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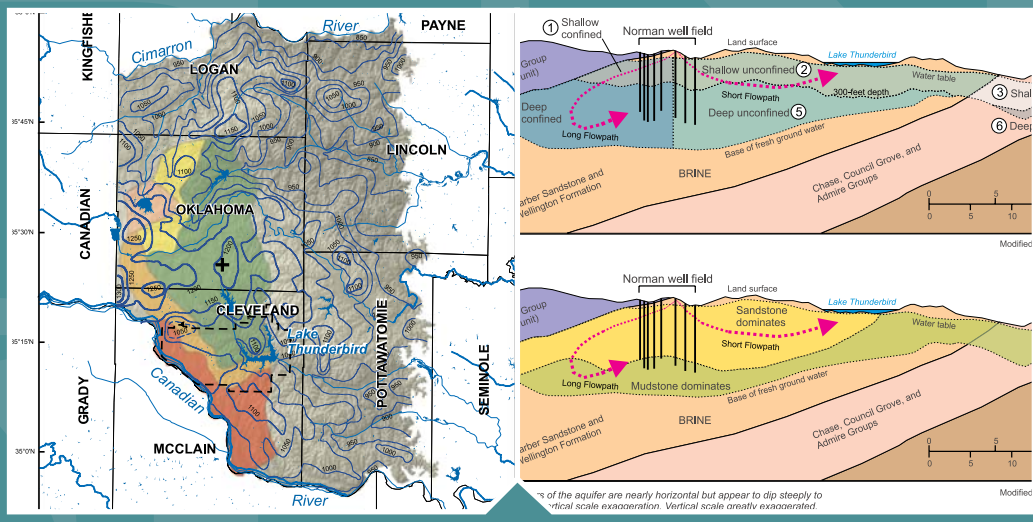
Flow Contribution and Water Quality with Depth in a Test Hole and Public-supply Wells:

Implications for Arsenic Remediation through Well Modification, Norman, Oklahoma, 2003-2006



Office of Research and Development
National Risk Management Research Laboratory, Ada, Oklahoma 74820

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Implications for Arsenic Remediation through Well Modification, Norman, Oklahoma, 2003-2006

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Foreword

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This publication has been produced as part of the Laboratory's strategic long-term research plan. It is published and made available by EPA's Office of Research and Development (ORD) to assist the user community and to link researchers with their clients. Arsenic is a common ground-water contaminant at hazardous waste sites and a widespread issue confronting many drinking water supplies in the U.S. The purpose of this document is to provide a hydrologic and geochemical basis for assessing the potential for restoring existing water supply wells impacted by elevated concentrations of arsenic from natural sources. This report will fill a need for a readily available source of information for water supply managers and others who are faced with this problem. The information provided in this document will be of use to stakeholders such as state and local environmental agencies, public water supply managers, Native American tribes, consultants, contractors, and other interested parties.

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Abstract

The City of Norman, Oklahoma, is one municipality affected by a change in the Environmental Protection Agency's National Primary Drinking Water Regulation for arsenic. In 2006, the maximum contaminant level for arsenic in drinking-water was lowered from 50 to 10 micrograms per liter. Arsenic concentrations in water produced by 32 Norman public-supply wells ranged from less than 1 to 232 micrograms per liter. Some Norman wells with arsenic concentrations marginally exceeding 10 micrograms per liter are suspected of producing water from zones with acceptably low arsenic concentrations and zones with unacceptably high arsenic concentrations. If water with high arsenic concentrations can be limited or excluded from production without causing an excessive decrease in well yield, these wells may be rehabilitated to comply with the new regulation.

The flow contribution and water quality of each producing zone was measured in 11 City of Norman wells to determine which wells were potential candidates for arsenic remediation by well rehabilitation. Depth-dependent flow-contribution and water-quality data were collected under normal production conditions using the U.S. Geological Survey combined well-bore flow and depth-dependent water sampler (U.S. Geological Survey well profiler). The depth-dependent water-quality data collected by the U.S. Geological Survey well profiler were extremely useful as a qualitative tool for identification of zones that degrade water quality in the Norman wells. The depth-dependent water-quality data, even without flow-contribution data, showed the depth at which the water mixture in the well bore was unsuitable for public supply.

Eleven Norman wells were investigated for remediation potential. Most of the selected wells (Wells 06, 07, 13, 15, 18, 23, and 31) showed elevated (greater than 10 micrograms per liter) or near-elevated arsenic concentrations at all depths in the well. For these wells, well-modification techniques would be ineffective in lowering well-head arsenic concentrations to less than 10 micrograms per liter. Wells 02, 05, 33, and 36 showed potential for successful application of well modification techniques for arsenic remediation because greater differences in arsenic concentrations between depths were observed.

Two of the eleven selected wells (05 and 36) were selected for repeated sampling to determine the effects of pump intake relocation on well yield and water quality. Both wells had elevated arsenic concentrations in water from the deepest zone and arsenic concentrations less than 10 micrograms per liter in water from shallower zones. Both wells showed short-term improvements in water quality as the pump was moved to higher locations in the well. In Well 05, arsenic concentration at the well head decreased by about 32 percent and well yield decreased by 12 percent. In Well 36, arsenic concentration at the well head decreased by 84 percent and well yield increased by 13 percent. However, additional samples collected a few months later in Well 36 revealed that improvements in well-head water quality were only temporary.

An alternate remedial approach of zonal isolation was implemented in Well 36. Only the deepest zone in Well 36 (648-658 feet below land surface) was suspected of contributing elevated arsenic concentrations to the well. A retrievable bridge plug was installed at a depth of 640 feet in Well 36 to isolate the suspect zone from production. Unfortunately, the installation of the bridge plug had little effect on well-head water quality. Compared to well-head samples collected prior to the installation of the bridge plug, specific conductance and concentrations of arsenic and chromium at the well head each decreased by only 2 percent after installation of the bridge plug. However, the bridge plug may have been placed too deep to exclude the arsenic-contaminated water from production.

1.0 Introduction

Arsenic in ground water is a major issue affecting many municipalities and water districts in the United States, especially those in the West, Midwest, and Northeast (Welch et al., 2000). Arsenic concentrations in ground water also are locally elevated in many other parts of the United States. Many public-water supply systems, including several systems in central Oklahoma, depend on ground water from aquifers in which arsenic has been identified as a naturally occurring contaminant.

