
20.00
10.00
1.00
0.10
Sep 2002
Nov 2002
Jan 2003
Mar 2003
May 2003
Jul 2003
Sep 2003
Nov 2003
Jan 2004
---------- EXPLANATION ----------
DAILY MEAN STREAMFLOW
ESTIMATED STREAMFLOW


USGS
USGS 06422500 BOXELDER CR NEAR NEMO,SD
DAILY MEAN STREAMFLOW, IN CUBIC FT PER SEC
50.0
40.0
30.0
20.0
10.0
1.0
Sep 2002
Nov 2002
Jan 2003
Mar 2003
May 2003
Jul 2003
Sep 2003
Nov 2003
Jan 2004
---------- EXPLANATION ----------
DAILY MEAN STREAMFLOW
ESTIMATED STREAMFLOW


USGS
USGS 06412500 RAPID CR ABOVE CANYON LAKE NEAR RAPID CITY,SD
DAILY MEAN STREAMFLOW, IN CUBIC FT PER SEC
200.0
100.0
10.0
2.0
Sep 2002
Nov 2002
Jan 2003
Mar 2003
May 2003
Jul 2003
Sep 2003
Nov 2003
Jan 2004
---------- EXPLANATION ----------
DAILY MEAN STREAMFLOW
ESTIMATED STREAMFLOW


USGS
USGS 06406920 SPRING CREEK ABOVE SHERIDAN LAKE NEAR KEYSTONE, SD
DAILY MEAN STREAMFLOW, IN CUBIC FT PER SEC
60.00
10.00
1.00
0.50
Sep 2002
Nov 2002
Jan 2003
Mar 2003
May 2003
Jul 2003
Sep 2003
Nov 2003
Jan 2004
---------- EXPLANATION ----------
DAILY MEAN STREAMFLOW
ESTIMATED STREAMFLOW


USGS
USGS 06403300 FRENCH CR ABOVE FAIRBURN SD
DAILY MEAN STREAMFLOW, IN CUBIC FT PER SEC
50.00
40.00
30.00
20.00
10.00
1.00
0.50
Sep 2002
Nov 2002
Jan 2003
Mar 2003
May 2003
Jul 2003
Sep 2003
Nov 2003
Jan 2004
---------- EXPLANATION ----------
DAILY MEAN STREAMFLOW
ESTIMATED STREAMFLOW

APPENDIX C

MONITORING WELL LITHOLOGIC LOGS

Monitoring Well R20-2002-02

Location Information

Legal Location:	NE¼ SW¼ SW¼ NW¼ SEC. 29, T. 1 S., R. 5 E.		
County:	PENNINGTON	Location:	001S05E29BCCA
Basin:	SPRING CREEK	Latitude:	43 56' 07"
Hydrologic Unit Code:	10120109	Longitude:	103 33' 25"
Land Owner:	CITY OF HILL CITY	Ground Surface Elev. (ft.):	4937 T

Project Information

Project:	SEPTIC SYSTEM DRAIN FIELD		
Drill Date:	07/11/2002	Geologist:	J. SAWYER/J. LESTER
Company:	SDGS	Geologist's Log:	X
Drilling Method:	AUGER	Driller:	D. IVERSON
Test Hole Number:	R20-2002-02	Driller's Log:	
Samples:		Total Drill Hole Depth (ft.):	17.0

Well Information

SDGS Well Name:	R20-2002-02	Aquifer:	ALLUVIUM
Water Rights Well:		Management Unit:	
Other Well Name:		Casing Top Elev. (ft.):	4939.08 T
Casing Type:	PVC	Casing Diameter (in.):	2.0
Screen Type:	PVC	Screen Length (ft.):	10.4
Total Casing and Screen (ft.):	18.8	Casing Stick-up (ft.):	2.08

Notes

NATURAL FILTER PACK FROM 17.0 TO 10.0 FEET BELOW LAND SURFACE, COARSE SAND FROM 10.0 TO 4.2 FEET BELOW LAND SURFACE, BENTONITE GROUT FROM 4.2 TO 3.0 FEET BELOW LAND SURFACE, CEMENT GROUT FROM 3.0 FEET BELOW LAND SURFACE TO LAND SURFACE, STEEL WELL HEAD PROTECTOR INSTALLED.

Lithologic Information

Elevation (ft.)	Depth (ft.)	Description
4937.0 - 4934.0	0.0 - 3.0	SILT, BROWN; MICACEOUS; ORGANIC-RICH; WITH VERY FINE, ANGULAR TO ROUNDED, WELL-SORTED QUARTZ SAND; GARNETS; DRY
4934.0 - 4932.0	3.0 - 5.0	SILT, DARK-BROWN; MICACEOUS; ORGANICS; ABUNDANT VERY FINE-GRAINED, ANGULAR TO ROUNDED QUARTZ SAND; GARNETS; PRECAMBRIAN SCHIST COBBLES AND BOULDERS AT 5 FEET
4932.0 - 4930.0	5.0 - 7.0	SILT, DARK-BROWN; MICACEOUS; ORGANICS; ABUNDANT ANGULAR QUARTZ FRAGMENTS; MINOR TOURMALINE FRAGMENTS; MINOR PRECAMBRIAN COBBLES AND BOULDERS AT 7 FEET
4930.0 - 4927.0	7.0 - 10.0	CLAY AND SILT, BLACK TO BROWN; MICACEOUS; ROUNDED TO ANGULAR QUARTZ GRAINS; GARNETS; PRECAMBRIAN COBBLES OR BOULDERS AT 10 FEET
4927.0 - 4925.0	10.0 - 12.0	CLAY AND SILT, BROWNISH-BLACK; MICACEOUS; WELL-ROUNDED QUARTZ SAND; WATER AT 11 FEET
4925.0 - 4922.0	12.0 - 15.0	CLAY AND SILT, BROWNISH-BLACK; MICACEOUS; FINE-GRAINED SAND; ABUNDANT COBBLES OR BOULDERS AT 13 FEET; WET
4922.0 - 4920.0	15.0 - 17.0	SILT AND CLAY, DARK-BROWN TO BLACK; MICACEOUS; MINOR ANGULAR, IRON-STAINED QUARTZ GRAINS; PRECAMBRIAN ROCK FRAGMENTS; WET