Arsenic is a known carcinogen (World Health Organization, 2001). Ingestion of inorganic arsenic, of which 30-90 percent may be supplied by drinking water, is believed to cause bladder, kidney, lung, and liver cancer in humans (Smith et al., 1992). An individual's risk of dying from arsenic-related cancers as a result of lifetime ingestion of water with arsenic concentration at 50 micrograms per liter (µg/L) could be as great as 13 in 1,000 (Smith et al., 1992). To address this risk, the U.S. Environmental Protection Agency (EPA) completed a review of the 1986 National Primary Drinking-Water Regulation for arsenic in 2000 (EPA, 2001). After considering the available research on the health effects of arsenic consumption, the EPA chose to lower the arsenic maximum contaminant level (MCL) from 50 µg/L to 10 µg/L (EPA, 2001). The EPA estimates that about 3,000 community water systems, including municipal water-supply systems, must employ treatment techniques to meet the revised regulation, which became enforceable on January 23, 2006 (EPA, 2001). Since that time, many water suppliers have been seeking ways to maintain their current level of water production and provide the public with safe, clean drinking water. Chemical treatment of arsenic in ground water is generally cost- and maintenance-prohibitive and remains an option for only the largest water suppliers. Well-head treatment is a potential option, but the costs associated with filter media replacement and disposal, sometimes as hazardous waste, are often too great for smaller municipalities and water districts. Multiwell blending is an available option, but blending often requires the installation of expensive conveyance infrastructure. Well modification to exclude or limit production of arsenic-bearing water is a simpler and more cost-effective solution, but not all wells are good candidates for this remediation technique.

The 2006 change in the arsenic MCL has affected several municipalities in central Oklahoma, including Edmond, Moore, Mustang, Nichols Hills, Noble, Norman, Piedmont, and Yukon (Figure 1), which operate wells in the Central Oklahoma (Garber-Wellington) aquifer. The City of Norman is one of the most affected municipalities in terms of the number of wells in which produced water exceeds 10 µg/L arsenic (Jon Craig, Oklahoma Department of Environmental Quality, written commun., January 26, 2005). Historical arsenic concentrations of produced water from 32 active Norman public-supply wells ranged from less than 1 µg/L to 232 µg/L. Based on maximum detected arsenic concentrations in well-head samples, 11 of these wells could be deemed noncompliant under the old MCL of 50 µg/L arsenic (Table 1). Of the 21 remaining wells, 10 additional wells, which account for about one-third of the total well-field production capacity, likely will be deemed noncompliant under the new arsenic MCL. Through 2003, two-thirds of the wells in the Norman well field had produced at least one well-head sample with arsenic concentration greater than 10 µg/L.

The City of Norman considered several engineering approaches to reduce arsenic concentrations in water reaching the consumer. Blending water from noncompliant wells with water from compliant wells to dilute arsenic in the delivery system was suggested as one cost-effective solution. Multiwell blending alone would not be sufficient, however, to meet the new MCL in most well groupings (CH2M-Hill, 2002). City officials also considered chemical treatment options and surface-water blending in an effort to reduce arsenic concentrations before the water reaches the consumer. These methods of arsenic remediation usually require ground water be pumped to and blended or treated at a treatment plant before the water can enter the distribution system. If these methods are employed, the current (2006) infrastructure and capacity of the water treatment plant are insufficient to satisfy peak demand during the summer months. Unfortunately, these solutions are expensive for municipalities and consumers.

According to one study, the most cost-effective solution currently is to abandon the high-arsenic wells and drill new wells in low-arsenic areas (CH2M-Hill, 2002). In the next phase of well construction, which began in 2006, the city will construct as many as 10 new wells in northeast Norman (Bryan Mitchell, City of Norman, oral commun., 2006) . These wells will replace production lost to the new arsenic rule and add new production to keep up with rapidly growing demand.

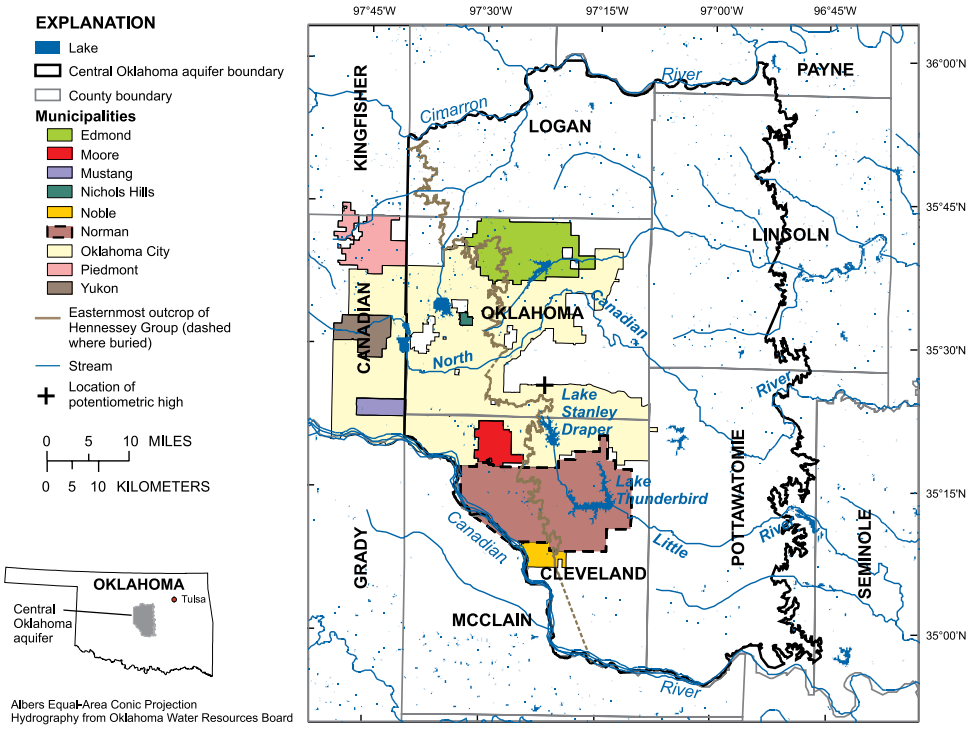


Figure 1. Map showing the Central Oklahoma aquifer and the municipalities that produce some ground water for public supply and have discovered elevated arsenic concentrations in produced water (Jon Craig, Oklahoma Department of Environmental Quality, written commun., 2005). The City of Oklahoma City (light yellow) uses only surface water for public supply. Many of the surrounding municipalities purchase water from Oklahoma City to make up for well production lost to the revised arsenic drinking-water regulation and to supplement supply during periods of high demand.

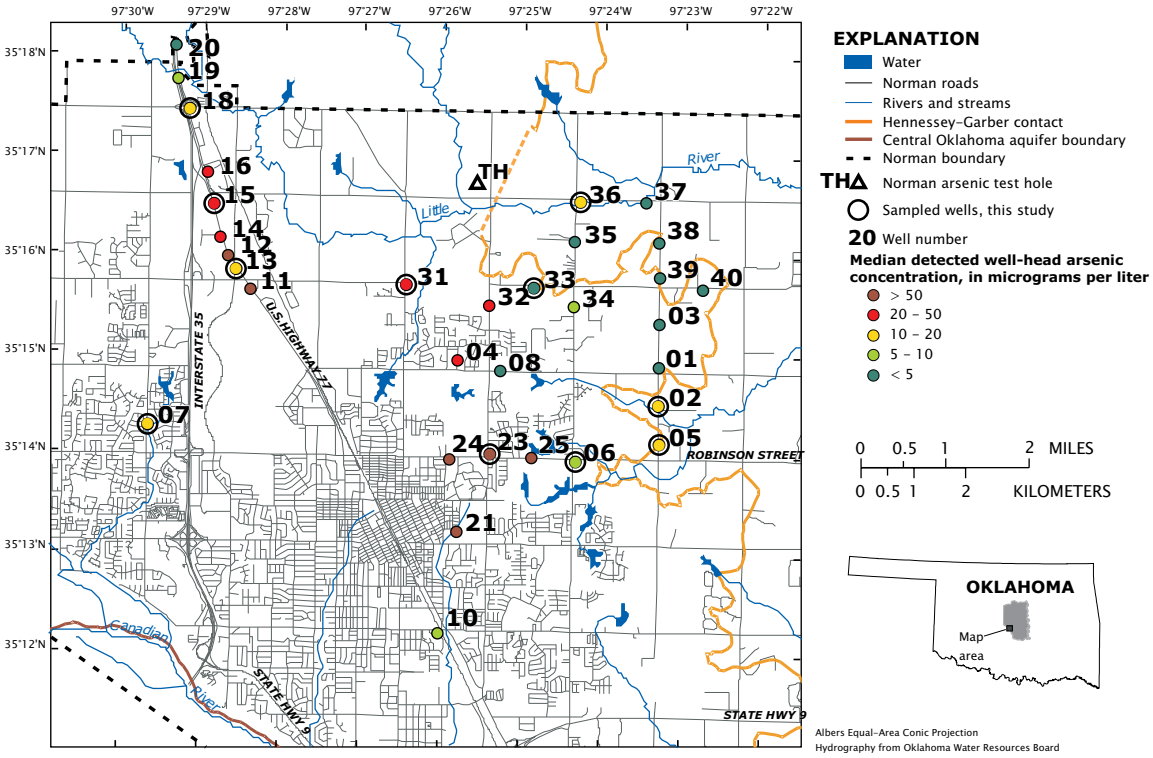


Figure 2. Wells and range of median detected arsenic concentrations in the Norman well field, 2003.

Table 1. Selected construction information and arsenic concentration statistics for public-supply wells in the Norman well field, 2003

Well number	USGS station number	Completion method	Year drilled	Approximate yield, in gallons per minute	Approximate well depth, in feet	Number of available well-head samples	Number of well-head samples below method detection limit	Minimum detected arsenic concentration, in µg/L	Median concentration of detections, in µg/L	Mean concentration of detections, in µg/L	Maximum detected arsenic concentration, in µg/L
01	351452097232201	Screen or wire wrap	1963	200	693	4	1	0.6	1.3	2.3	5.0
02	351426097232201	Screen or wire wrap	1963	335	732	9	2	5.0	10.1	9.8	13.0
03	351518097231801	Screen or wire wrap	1963	230	726	5	2	0.4	0.9	0.8	1.1
04	351458097254901	Gun perforated	1964	215	--	13	1	20.0	47.0	58.3	112.0
05	351409097231801	Gun perforated	1982	212	695	8	1	2.0	12.0	30.2	150.0
06	351357097242001	Gun perforated	1982	218	645	8	1	7.1	9.4	10.6	16.0
07	351414097293901	Gun perforated	1982	182	745	11	1	3.2	14.5	17.5	28.0
08	351451097251701	Gun perforated	1982	228	--	8	3	1.8	1.9	2.1	3.0
10	351213097260001	--	--	177	567	5	1	3.7	7.7	7.3	10.0
11	351538097283401	Slotted casing	1942	144	635	9	1	35.0	67.0	61.5	82.0
12	351559097283601	Slotted casing	1944	193	670	7	1	26.0	69.3	67.6	102.0
13	351550097283801	Gun perforated	1951	190	678	8	2	7.9	11.3	17.3	32.0
14	351609097284601	Gun perforated	1952	182	--	9	0	27.0	47.0	49.1	79.7
15	351648097285101	Gun perforated	1953	164	674	9	1	15.0	36.5	36.1	53.0
16	351643097285601	Slotted casing	1953	122	--	5	0	1.6	34.0	25.1	39.0
18	351726097290901	Gun perforated	1953	147	693	7	2	9.3	10.8	12.4	20.0
19	351742097291501	Gun perforated	1953	174	700	8	2	5.7	9.8	9.9	15.0

Table 1. Selected construction information and arsenic concentration statistics for public-supply wells in the Norman well field, 2003 (continued)

Well number	USGS station number	Completion method	Year drilled	Approximate yield, in gallons per minute	Approximate well depth, in feet	Number of available well-head samples	Number of well-head samples below method detection limit	Minimum detected arsenic concentration, in µg/L	Median concentration of detections, in µg/L	Mean concentration of detections, in µg/L	Maximum detected arsenic concentration, in µg/L
20	351807097292101	Gun perforated	1953	163	704	8	2	2.9	3.9	15.3	73.0
21	351314097254701	Gun perforated	1955	164	648	12	0	21.0	58.0	50.1	69.0
23	351401097252301	Gun perforated	1957	250	650	31	0	14.0	84.0	77.4	135.0
24	351357097255401	Gun perforated	1957	--	--	2	0	230.0	231.0	231.0	232.0
25	351358097245701	Screen or wire wrap	1959	--	624	8	0	41.0	57.5	57.5	69.0
31	351542097262801	Gun perforated	1997	172	660	4	0	16.0	21.0	25.3	43.0
32	351530097252601	Gun perforated	1997	112	--	4	1	16.0	33.9	29.0	37.0
33	351541097245301	Gun perforated	1998	219a	--	3	1	1.3	2.0	2.0	2.6
34	351530097242301	Gun perforated	1998	257	--	4	1	0.9	8.3	6.2	9.6
35	351609097242301	Gun perforated	1998	144	--	5	3	0.8	1.0	1.0	1.3
36	351633097241901	Gun perforated	1999	260a	695	5	1	0.7	19.8	16.6	26.0
37	351633097233001	Gun perforated	1999	247	--	4	0	0.8	1.5	1.5	2.2
38	351609097232001	Screen or wire wrap	2000	255	--	5	3	0.8	1.0	1.0	1.2
39	351548097231901	Gun perforated	2000	246	--	5	1	4.0	4.7	5.1	6.9
40	351541097224701	Gun perforated	2000	255	--	5	2	0.4	1.3	1.1	1.5

[a] Measurements by City of Norman on 7-26-04

The City of Norman produces ground water from the Central Oklahoma (Garber-Wellington) aquifer, a multilayered sandstone, siltstone, and mudstone aquifer. Arsenic occurs naturally in small concentrations throughout water of the Central Oklahoma aquifer, but some areas are underlain by ground water with arsenic concentrations much greater than the new MCL (Parkhurst et al., 1994; Schlottmann et al., 1998). Most elevated arsenic concentrations (greater than 10 µg/L) occur in deep wells in the western, confined part of the aquifer (Figure 3). The City of Norman recognized this geographic trend in ground-water arsenic concentrations at the well-field scale and stopped drilling new exploratory test holes in western parts of the city. Since 1990, Norman well-field expansion has been almost exclusively in the unconfined aquifer northeast of the city. Arsenic concentrations measured in these newer wells are often less than 10 µg/L.