Monitoring Well R20-2002-03

Location Information

Legal Location:	SE¼ NW¼ NW¼ SW¼ SEC. 29, T. 1 S., R. 5 E.		
County:	PENNINGTON	Location:	001S05E29CBBD
Basin:	SPRING CREEK	Latitude:	43 55' 56"
Hydrologic Unit Code:	10120109	Longitude:	103 33' 23"
Land Owner:	CITY OF HILL CITY	Ground Surface Elev. (ft.):	4945 T

Project Information

Project:	SEPTIC SYSTEM DRAIN FIELD		
Drill Date:	07/11/2002	Geologist:	J. SAWYER/J. LESTER
Company:	SDGS	Geologist's Log:	X
Drilling Method:	AUGER	Driller:	D. IVERSON
Test Hole Number:	R20-2002-03	Driller's Log:	
Samples:		Total Drill Hole Depth (ft.):	16.0

Well Information

SDGS Well Name:	R20-2002-03	Aquifer:	ALLUVIUM
Water Rights Well:		Management Unit:	
Other Well Name:		Casing Top Elev. (ft.):	4947.50 T
Casing Type:	PVC	Casing Diameter (in.):	2.0
Screen Type:	PVC	Screen Length (ft.):	10.4
Total Casing and Screen (ft.):	17.8	Casing Stick-up (ft.):	2.50

Notes

NATURAL FILTER PACK AND COARSE SAND FROM 16.0 TO 2.5 FEET BELOW LAND SURFACE, BENTONITE GROUT FROM 2.6 TO 2.0 FEET BELOW LAND SURFACE, CEMENT GROUT FROM 2.0 FEET BELOW LAND SURFACE TO LAND SURFACE, STEEL WELL HEAD PROTECTOR INSTALLED.

Lithologic Information

Elevation (ft.)	Depth (ft.)	Description
4945.0 - 4942.0	0.0 - 3.0	TOPSOIL; SILT, DARK-BROWN TO BLACK, QUARTZOSE, VERY MICACEOUS; MINOR CLAY; ABUNDANT GARNETS; DAMP
4942.0 - 4940.0	3.0 - 5.0	SILT AND CLAY, DARK-BROWN TO BLACK; VERY FINE QUARTZ SAND; MICA; GARNETS; DAMP
4940.0 - 4938.0	5.0 - 7.0	SILT AND CLAY, DARK-BROWN; MICACEOUS; ORGANICS; ROUNDED TO ANGULAR QUARTZ SAND; ROUNDED GRANITE CLASTS; GARNETS; WET AT 7 FEET
4938.0 - 4935.0	7.0 - 10.0	CLAY, BROWN TO BLACK; MICACEOUS; MINOR ROUNDED TO ANGULAR QUARTZ SAND; GARNETS; WET; ODOR OF DECAY
4935.0 - 4934.0	10.0 - 11.0	CLAY AND SAND; MICACEOUS; GARNETS; TOURMALINE FRAGMENTS; MINOR ORGANICS; FELDSPAR GRAINS; WET
4934.0 - 4930.0	11.0 - 15.0	CLAY, LIGHT-BROWN TO GRAY; MICACEOUS; FINE- TO VERY FINE-GRAINED QUARTZ SAND; VERY WET
4930.0 - 4929.0	15.0 - 16.0	CLAY, SILT AND GRAVEL, DARK-BROWN TO GRAY; ROUNDED QUARTZ SAND; MINOR ORGANICS; PRECAMBRIAN ROCK FRAGMENTS

Monitoring Well R20-2002-04

Location Information

Legal Location:	SW¼ NW¼ NW¼ SW¼ SEC. 29, T. 1 S., R. 5 E.		
County:	PENNINGTON	Location:	001S05E29CBBC
Basin:	SPRING CREEK	Latitude:	43 55' 55"
Hydrologic Unit Code:	10120109	Longitude:	103 33' 30"
Land Owner:	CITY OF HILL CITY	Ground Surface Elev. (ft.):	4950 T

Project Information

Project:	SEPTIC SYSTEM DRAIN FIELD		
Drill Date:	07/12/2002	Geologist:	J. SAWYER/J. LESTER
Company:	SDGS	Geologist's Log:	X
Drilling Method:	AUGER	Driller:	D. IVERSON
Test Hole Number:	R20-2002-04	Driller's Log:	
Samples:		Total Drill Hole Depth (ft.):	16.5

Well Information

SDGS Well Name:	R20-2002-04	Aquifer:	ALLUVIUM
Water Rights Well:		Management Unit:	
Other Well Name:		Casing Top Elev. (ft.):	4953.00 T
Casing Type:	PVC	Casing Diameter (in.):	2.0
Screen Type:	PVC	Screen Length (ft.):	10.4
Total Casing and Screen (ft.):	19.5	Casing Stick-up (ft.):	3.00

Notes

NATURAL FILTER PACK FROM 16.5 TO 5.4 FEET BELOW LAND SURFACE, COARSE SAND FROM 5.4 TO 2.1 FEET BELOW LAND SURFACE, BENTONITE GROUT FROM 2.1 TO 1.5 FEET BELOW LAND SURFACE, CEMENT GROUT FROM 1.5 FEET BELOW LAND SURFACE TO LAND SURFACE, STEEL WELL HEAD PROTECTOR INSTALLED.

Lithologic Information

Elevation (ft.)	Depth (ft.)	Description
4950.0 - 4949.0	0.0 - 1.0	GRAVEL, DARK-BROWN; MICACEOUS; ROUNDED QUARTZ SAND; BLACK CLAY; SILT; ORGANICS; DAMP
4949.0 - 4946.0	1.0 - 4.0	SILT, DARK-BROWN TO BLACK; MICACEOUS; ORGANIC-RICH; BLACK CLAY; PRECAMBRIAN GRAVEL CLASTS; ANGULAR TO ROUNDED QUARTZ SAND; GARNETS; WET AT 4 FEET
4946.0 - 4944.0	4.0 - 6.0	CLAY, BLACK; MICACEOUS; ORGANICS; ANGULAR TO SUBROUNDED QUARTZ; IRON-STAINED PRECAMBRIAN CLASTS; MINOR GARNETS; TOURMALINE FRAGMENTS; DAMP
4944.0 - 4943.0	6.0 - 7.0	SILT, BLACK; GRAVEL; SAND; WET
4943.0 - 4939.0	7.0 - 11.0	CLAY, BLACK; MICACEOUS; ORGANICS; DARK-BROWN SILT; FINE-GRAINED, ROUNDED QUARTZ SAND; MINOR GARNETS; GRAVEL AT 11 FEET; WET
4939.0 - 4937.0	11.0 - 13.0	SILT; GRAVEL; HARD DRILLING AT 13 FEET
4937.0 - 4936.0	13.0 - 14.0	CLAY, BLACK TO TAN; MICACEOUS; GRAY-BROWN, COARSE, SUBROUNDED SAND; PRECAMBRIAN ROCK FRAGMENTS, MINOR GARNETS
4936.0 - 4933.5	14.0 - 16.5	CLAY AND SILT, BLACK AND TAN; MICACEOUS; MINOR ORGANICS; DARK-BROWN TO BLACK, FINE-GRAINED, ANGULAR TO ROUNDED SAND; GARNETS; PRECAMBRIAN CLASTS; HARD DRILLING AT 16 FEET; VERY WET

Monitoring Well R20-2002-05

Location Information

Legal Location:	NE¼ SW¼ NE¼ SE¼ SEC. 30, T. 1 S., R. 5 E.		
County:	PENNINGTON	Location:	001S05E30DACA
Basin:	SPRING CREEK	Latitude:	43 55’ 55’’
Hydrologic Unit Code:	10120109	Longitude:	103 33’ 40’’
Land Owner:	CITY OF HILL CITY	Ground Surface Elev. (ft.):	4952 T

Project Information

Project:	SEPTIC SYSTEM DRAIN FIELD		
Drill Date:	07/12/2002	Geologist:	J. SAWYER/J. LESTER
Company:	SDGS	Geologist’s Log:	X
Drilling Method:	AUGER	Driller:	D. IVERSON
Test Hole Number:	R20-2002-05	Driller’s Log:	
Samples:		Total Drill Hole Depth (ft.):	20.0

Well Information

SDGS Well Name:	R20-2002-05	Aquifer:	ALLUVIUM
Water Rights Well:		Management Unit:	
Other Well Name:		Casing Top Elev. (ft.):	4952.00 T
Casing Type:	PVC	Casing Diameter (in.):	2.0
Screen Type:	PVC	Screen Length (ft.):	10.4
Total Casing and Screen (ft.):	17.7	Casing Stick-up (ft.):	0

Notes

NATURAL FILTER PACK FROM 20.0 TO 6.0 FEET BELOW LAND SURFACE, COARSE SAND FROM 6.0 TO 2.0 FEET BELOW LAND SURFACE, BENTONITE GROUT FROM 2.0 FEET TO 1.0 FOOT BELOW LAND SURFACE, CEMENT GROUT FROM 1.0 FOOT BELOW LAND SURFACE TO LAND SURFACE, FLUSH WELL PROTECTOR, CASING TOP 3.0 INCHES BELOW LAND SURFACE.

Lithologic Information

Elevation (ft.)	Depth (ft.)	Description
4952.0 - 4949.0	0.0 - 3.0	TOPSOIL; VERY MICACEOUS; ORGANICS; GRAY TO BROWN SILT; ANGULAR TO WELL-ROUNDED QUARTZ SAND; PRECAMBRIAN CLASTS UP TO 1 INCH; ABUNDANT GARNETS; DRY
4949.0 - 4946.0	3.0 - 6.0	GRAVEL AND SILT, BROWN; MICACEOUS; COARSE TO FINE, ANGULAR TO ROUNDED QUARTZ SAND; ABUNDANT GARNETS; DRY
4946.0 - 4940.0	6.0 - 12.0	SILT, BROWN; BROWN, COARSE, GRAVEL; GRAVEL LAYER AT 8 FEET; WATER AT 10 FEET
4940.0 - 4937.0	12.0 - 15.0	SILT; GRAVEL; FINE-GRAINED SAND
4937.0 - 4935.0	15.0 - 17.0	SILT, BROWN TO LIGHT-GRAY; VERY FINE-GRAINED SAND; MINOR GRAVEL; COBBLES
4935.0 - 4932.0	17.0 - 20.0	SAND, SUBROUNDED; MICACEOUS; LIGHT-GRAY TO BROWN CLAY AND SILT; DAMP

Monitoring Well R20-2002-10

Location Information

Legal Location:	NE¼ NE¼ NW¼ NE¼ SEC. 36, T. 1 S., R. 4 E.		
County:	PENNINGTON	Location:	001S04E36ABAA 1
Basin:	SPRING CREEK	Latitude:	43 55’ 33’’
Hydrologic Unit Code:	10120109	Longitude:	103 34’ 58’’
Land Owner:	PRIVATE RESIDENCE	Ground Surface Elev. (ft.):	5040 T

Project Information

Project:	SEPTIC SYSTEM DRAIN FIELD		
Drill Date:	07/14/2002	Geologist:	J. SAWYER/J. LESTER
Company:	SDGS	Geologist’s Log:	X
Drilling Method:	AUGER	Driller:	D. IVERSON
Test Hole Number:	R20-2002-10	Driller’s Log:	
Samples:		Total Drill Hole Depth (ft.):	27.0

Well Information

SDGS Well Name:	R20-2002-10	Aquifer:	ALLUVIUM
Water Rights Well:		Management Unit:	
Other Well Name:		Casing Top Elev. (ft.):	5040.00 T
Casing Type:	PVC	Casing Diameter (in.):	2.0
Screen Type:	PVC	Screen Length (ft.):	10.4
Total Casing and Screen (ft.):	26.9	Casing Stick-up (ft.):	0

Notes

NATURAL AND COARSE SAND FILTER PACK FROM 26.9 FEET TO 8.0 FEET BELOW LAND SURFACE, BENTONITE GROUT FROM 8.0 FEET TO 1.0 FOOT BELOW LAND SURFACE, CEMENT GROUT FROM 1.0 FOOT BELOW LAND SURFACE TO LAND SURFACE, FLUSH WELL PROTECTOR, CASING TOP 2.0 INCHES BELOW LAND SURFACE.