Observations of elevated arsenic concentrations at depth in the aquifer (Schlottmann et al., 1998) indicate that well remediation could be employed to improve water quality and retain production capacity at noncompliant wells. Based on historical well-head samples, some Norman wells with marginal arsenic concentrations are suspected of producing water from zones with both acceptably low and unacceptably high arsenic concentrations. If water with high arsenic concentrations can be limited or excluded from production without causing an excessive decrease in well yield, these wells may be rehabilitated to comply with the new arsenic drinking-water regulation. To determine which wells were potential candidates for arsenic remediation by well rehabilitation, though, the flow contribution and water quality of each producing zone needed to be measured in individual wells.

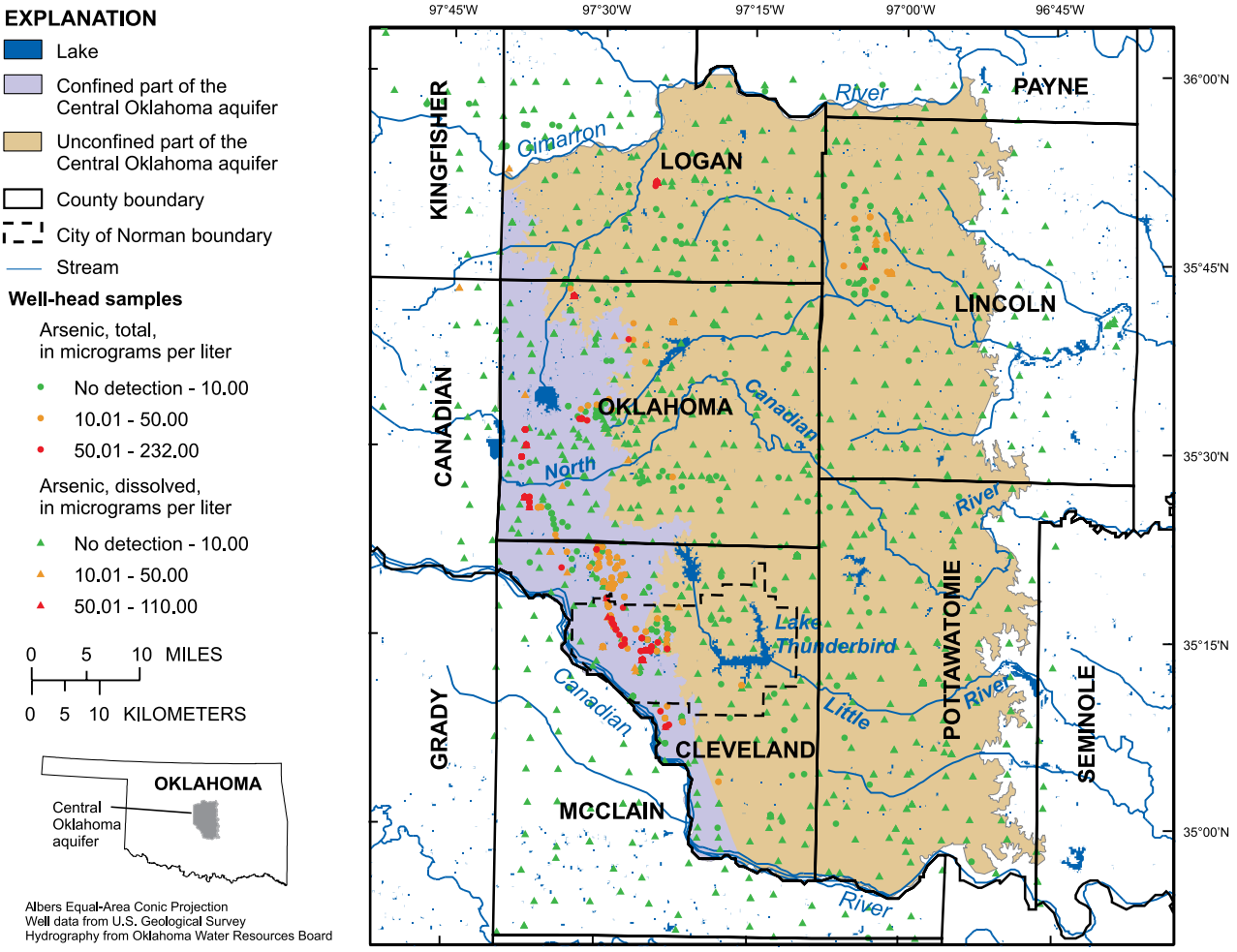


Figure 3. Map showing the extent and location of the Central Oklahoma aquifer with well-head arsenic concentrations (1977-2004). The dark shaded part of the aquifer represents the part that is confined by the Hennessey Group. Deep municipal supply wells in the confined part are most likely to exceed the U.S. Environmental Protection Agency maximum contaminant level for arsenic.

2.0
Purpose and Scope

This report summarizes an investigation of ground-water quality with depth and well-rehabilitation techniques in one part of the Central Oklahoma aquifer. The study location of Norman, Oklahoma, was selected because the regional and local Central Oklahoma aquifer systems have been well characterized after more than 20 years of concentrated research. Using this research as a foundation, an investigation of individual well construction and dynamics was undertaken to determine if rehabilitation of water wells by well modification is possible. The primary goals of this report are to (1) present depth-specific water-quality data from a test hole and selected public-supply wells that exceed the 10 µg/L arsenic MCL, (2) describe the utility and limitations of a new method for determining flow contribution and water quality with depth in a pumping well, (3) assess wells for the possibility of remediation by well modification, and (4) evaluate the effectiveness of well-modification approaches in bringing marginally noncompliant wells into compliance with the 10 µg/L arsenic MCL.

This report, which relies heavily on a conceptual model of the Central Oklahoma aquifer system developed under the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program, is presented in three parts (Figure 4). The first part includes description and analysis of cored sections, logs, and water samples retrieved from a test hole in northern Norman. The second part describes an 11-well investigation that measured changes in water quality with depth to identify wells that might be good candidates for remediation. Included in this part are an analysis of ground-water flow to selected wells and a comparison of selected techniques for depth-dependent data collection in wells. The third part documents attempts to decrease well-head arsenic concentrations using well-modification techniques at two City of Norman wells.

Concurrent with this report, a study (S.T. Paxton, Oklahoma State University, written commun., 2005) provided a detailed stratigraphic framework of the aquifer units that, in part, guided the selection of a location for drilling, logging, coring, and water-quality sampling of a test hole in the Norman area (Figure 4). The purposes of the test hole were to (1) provide information on water quality with depth in an area of the aquifer that was relatively undeveloped and undisturbed (prior to well completion and production), and (2) collect rock material for laboratory analysis of rock and water interactions (outside the scope of this report) for proposed future research involving in-situ arsenic remediation (Figure 4).