Lithologic Information

Elevation (ft.)	Depth (ft.)	Description
5040.0 - 5038.0	0.0 - 2.0	SILT AND CLAY, DARK-BROWN TO BLACK; MICACEOUS; ORGANICS; PRECAMBRIAN SCHIST FRAGMENTS; DRY
5038.0 - 5033.0	2.0 - 7.0	SILT AND CLAY, BLACK TO DARK-BROWN; VERY MICACEOUS; MINOR ORGANICS; PRECAMBRIAN SCHIST FRAGMENTS; DAMP AT 7 FEET
5033.0 - 5029.0	7.0 - 11.0	SAND, DARK-BROWN; MICACEOUS; PRECAMBRIAN SCHIST FRAGMENTS; WET AT 11 FEET
5029.0 - 5026.0	11.0 - 14.0	SILT, DARK-BROWN; MICACEOUS; MINOR CLAY; PRECAMBRIAN SCHIST FRAGMENTS
5026.0 - 5021.0	14.0 - 19.0	SILT, BROWN TO LIGHT-GRAY; MICACEOUS; MINOR CLAY; COBBLE-SIZED PRECAMBRIAN SCHIST FRAGMENTS
5021.0 - 5014.0	19.0 - 26.0	CLAY, GRAY-BROWN; MICACEOUS; MINOR SILT; PRECAMBRIAN SCHIST FRAGMENTS
5014.0 - 5013.0	26.0 - 27.0	SCHIST; GRAY TO BLACK SILT AND CLAY; VERY WET (PRECAMBRIAN BEDROCK)

Monitoring Well R20-2002-11

Location Information

Legal Location:	NE¼ NE¼ NW¼ NE¼ SEC. 36, T. 1 S., R. 4 E.		
County:	PENNINGTON	Location:	001S04E36ABAA 2
Basin:	SPRING CREEK	Latitude:	43 55’ 32’’
Hydrologic Unit Code:	10120109	Longitude:	103 34’ 57’’
Land Owner:	PRIVATE RESIDENCE	Ground Surface Elev. (ft.):	5035 T

Project Information

Project:	SEPTIC SYSTEM DRAIN FIELD		
Drill Date:	07/14/2002	Geologist:	J. SAWYER/J. LESTER
Company:	SDGS	Geologist’s Log:	X
Drilling Method:	AUGER	Driller:	D. IVERSON
Test Hole Number:	R20-2002-11	Driller’s Log:	
Samples:		Total Drill Hole Depth (ft.):	28.5

Well Information

SDGS Well Name:	R20-2002-11	Aquifer:	ALLUVIUM
Water Rights Well:		Management Unit:	
Other Well Name:		Casing Top Elev. (ft.):	5035.00 T
Casing Type:	PVC	Casing Diameter (in.):	2.0
Screen Type:	PVC	Screen Length (ft.):	10.3
Total Casing and Screen (ft.):	28.0	Casing Stick-up (ft.):	0

Notes

NATURAL FILTER PACK FROM 28.5 TO 26.0 FEET BELOW LAND SURFACE, COARSE SAND FROM 26.0 TO 13.8 FEET BELOW LAND SURFACE, BENTONITE GROUT FROM 13.8 FEET TO 1.0 FOOT BELOW LAND SURFACE, CEMENT GROUT FROM 1.0 FOOT BELOW LAND SURFACE TO LAND SURFACE, FLUSH WELL PROTECTOR, CASING TOP 3.0 INCHES BELOW LAND SURFACE.

Lithologic Information

Elevation (ft.)	Depth (ft.)	Description
5035.0 - 5033.0	0.0 - 2.0	TOPSOIL, BLACK TO DARK-BROWN; MICACEOUS; ORGANICS
5033.0 - 5030.0	2.0 - 5.0	SILT AND CLAY, BLACK TO DARK-BROWN; MICACEOUS; IRON-STAINED PRECAMBRIAN SCHIST FRAGMENTS; QUARTZ FRAGMENTS
5030.0 - 5028.0	5.0 - 7.0	SILT AND CLAY, DARK-BROWN
5028.0 - 5026.0	7.0 - 9.0	CLAY AND SILT, BROWN; MINOR GRAVEL
5026.0 - 5024.0	9.0 - 11.0	SILT, GRAY-BROWN; MINOR CLAY; VERY MICACEOUS; PRECAMBRIAN ROCK FRAGMENTS; DRY
5024.0 - 5023.0	11.0 - 12.0	SILT, LIGHT-GRAY TO LIGHT-BROWN; MINOR CLAY; GRAVEL; PRECAMBRIAN SCHIST FRAGMENTS
5023.0 - 5017.0	12.0 - 18.0	SILT, LIGHT-GRAY TO LIGHT-BROWN; GRAVEL; MINOR CLAY; PRECAMBRIAN SCHIST FRAGMENTS UP TO 1 INCH
5017.0 - 5016.0	18.0 - 19.0	CLAY, DARK-BROWN TO LIGHT-BROWN; GRAVEL; PRECAMBRIAN SCHIST FRAGMENTS
5016.0 - 5006.5	19.0 - 28.5	SILT, LIGHT-GRAY TO LIGHT-BROWN; GRAVEL; MINOR CLAY; PRECAMBRIAN SCHIST FRAGMENTS

Monitoring Well R20-2002-12

Location Information

Legal Location:	SE¼ NW¼ NE¼ SE¼ SEC. 27, T. 3 S., R. 4 E.		
County:	CUSTER	Location:	003S04E27DABD
Basin:	FRENCH CREEK	Latitude:	43 45’ 32’’
Hydrologic Unit Code:	10120109	Longitude:	103 37’ 17’’
Land Owner:	CITY OF CUSTER	Ground Surface Elev. (ft.):	5350 T

Project Information

Project:	SEPTIC SYSTEM DRAIN FIELD		
Drill Date:	07/15/2002	Geologist:	J. SAWYER/J. LESTER
Company:	SDGS	Geologist’s Log:	X
Drilling Method:	AUGER	Driller:	D. IVERSON
Test Hole Number:	R20-2002-12	Driller’s Log:	
Samples:		Total Drill Hole Depth (ft.):	10.0

Well Information

SDGS Well Name:	R20-2002-12	Aquifer:	ALLUVIUM
Water Rights Well:		Management Unit:	
Other Well Name:		Casing Top Elev. (ft.):	5350.00 T
Casing Type:	PVC	Casing Diameter (in.):	2.0
Screen Type:	PVC	Screen Length (ft.):	6.2
Total Casing and Screen (ft.):	9.6	Casing Stick-up (ft.):	0

Notes

NATURAL FILTER PACK FROM 10.0 TO 4.0 FEET BELOW LAND SURFACE, COARSE SAND FROM 4.0 TO 1.9 FEET BELOW LAND SURFACE, BENTONITE GROUT FROM 1.9 FEET TO 1.0 FOOT BELOW LAND SURFACE, FLUSH WELL PROTECTOR, CASING TOP 3.0 INCHES BELOW LAND SURFACE.

Lithologic Information

Elevation (ft.)	Depth (ft.)	Description
5350.0 - 5348.0	0.0 - 2.0	CLAY AND SILT, DARK-BLACK TO BROWN; MICACEOUS; MEDIUM-GRAINED, SUBANGULAR SAND; BLACK ORGANICS
5348.0 - 5347.0	2.0 - 3.0	CLAY AND SILT, BLACK; ORGANICS
5347.0 - 5345.0	3.0 - 5.0	CLAY, SILT, BROWN TO TAN; MICACEOUS; MINOR FINE-GRAINED, SUBANGULAR QUARTZ SAND; WET AT 4 FEET
5345.0 - 5343.0	5.0 - 7.0	SILT, GOLD-BROWN; ANGULAR TO ROUNDED, FINE-GRAINED, WELL-SORTED SAND; MINOR BLACK CLAY STREAKS; COBBLES OR BOULDERS AT 6 TO 7 FEET; VERY WET
5343.0 - 5340.0	7.0 - 10.0	SILT, GRAY-BROWN TO TAN; WELL-SORTED, ROUNDED QUARTZ SAND; LARGE, ANGULAR PRECAMBRIAN ROCK FRAGMENTS; MINOR CLAY; COBBLES OR BOULDERS AT 10 FEET

Monitoring Well R20-2002-14

Location Information

Legal Location:	SE¼ NW¼ NE¼ SE¼ SEC. 27, T. 3 S., R. 4 E.		
County:	CUSTER	Location:	003S04E27DABD 2
Basin:	FRENCH CREEK	Latitude:	43 45’ 32’’
Hydrologic Unit Code:	10120109	Longitude:	103 37’ 17’’
Land Owner:	CITY OF CUSTER	Ground Surface Elev. (ft.):	5350 T

Project Information

Project:	SEPTIC SYSTEM DRAIN FIELD		
Drill Date:	07/15/2002	Geologist:	J. SAWYER/J. LESTER
Company:	SDGS	Geologist’s Log:	X
Drilling Method:	AUGER	Driller:	D. IVERSON
Test Hole Number:	R20-2002-14	Driller’s Log:	
Samples:		Total Drill Hole Depth (ft.):	14.0

Well Information

SDGS Well Name:	R20-2002-14	Aquifer:	ALLUVIUM
Water Rights Well:		Management Unit:	
Other Well Name:		Casing Top Elev. (ft.):	5350.00 T
Casing Type:	PVC	Casing Diameter (in.):	2.0
Screen Type:	PVC	Screen Length (ft.):	10.3
Total Casing and Screen (ft.):	13.5	Casing Stick-up (ft.):	0

Notes

NATURAL FILTER PACK FROM 14.0 TO 6.0 FEET BELOW LAND SURFACE, COARSE SAND FROM 6.0 TO 2.0 FEET BELOW LAND SURFACE, BENTONITE GROUT FROM 2.0 FEET TO 1.0 FOOT BELOW LAND SURFACE, FLUSH WELL PROTECTOR, CASING TOP 3.0 INCHES BELOW LAND SURFACE.

Lithologic Information

Elevation (ft.)	Depth (ft.)	Description
5350.0 - 5348.0	0.0 - 2.0	CLAY AND SILT, DARK-BROWN TO BLACK; MICACEOUS
5348.0 - 5346.0	2.0 - 4.0	CLAY AND SILT, BLACK
5346.0 - 5344.0	4.0 - 6.0	SILT, GOLDEN-BROWN; MICACEOUS; WELL-SORTED, MODERATELY ROUNDED QUARTZ SAND; MINOR PRECAMBRIAN CLASTS AND GARNETS; COBBLES OR BOULDERS AT 5 FEET
5344.0 - 5342.0	6.0 - 8.0	SAND AND CLAY, BROWN TO TAN, POORLY SORTED; MICACEOUS; LARGE PRECAMBRIAN CLASTS; MINOR ORGANICS
5342.0 - 5338.0	8.0 - 12.0	SAND, TAN, COARSE-GRAINED, SUBANGULAR; MICACEOUS CLAY; MINOR GARNETS; WET
5338.0 - 5336.0	12.0 - 14.0	SAND, TAN TO BROWN, POORLY SORTED, SUBANGULAR; MICACEOUS; MINOR PRECAMBRIAN CLASTS; MINOR BLACK STREAKS OF CLAY; GARNETS; VERY WET

Monitoring Well R20-2002-15

Location Information

Legal Location:	SE¼ NW¼ NE¼ SE¼ SEC. 27, T. 3 S., R. 4 E.		
County:	CUSTER	Location:	003S04E27DABD 3
Basin:	FRENCH CREEK	Latitude:	43 45’ 31’’
Hydrologic Unit Code:	10120109	Longitude:	103 37’ 15’’
Land Owner:	CITY OF CUSTER	Ground Surface Elev. (ft.):	5360 T

Project Information

Project:	SEPTIC SYSTEM DRAIN FIELD		
Drill Date:	07/15/2002	Geologist:	J. SAWYER/J. LESTER
Company:	SDGS	Geologist’s Log:	X
Drilling Method:	AUGER	Driller:	D. IVERSON
Test Hole Number:	R20-2002-15	Driller’s Log:	
Samples:		Total Drill Hole Depth (ft.):	22.5

Well Information

SDGS Well Name:	R20-2002-15	Aquifer:	ALLUVIUM
Water Rights Well:		Management Unit:	
Other Well Name:		Casing Top Elev. (ft.):	5360.00 T
Casing Type:	PVC	Casing Diameter (in.):	2.0
Screen Type:	PVC	Screen Length (ft.):	10.4
Total Casing and Screen (ft.):	22.0	Casing Stick-up (ft.):	0

Notes

NATURAL FILTER PACK FROM 22.5 TO 14.5 FEET BELOW LAND SURFACE, COARSE SAND FROM 14.5 TO 10.0 FEET BELOW LAND SURFACE, BENTONITE GROUT FROM 10.0 FEET BELOW LAND SURFACE TO LAND SURFACE, FLUSH WELL PROTECTOR, CASING TOP 3.0 INCHES BELOW LAND SURFACE.