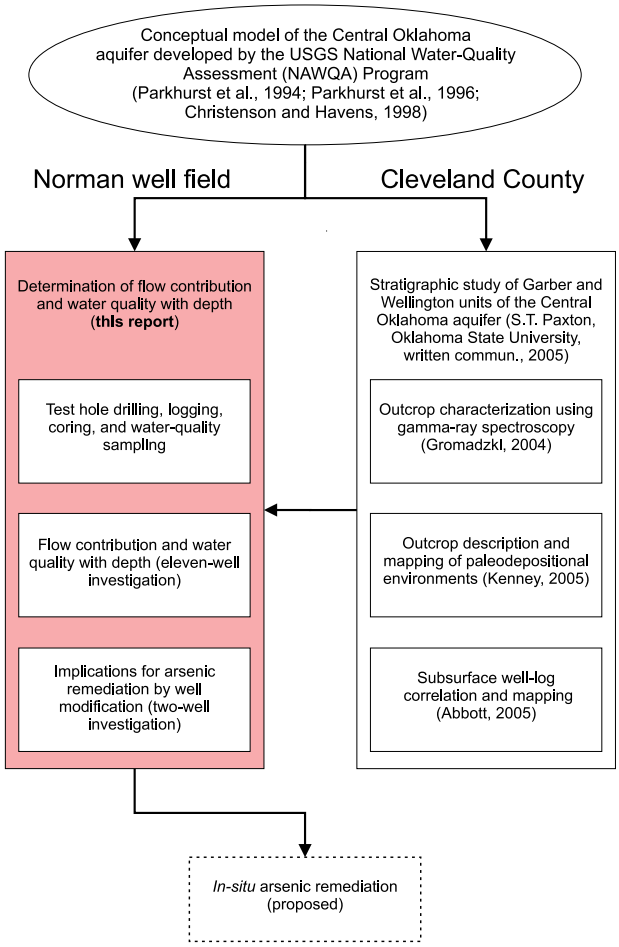


Figure 4. Organizational structure of research activities in the Central Oklahoma aquifer.

By undertaking this project, the USGS and the EPA have benefited from an improved description of the hydrologic and geochemical controls on naturally occurring arsenic with depth in an oxygen-rich aquifer. This knowledge may benefit water suppliers in the United States that draw high-arsenic water from multilayered aquifers. With the prospect of identifying individual wells for remediation using new depth-dependent sampling techniques, the City of Norman may be able to avoid costly installation of new wells, conveyance infrastructure, and treatment technologies. Most importantly, the City may be able to decrease arsenic exposure to citizens and protect them against potentially unnecessary costs associated with treatment.

3.0
Description of the Study Area

The City of Norman is the county seat and primary population center of Cleveland County, which occupies 536 square miles (U.S. Census Bureau, 2000) in the southwestern part of the Central Oklahoma aquifer (Figure 1). Norman was the third most populous city in Oklahoma in 2000 (after Oklahoma City and Tulsa) with about 95,000 residents (U.S. Census Bureau, 2000). Norman also was one of the fastest-growing Oklahoma municipalities of 20,000 or more people in 2000, with an increase of 15,623 people (19.5 percent) since 1990 (U.S. Census Bureau, 2000). At the current growth rate, the city population will exceed 200,000 by 2040.

City of Norman Water Use

Historically, the City of Norman has supplied residents with municipal water from a combination of ground-water and surface-water sources. Prior to the impoundment of Lake Thunderbird (Figure 1) in 1965 and the construction of the Norman water-treatment plant in 1966, Norman relied solely on ground-water wells to supply drinking water to residents (City of Norman, 2002). Since 1983, when the capacity of the Norman water-treatment plant expanded, Lake Thunderbird has served as the primary source of water for the City of Norman. The annual allocation of Lake Thunderbird water to Norman is 9,460 acre-feet (3,082 million gallons; City of Norman, 2002). Norman first met this allotment in 1988, and has routinely met or exceeded this allotment since 1995. In 2003, total water production by the City of Norman was about 3,500 million gallons (City of Norman, 2004), which is slightly less than previous years. About 79 percent of this total was supplied by Lake Thunderbird (Figure 1), and about 20 percent was supplied by the Norman well field.

The City of Norman, like most other municipalities in Central Oklahoma, maintains a connection to the Oklahoma City water distribution system and can augment supply in times of emergency or increased demand by purchasing treated surface water from Oklahoma City (City of Norman, 2004). In 2003, Norman purchased about 53 million gallons (about 1 percent) of supplemental water from Oklahoma City to satisfy peak demand during the summer months (Table 2). However, the purchased water is provided at a cost that is greater than the Norman water-rate structure can support over long periods of time (City of Norman, 2002). To decrease annual water usage and reliance on Oklahoma City, the City of Norman adopted a strategy of conservation education paired with a water-rate structure in which users with greatest consumption purchase water at the greatest unit price (City of Norman, 2002).

Because Norman and other municipalities surrounding Oklahoma City began taking noncompliant wells out of production in response to the new arsenic drinking-water regulation, reliance on Oklahoma City supply is likely to increase. Norman will need to add new water wells to satisfy increasing demand and remain self-sufficient in the future. A plan has been developed for placing and constructing new wells, but Norman officials also are interested in possible remediation of marginally noncompliant wells.

Table 2. Annual water production by the City of Norman, 1983-2003

Year	Total production (million gallons)	Source		
		Treatment plant (Lake Thunderbird)	Well field	Purchase from Oklahoma City
1983	2,956.4	2,131.9	824.5	0.0
1984	2,803.6	2,500.4	303.2	0.0
1985	2,903.7	2,424.1	479.6	0.0
1986	2,847.3	2,651.9	195.4	0.0
1987	3,006.2	2,839.0	227.2	0.0
1988	3,425.5	3,139.0	286.5	0.0
1989	3,163.2	2,910.7	252.5	0.0
1990	3,427.2	3,139.4	287.8	0.0
1991	3,458.8	3,073.0	385.8	0.0
1992	3,277.7	2,998.5	279.2	0.0
1993	3,534.1	3,043.1	491.0	0.0
1994	3,714.0	3,346.5	367.5	0.0
1995	3,912.3	3,236.3	676.0	0.0
1996	3,965.5	2,930.3	1,035.2	0.0
1997	3,811.5	2,832.2	979.3	0.0
1998	4,572.4	3,313.0	1,259.4	0.0
1999	4,291.8	3,327.0	964.8	0.0
2000	4,366.7	3,364.6	993.0	9.1
2001	4,446.6	3,498.4	923.9	24.3
2002	4,289.2	3,571.9	715.8	2.5
2003	3,463.0	2,662.0	747.6	53.4

City of Norman Well Field

The Norman well field consisted of 32 wells in the Central Oklahoma aquifer in 2003 (Table 1, Figure 2). Based on maximum detected arsenic concentrations in well-head samples, 11 of these wells could be deemed noncompliant under the old MCL of 50 µg/L arsenic (Table 1). Of the 21 remaining wells, eight had median detected arsenic concentrations greater than 10 µg/L and ten had maximum detected arsenic concentrations greater than 10 µg/L (Table 1, Figure 2). Eight to ten additional wells, which account for about one-third of the total well-field production capacity, likely will be deemed noncompliant under the new arsenic MCL. The loss of these additional wells will force the city to buy additional water from Oklahoma City during the summer months.