Lithologic Information

Elevation (ft.)	Depth (ft.)	Description
5360.0 - 5358.0	0.0 - 2.0	SILT AND CLAY, DARK-BROWN; MICACEOUS; ORGANICS; MEDIUM- TO FINE-GRAINED, ANGULAR TO ROUNDED SAND; MINOR GARNETS; DRY
5358.0 - 5356.0	2.0 - 4.0	CLAY AND SILT, VERY DARK-BROWN; MICACEOUS; ORGANICS; ANGULAR QUARTZ FRAGMENTS; MINOR IRON STAINING
5356.0 - 5354.0	4.0 - 6.0	GRAVEL, BROWN; PRECAMBRIAN GRANITE FRAGMENTS; ORGANICS; MICA; ANGULAR TO SUBANGULAR QUARTZ; MINOR BROWN SILT AND CLAY; DRY
5354.0 - 5352.0	6.0 - 8.0	GRAVEL; PRECAMBRIAN GRANITE FRAGMENTS; MINOR TAN TO LIGHT-GRAY CLAY AND SILT; DRY
5352.0 - 5349.0	8.0 - 11.0	SAND, GRAY, WELL-SORTED, ANGULAR TO ROUNDED; GOLD-BROWN MICA
5349.0 - 5345.0	11.0 - 15.0	SAND, GRAY-BROWN, WELL-SORTED, ANGULAR TO ROUNDED; MICACEOUS; MINOR CLAY; WET AT 13 FEET
5345.0 - 5337.5	15.0 - 22.5	SAND, GRAY, VERY FINE-GRAINED; CLAY; MICA; VERY WET; HARD DRILLING FROM 16.0 TO 22.5 FEET

Monitoring Well R20-2002-17

Location Information

Legal Location:	SW¼ SE¼ SW¼ NW¼ SEC. 29, T. 1 S., R. 5 E.		
County:	PENNINGTON	Location:	001S05E29BCDC
Basin:	SPRING CREEK	Latitude:	43 56’ 01’’
Hydrologic Unit Code:	10120109	Longitude:	103 33’ 22’’
Land Owner:	CITY OF HILL CITY	Ground Surface Elev. (ft.):	4947 T

Project Information

Project:	SEPTIC SYSTEM DRAIN FIELD		
Drill Date:	07/24/2002	Geologist:	J. SAWYER/J. LESTER
Company:	SDGS	Geologist’s Log:	X
Drilling Method:	AUGER	Driller:	D. IVERSON
Test Hole Number:	R20-2002-17	Driller’s Log:	
Samples:		Total Drill Hole Depth (ft.):	17.0

Well Information

SDGS Well Name:	R20-2002-17	Aquifer:	ALLUVIUM
Water Rights Well:		Management Unit:	
Other Well Name:		Casing Top Elev. (ft.):	4949.67 T
Casing Type:	PVC	Casing Diameter (in.):	2.0
Screen Type:	PVC	Screen Length (ft.):	10.3
Total Casing and Screen (ft.):	19.0	Casing Stick-up (ft.):	2.67

Notes

NATURAL FILTER PACK FROM 17.0 TO 8.6 FEET BELOW LAND SURFACE, COARSE SAND FROM 8.6 TO 4.8 FEET BELOW LAND SURFACE, BENTONITE GROUT FROM 4.8 TO 2.0 FEET BELOW LAND SURFACE, CEMENT GROUT FROM 2.0 FEET BELOW LAND SURFACE TO LAND SURFACE, STEEL WELL PROTECTOR.

Lithologic Information

Elevation (ft.)	Depth (ft.)	Description
4947.0 - 4944.0	0.0 - 3.0	TOPSOIL, DARK-BROWN; MICACEOUS; ORGANIC-RICH; BLACK CLAY; VERY FINE-GRAINED, SUBROUNDED QUARTZ SAND
4944.0 - 4941.0	3.0 - 6.0	CLAY, BROWN TO TAN; MICA; FINE-GRAINED QUARTZ SAND; WET AT 5 FEET
4941.0 - 4937.0	6.0 - 10.0	SAND, COARSE-GRAINED, SUBANGULAR; QUARTZ-RICH; MICACEOUS; VERY FINE-GRAINED, SUBROUNDED QUARTZ; BROWN CLAY; GARNETS; GRAVEL AT 7 FEET
4937.0 - 4935.0	10.0 - 12.0	CLAY, BROWN; MICACEOUS; POORLY SORTED, SUBANGULAR TO VERY WELL-ROUNDED QUARTZ SAND; PRECAMBRIAN CLASTS
4935.0 - 4932.0	12.0 - 15.0	SAND, MEDIUM-GRAINED, ROUNDED; PRECAMBRIAN CLASTS; DARK-BROWN, STREAKY CLAY; MINOR REDDISH-GOLDEN-BROWN MICA; GRAVEL AT 14 FEET
4932.0 - 4930.0	15.0 - 17.0	CLAY, TAN-BROWN; MICA; MEDIUM-GRAINED, SUBROUNDED QUARTZ SAND; PRECAMBRIAN CLASTS; GARNETS; IRON-STAINED CLAY STREAKS

Monitoring Well R20-2002-18

Location Information

Legal Location:	NE¼ NE¼ NW¼ NE¼ SEC. 36, T. 1 S., R. 4 E.		
County:	PENNINGTON	Location:	001S04E36ABAA 3
Basin:	SPRING CREEK	Latitude:	43 55’ 32’’
Hydrologic Unit Code:	10120109	Longitude:	103 34’ 57’’
Land Owner:	PRIVATE RESIDENCE	Ground Surface Elev. (ft.):	5036 T

Project Information

Project:	SEPTIC SYSTEM DRAIN FIELD		
Drill Date:	07/24/2002	Geologist:	J. SAWYER/J. LESTER
Company:	SDGS	Geologist’s Log:	X
Drilling Method:	AUGER	Driller:	D. IVERSON
Test Hole Number:	R20-2002-18	Driller’s Log:	
Samples:		Total Drill Hole Depth (ft.):	27.0

Well Information

SDGS Well Name:	R20-2002-18	Aquifer:	ALLUVIUM
Water Rights Well:		Management Unit:	
Other Well Name:		Casing Top Elev. (ft.):	5036.00 T
Casing Type:	PVC	Casing Diameter (in.):	2.0
Screen Type:	PVC	Screen Length (ft.):	10.3
Total Casing and Screen (ft.):	26.0	Casing Stick-up (ft.):	0

Notes

NATURAL FILTER PACK FROM 27.0 TO 16.0 FEET BELOW LAND SURFACE, COARSE SAND FROM 16.0 TO 7.5 FEET BELOW LAND SURFACE, BENTONITE GROUT FROM 7.5 FEET TO 1.0 FOOT BELOW LAND SURFACE, CEMENT GROUT FROM 1.0 FOOT BELOW LAND SURFACE TO LAND SURFACE, FLUSH WELL PROTECTOR, CASING TOP 3.0 INCHES BELOW LAND SURFACE.