Though the oldest active wells in the Norman well field were constructed in the 1940s, most older active wells were constructed from 1951 through 1964 (Table 1). Older active wells are concentrated along U.S. Highway 77 in northwest Norman (Figure 2). Most of the younger wells were constructed from 1997 through 2000 and are located in the northeast part of the well field (Table 1; Figure 2). In 2006, as part of a city water plan, the city began construction of new wells in the northeast part of the well field. In 2004, Norman also acquired three additional wells when the City of Norman annexed the community of Hall Park. These wells were not included in this study. The University of Oklahoma also operated several wells in the area of the Norman well field; these wells were not included in this study.

City of Norman Well Construction

Well construction in the City of Norman well field is typical of municipal well construction throughout the Central Oklahoma aquifer. Most Norman public-supply wells have a cement-annulus and gun-perforated openings (Figure 5, Table 1). A smaller percentage of wells have a gravel-pack annulus and screen, wire-wrap, or slotted openings (Table 1).

Depths of Norman wells are usually 600 to 800 feet below land surface (Table 1), and perforations (openings through which water enters the well) begin around 300 feet below land surface (Bryan Hapke, City of Norman, oral commun., 2003). The density of perforations across perforated intervals is usually 4 to 6 shots per foot, though some older wells may have been reperforated to increase production. The number of distinct perforated sandstone zones ranges from 5 to 21, but is most commonly around 10. Typically, the upper perforated zones are considerably thicker than the lower perforated zones. Individual sandstone zones in the Garber Sandstone can exceed 40 feet in thickness, but typically are from 5 to 15 feet in thickness. Open (perforated or screened) intervals usually coincide with a single, well-defined sandstone zone, but occasionally, open intervals extend across multiple sandstone zones with thin intervening mudstones (Figure 5).

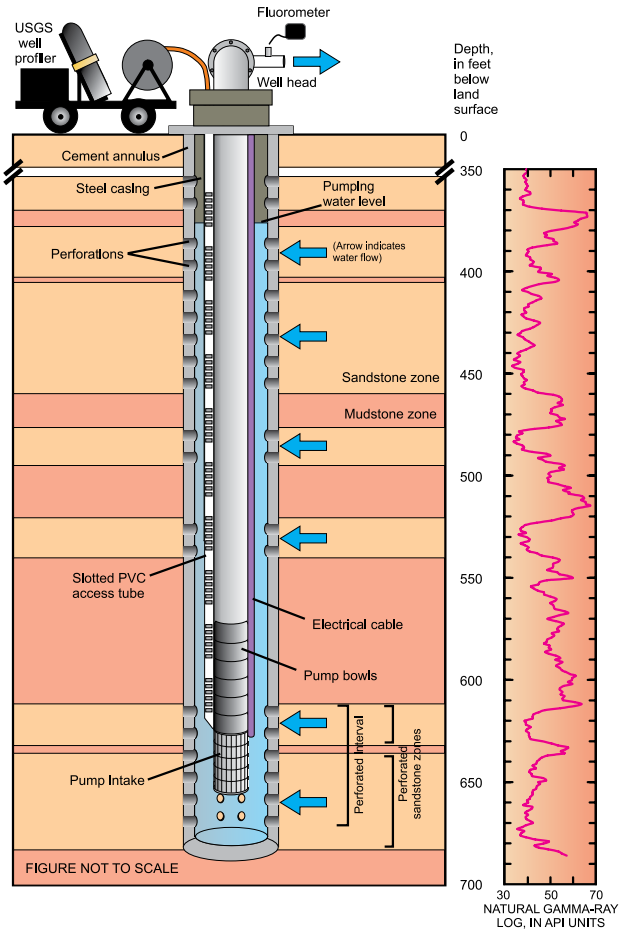


Figure 5. Typical construction of a gun-perforated public-supply well showing deployment of the combined well-bore flow and depth-dependent water sampler. (USGS well profiler). Increased gamma radiation generally indicates a greater percentage of clay (mudstone) in the aquifer rocks. Lesser gamma radiation indicates a coarser-grained (sandstone) unit. Abbreviation: API, American Petroleum Institute.

Casing diameter is 10 inches in all Norman wells except Wells 31 and 32, which are 12 inches in diameter (Bryan Hapke, City of Norman, oral commun., 2003). Scale buildup is common, occasionally exceeding 0.125 inch in older wells and in sections near the water level. Most scaling in the Norman well field is dark and multicolored with presumed iron oxides (orange, yellow, and red). Only some lighter-colored spots effervesced with the application of dilute (5 percent) hydrochloric acid. Another variety of scale is light-colored with

white, cream, and pink layering (Figure 6). The light-colored variety of scale is rich in calcium carbonate as it effervesces strongly with the application of dilute hydrochloric acid. The light-colored scale is a substantial problem for well maintenance in parts of the city of Moore, about 2 miles north of Norman (Robert Pistole, Veolia Water, oral commun., 2005).

Figure 6. Varieties of well-scale buildup in Cleveland County, Oklahoma. The dark scale (left) is typical of wells in the Norman well field. The light scale (right) is more commonly found in the southern Moore well field. Photographs by Jerrod Smith, U.S. Geological Survey.

Lift in a Norman well typically is by a 40- to 60-horsepower submersible pump supported by a column of 4- or 5-inch diameter steel pipe (Bryan Hapke, City of Norman, oral commun., 2003). The pump is typically installed near the bottom of the well, usually within 100 feet of the bottom (Bryan Hapke, City of Norman, oral commun., 2003; Figure 5). When the pump is set below all perforations, a pump shroud is sometimes installed for cooling purposes. The motor is powered by a one-inch diameter electrical cable which is banded to the pump column between pipe connections. In some wells, centralizers were attached to the pump column.

In 2006, the static water level in most selected Norman municipal wells was from 300 to 500 feet below land surface. The pumping water level was greater than 500 feet below land surface in one well in western Norman. Drawdown, or the difference between the static and pumping water levels in Norman wells, can be in excess of 100 feet. Yield of Norman wells is usually from 150 to 300 gallons per minute, though well output across the well field ranges from about 100 to about 350 gallons per minute (Table 1).

Geophysical logs are available (from the City of Norman) for nearly every well in the Norman well field. Typically a natural gamma-ray or spontaneous-potential log was used for qualitative determinations of basic lithology. Often, a perforation log was noted on the gamma-ray or spontaneous-potential log, and each sandstone with a thickness of 5 feet or greater was usually perforated. Natural gamma-ray logs are most common, but not all gamma-ray logs were calibrated to American Petroleum Institute (API) standard units. On the gamma-ray log trace, deflections to the right (higher values) indicate finer-grained mudstones and deflections to the left (lower values) indicate coarser-grained sandstones (Figure 5). Resistivity and neutron logs are available for some wells and can yield useful information about the zonal water content. For gun-perforated wells, cement-bond logs are often available, and bond is usually greater than 90 percent, indicating few cavities between the casing and aquifer material.