Lithologic Information

Elevation (ft.)	Depth (ft.)	Description
5036.0 - 5032.0	0.0 - 4.0	CLAY AND SILT, VERY DARK-BROWN TO BLACK; ORGANICS; MICACEOUS; ROUNDED PRECAMBRIAN CLASTS; DRY
5032.0 - 5029.0	4.0 - 7.0	SILT AND CLAY, BROWN TO DARK-BROWN; MICACEOUS; MINOR ROUNDED PRECAMBRIAN CLASTS; MINOR GARNETS; PLANT ROOTS
5029.0 - 5024.0	7.0 - 12.0	SAND, DARK-BROWN TO TAN, VERY FINE-GRAINED, WELL-SORTED; BLACK CLAY STREAKS; MICACEOUS; GRAVEL AT 11 FEET; DRY
5024.0 - 5019.0	12.0 - 17.0	SAND, GRAY; GRAVEL; MAINLY PRECAMBRIAN SCHIST CLASTS, VERY MICACEOUS; DRY
5019.0 - 5014.0	17.0 - 22.0	GRAVEL; VERY FINE-GRAINED, WELL-ROUNDED QUARTZ SAND; TAN TO ORANGE SILT; PRECAMBRIAN SCHIST CLASTS; MICACEOUS
5014.0 - 5009.0	22.0 - 27.0	CLAY, ORANGE; BROWN TO TAN SILT; VERY MICACEOUS; VERY FINE-GRAINED QUARTZ SAND; GARNETS; ORGANICS; DAMP

Monitoring Well R20-2002-20

Location Information			
Legal Location:	NE¼ NW¼ SW¼ NW¼ SEC. 05, T. 1 N., R. 6 E.		
County:	PENNINGTON	Location:	001N06E05BCBA
Basin:	RAPID CREEK	Latitude:	44 04’ 42’’
Hydrologic Unit Code:	10120110	Longitude:	103 25’ 52’’
Land Owner:	PRIVATE RESIDENCE	Ground Surface Elev. (ft.):	4250 T

Project Information			
Project:	SEPTIC SYSTEM DRAIN FIELD		
Drill Date:	07/25/2002	Geologist:	J. SAWYER/J. LESTER
Company:	SDGS	Geologist’s Log:	X
Drilling Method:	AUGER	Driller:	D. IVERSON
Test Hole Number:	R20-2002-20	Driller’s Log:	
Samples:		Total Drill Hole Depth (ft.):	22.5

Well Information			
SDGS Well Name:	R20-2002-20	Aquifer:	ALLUVIUM
Water Rights Well:		Management Unit:	
Other Well Name:		Casing Top Elev. (ft.):	4250.00 T
Casing Type:	PVC	Casing Diameter (in.):	2.0
Screen Type:	PVC	Screen Length (ft.):	10.4
Total Casing and Screen (ft.):	22.2	Casing Stick-up (ft.):	0

Notes

NATURAL FILTER PACK FROM 22.5 TO 11.0 FEET BELOW LAND SURFACE, COARSE SAND FROM 11.0 TO 6.9 FEET BELOW LAND SURFACE, BENTONITE GROUT FROM 6.9 FEET TO 1.0 FOOT BELOW LAND SURFACE, CEMENT GROUT FROM 1.0 FOOT BELOW LAND SURFACE TO LAND SURFACE, FLUSH WELL PROTECTOR, CASING TOP 2.8 INCHES BELOW LAND SURFACE.

Lithologic Information

Elevation (ft.)	Depth (ft.)	Description
4250.0 - 4249.0	0.0 - 1.0	TOPSOIL, DARK-BROWN, SILTY; MICACEOUS; SUBANGULAR, FINE- TO MEDIUM-GRAINED SAND; IRON STAINED, ROUNDED PRECAMBRIAN FRAGMENTS; WELL-ROUNDED, MINOR GARNETS; ROOTS
4249.0 - 4238.0	1.0 - 12.0	SILT, ORANGE-BROWN; CLAY; MINOR SAND; MICACEOUS; MINOR, ANGULAR TO ROUNDED GRAVEL; ROOTS; DRY
4238.0 - 4236.0	12.0 - 14.0	SILT, DARK-GRAY TO BLACK; ROUNDED TO SUBROUNDED, BROWN SAND WITH MINOR ORANGE STAINING; GRAVEL; MICA
4236.0 - 4235.0	14.0 - 15.0	GRAVEL, ANGULAR TO ROUNDED; BROWN, MICACEOUS SILT; GRAY TO BROWN SUBROUNDED QUARTZ SAND WITH SOME IRON OXIDE COATING; ROOTS; DRY
4235.0 - 4234.5	15.0 - 15.5	SILT, GRAY; VERY MICACEOUS; VERY FINE-GRAINED, ROUNDED QUARTZ SAND; GRAVEL; ROOTS; DRY
4234.5 - 4234.0	15.5 - 16.0	SILT, YELLOW-GRAY; CLAY; GRAVEL; MICACEOUS
4234.0 - 4233.0	16.0 - 17.0	CLAY, ORANGE-GRAY TO TAN; SILT; VERY FINE, ANGULAR TO SUBROUNDED QUARTZ SAND; MICACEOUS; MINOR DARK-GRAY PRECAMBRIAN FRAGMENTS; DAMP
4233.0 - 4227.5	17.0 - 22.5	CLAY, OLIVE-GREEN; MICACEOUS; MINOR, MODERATELY IRON-STAINED PRECAMBRIAN CLASTS; VERY WET

Monitoring Well R20-2002-21

Location Information

Legal Location:	NE¼ NW¼ SW¼ NW¼ SEC. 05, T. 1 N., R. 6 E.		
County:	PENNINGTON	Location:	001N06E05BCBA 1
Basin:	RAPID CREEK	Latitude:	44 04’ 42’’
Hydrologic Unit Code:	10120110	Longitude:	103 25’ 51’’
Land Owner:	PRIVATE RESIDENCE	Ground Surface Elev. (ft.):	4244 T