Hydrogeologic Setting: Central Oklahoma Aquifer

All wells in the Norman well field are completed in the Central Oklahoma aquifer. The Central Oklahoma aquifer, as defined by Parkhurst et al. (1994), underlies about 3,000 square miles in parts of Cleveland, Lincoln, Logan, Oklahoma, Payne, Pottawatomie, and Seminole Counties. The aquifer is bounded by the Cimarron River on the north, the Canadian River on the south, and the easternmost outcrop of aquifer rocks on the east (Parkhurst et al., 1994). The western boundary of the aquifer is the Canadian-Kingfisher/Cleveland-Oklahoma-Logan County line which approximately represents the westernmost extent of freshwater circulation (Figure 7; Parkhurst et al., 1994).

The Central Oklahoma aquifer is composed of the Garber Sandstone, Wellington Formation, and Chase, Council Grove, and Admire Groups of Permian age (Figure 7). The overlying alluvial and terrace deposits also are included in the aquifer (Figure 7) because no confining layer underlies the alluvial and terrace deposits, and ground water flows readily between these deposits and the underlying Permian-age geologic units. The Central Oklahoma aquifer is partially confined above by Permian-age shale of the Hennessey Group

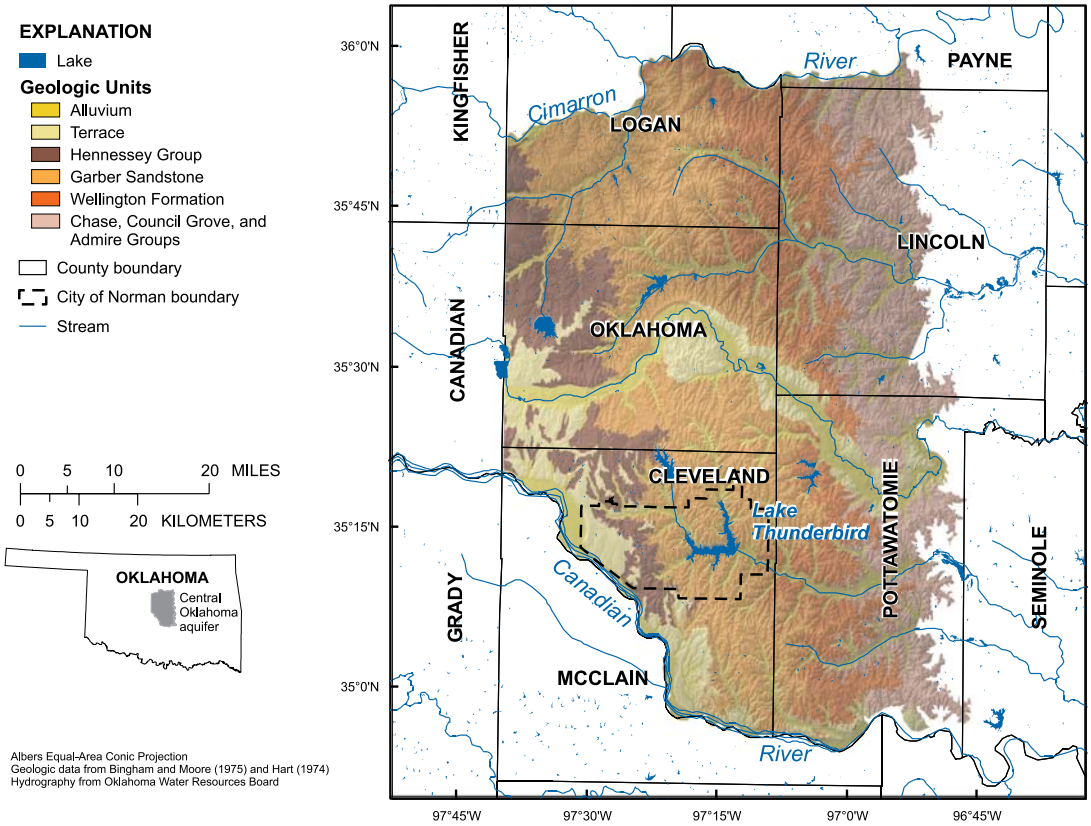


Figure 7. Surficial geology of the Central Oklahoma aquifer.

and below by Pennsylvanian-age shale of the Vanoss Formation. The aquifer is unconfined to the east of Edmond, Oklahoma City, and Norman (Figure 1).

The Central Oklahoma aquifer is known locally as the Garber-Wellington aquifer because the greatest quantities of usable water are in the Garber Sandstone and Wellington Formation (Garber-Wellington) of Permian age. Though the Garber Sandstone and Wellington Formation extend beyond the Central Oklahoma aquifer boundary, these units typically do not produce large quantities of water beyond the boundary. The units include fine-grained, crossbedded sandstones interbedded with siltstones and mudstones of fluvial-deltaic origin (Parkhurst et al., 1994). Because the Garber-Wellington contact is difficult to delineate in the area of Norman, the Garber Sandstone and Wellington Formation are treated as one unit in this report. About 75 percent of the total thickness of the Garber-Wellington is sandstone in southeastern Oklahoma County (Parkhurst et al., 1994). The percentage of sandstone decreases in all directions from southeast Oklahoma County, reaching as little as 25 percent in parts of Cleveland County south of Norman (Wood and Burton, 1968). The total thickness of the Garber Sandstone and Wellington Formation is usually from 1,100 to 1,600 feet (Christenson et al., 1992).

Annual rainfall in central Oklahoma is about 36 inches (Johnson and Duchon, 1995). Recharge to the saturated zone of the Central Oklahoma aquifer is estimated to be 1 to 2 inches per year (Parkhurst et al., 1996). According to maps of the potentiometric surface of the aquifer, rivers in the study unit are not a substantial source of recharge (Parkhurst et al., 1996). Instead, potentiometric contours of the Central Oklahoma aquifer indicate that ground water discharges to most river systems, with the notable exception of the North Canadian River (Parkhurst et al., 1996).

Specific capacities computed for wells in the Central Oklahoma aquifer range from 0.16 to 15 gallons per minute per foot of drawdown, but are usually less than 5 gallons per minute per foot of drawdown (Parkhurst et al., 1996). Transmissivities vary widely across the aquifer, but median transmissivities computed by Parkhurst et al. (1996) ranged from 260 to 450 square feet per day. Computed hydraulic conductivities were mostly from 2.5 to 10 feet per day, with a median of 4.5 feet per day (Parkhurst et al., 1996).