Project Information

Project:	SEPTIC SYSTEM DRAIN FIELD		
Drill Date:	07/25/2002	Geologist:	J. SAWYER/J. LESTER
Company:	SDGS	Geologist’s Log:	X
Drilling Method:	AUGER	Driller:	D. IVERSON
Test Hole Number:	R20-2002-21	Driller’s Log:	
Samples:		Total Drill Hole Depth (ft.):	13.5

Well Information

SDGS Well Name:	R20-2002-21	Aquifer:	ALLUVIUM
Water Rights Well:		Management Unit:	
Other Well Name:		Casing Top Elev. (ft.):	4244.00 T
Casing Type:	PVC	Casing Diameter (in.):	2.0
Screen Type:	PVC	Screen Length (ft.):	10.3
Total Casing and Screen (ft.):	13.0	Casing Stick-up (ft.):	0

Notes

NATURAL FILTER PACK FROM 13.5 TO 6.8 FEET BELOW LAND SURFACE, COARSE SAND FROM 6.8 TO 2.0 FEET BELOW LAND SURFACE, BENTONITE GROUT FROM 2.0 FEET TO 1.0 FOOT BELOW LAND SURFACE, CEMENT GROUT FROM 1.0 FOOT BELOW LAND SURFACE TO LAND SURFACE, FLUSH WELL PROTECTOR, CASING TOP 3.0 INCHES BELOW LAND SURFACE.

Lithologic Information

Elevation (ft.)	Depth (ft.)	Description
4244.0 - 4243.0	0.0 - 1.0	TOPSOIL; DARK-BROWN, VERY FINE-GRAINED, ROUNDED SAND WITH IRON OXIDE COATING; MICACEOUS; SILTY; ANGULAR TO ROUND PRECAMBRIAN CLASTS; ORGANICS; MINOR GARNETS; DRY
4243.0 - 4241.0	1.0 - 3.0	SAND, MEDIUM- TO FINE-GRAINED, MODERATE TO SUBROUNDED; ABUNDANT PRECAMBRIAN CLASTS; GARNETS; MINOR FELDSPAR GRAINS; IRON OXIDE COATING; DARK-BROWN SILT AND CLAY; MICACEOUS; ORGANICS; DRY
4241.0 - 4239.0	3.0 - 5.0	CLAY, DARK-BROWN TO BLACK; MICACEOUS; MEDIUM-GRAINED SAND; MINOR PRECAMBRIAN CLASTS; ROOTS; WET
4239.0 - 4237.0	5.0 - 7.0	CLAY, RED AND BLACK, STREAKY; IRON-STAINED, VERY FINE-GRAINED, SUBROUNDED SAND; GOLDEN-BROWN MICA; ORGANICS; GRAVEL AT 6 FEET; WET
4237.0 - 4236.0	7.0 - 8.0	CLAY, BLACK WITH ORANGE AND TAN STREAKS; SILT; MICACEOUS; ANGULAR TO ROUNDED QUARTZ SAND AND GRAVEL; PRECAMBRIAN CLASTS
4236.0 - 4230.5	8.0 - 13.5	CLAY, GRAY AND TAN; SILT; SAND; ANGULAR TO SUBANGULAR GRAVEL; BROWN MICA; WET

Monitoring Well R20-2002-22

Location Information

Legal Location:	NE¼ NW¼ SW¼ NW¼ SEC. 05, T. 1 N., R. 6 E.		
County:	PENNINGTON	Location:	001N06E05BCBA 2
Basin:	RAPID CREEK	Latitude:	44 04’ 41’’
Hydrologic Unit Code:	10120110	Longitude:	103 25’ 52’’
Land Owner:	PRIVATE RESIDENCE	Ground Surface Elev. (ft.):	4244 T

Project Information

Project:	SEPTIC SYSTEM DRAIN FIELD		
Drill Date:	07/25/2002	Geologist:	J. SAWYER/J. LESTER
Company:	SDGS	Geologist’s Log:	X
Drilling Method:	AUGER	Driller:	D. IVERSON
Test Hole Number:	R20-2002-22	Driller’s Log:	
Samples:		Total Drill Hole Depth (ft.):	9.0

Well Information

SDGS Well Name:	R20-2002-22	Aquifer:	ALLUVIUM
Water Rights Well:		Management Unit:	
Other Well Name:		Casing Top Elev. (ft.):	4244.00 T
Casing Type:	PVC	Casing Diameter (in.):	2.0
Screen Type:	PVC	Screen Length (ft.):	5.4
Total Casing and Screen (ft.):	8.4	Casing Stick-up (ft.):	0

Notes

NATURAL FILTER PACK FROM 9.0 TO 4.5 FEET BELOW LAND SURFACE, COARSE SAND FROM 4.5 TO 2.0 FEET BELOW LAND SURFACE, BENTONITE GROUT FROM 2.0 FEET TO 1.0 FOOT BELOW LAND SURFACE, CEMENT GROUT FROM 1.0 FOOT BELOW LAND SURFACE TO LAND SURFACE, FLUSH WELL PROTECTOR, CASING TOP 3.0 INCHES BELOW LAND SURFACE.

Lithologic Information

Elevation (ft.)	Depth (ft.)	Description
4244.0 - 4242.0	0.0 - 2.0	SILT AND CLAY, REDDISH-BROWN; MICACEOUS; ORGANIC MATERIAL; FINE-GRAINED, ANGULAR TO ROUNDED SAND; IRON OXIDE STAINING; DRY
4242.0 - 4240.0	2.0 - 4.0	CLAY, BLACK TO DARK-BROWN; REDDISH-BROWN SILT; ROUNDED, IRON-STAINED SAND; MINOR MICA; ORGANIC MATERIAL; DRY
4240.0 - 4238.0	4.0 - 6.0	SAND, RED-ORANGE, VERY FINE-GRAINED, ROUNDED; IRON STAINED, ANGULAR, PRECAMBRIAN PEBBLES; DARK-BROWN CLAY AND SILT WITH MINOR ORANGE STREAKS; MINOR MICA; GRAVEL AT 5 TO 6 FEET
4238.0 - 4235.0	6.0 - 9.0	SAND AND GRAVEL, BROWN, FINE- TO MEDIUM-GRAINED, ROUNDED TO SUBANGULAR; ROUNDED PRECAMBRIAN PEBBLES; MINOR CLAY AND SILT; ORGANIC MATERIAL; AUGER REFUSAL AT 9 FEET; WET