Given the economic importance of the aquifer, surprisingly little published research has focused on the stratigraphy of the Garber Sandstone and Wellington Formation (Kenney, 2005). USGS NAWQA

studies (Parkhurst et al., 1994; Parkhurst et al., 1996; Christenson and Havens, 1998) have focused mostly on the geochemical and geohydrologic characteristics of the aquifer rather than the sedimentary geology. Because a comprehensive understanding of the aquifer system could not be achieved without some insight into the stratigraphic framework of the aquifer, the EPA commissioned an investigation of aquifer stratigraphy. Gromadzki (2004), Abbott (2005), and Kenney (2005) completed the most recent studies of the stratigraphy and sedimentology of the Garber Sandstone and Wellington Formation. The results of these stratigraphic studies of the Garber Sandstone and Wellington Formation were summarized and presented to EPA (S.T. Paxton, Oklahoma State University, written commun., 2005).

The Garber Sandstone and Wellington Formation consist of stacked channel bars, floodplain deposits, and related fluvial facies (S.T. Paxton, Oklahoma State University, written commun., 2005). These facies grade into one another vertically and horizontally. The variable lithofacies, lack of continuous marker beds, and scarcity of fossils in the aquifer makes traditional stratigraphic correlation difficult, especially over distances greater than one mile (Abbott, 2005). The heterogeneous stratigraphy also supports a complex ground-water flow system with the potential for complex geochemical interactions over time and space.

Schlottmann et al. (1998) divided the Central Oklahoma aquifer into six geohydrologic zones on the basis of changes in lithology, water chemistry, and the presence of confined or unconfined conditions (Figure 8). Most domestic, stock, and irrigation wells in the aquifer draw water from the shallow (confined and unconfined) aquifer system, which is less than 300 feet below land surface. Wells in the shallow unconfined aquifer system typically do not produce water with arsenic concentrations greater than 10 µg/L (Schlottmann et al., 1998; Becker, 2006). Most public-supply wells, however, bypass the shallow aquifer system and produce water from the deep (confined and unconfined) aquifer system, which is greater than 300 feet below land surface (Figure 8). Wells that tap the deep confined aquifer system (like those in western Norman) are more likely to exceed the arsenic MCL, according to statistics calculated by Schlottmann et al. (1998). Schlottmann et al. (1998) estimated that 30.2 percent of wells in

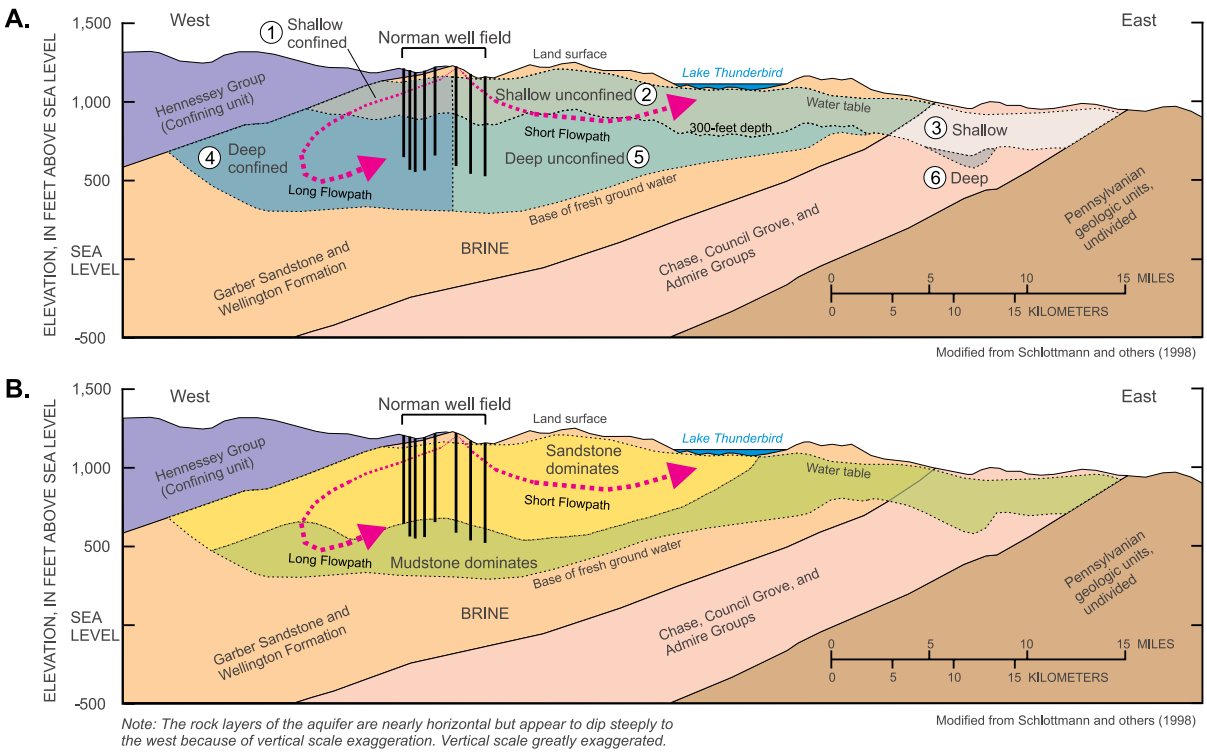


Figure 8. Generalized hydrogeologic section of the Central Oklahoma aquifer showing the location of the Norman well field. Red arrows illustrate theoretical long and short flow paths that supply water to the Norman well field. Flow paths begin as recharge that enters the aquifer to the north of these cross-sections. The aquifer is composed of six geohydrologic zones (A, 1-6) based on changes in lithology, water chemistry, and the presence of confined or unconfined conditions. The amount of mudstone in the aquifer rocks (B) generally increases with depth in the Norman well field, becoming the dominant lithology in the deep confined and deep unconfined zones.

the deep confined aquifer system produced water with arsenic concentrations exceeding 50 µg/L, and that only 2.4 percent of wells in the deep unconfined aquifer system produced water with arsenic concentrations exceeding 50 µg/L.

Conceptual Model of the Central Oklahoma Aquifer

Arsenic is an element commonly found in aquifer rocks, therefore most arsenic contamination in ground water used for public supply results from natural processes. In aquifers containing abundant iron oxides, arsenic can become a natural water contaminant in two general ways: (1) reductive dissolution of iron oxides, and (2) desorption from iron oxides. The latter process is the main cause of elevated arsenic in water of the Central Oklahoma aquifer (Christenson et al., 1998). Arsenic in ground water also commonly occurs in two oxidation states: arsenite (As III) and arsenate (As V). These arsenic species tend to occur as protonated oxyanionic complexes in ground water (Stollenwerk, 2003). In this report, unless otherwise noted, the word arsenic is used to refer to all arsenic regardless of oxidation state.

The geohydrologic processes and geochemical conditions in the Central Oklahoma aquifer are well characterized as a result of monitoring conducted during the USGS NAWQA Program. A series of NAWQA-supported studies, beginning in the late 1980s and concluding in the mid-1990s, determined the rock composition, water chemistry, and ground-water movement in the Central Oklahoma aquifer. The findings of these studies led to the development of a comprehensive conceptual model of the aquifer system. For detailed explanation and results of previous studies, see Christenson and Havens, 1998 (USGS Water-Supply Paper 2357-A) for a summary of NAWQA findings with a focus on rock and water chemistry; Parkhurst et al., 1994 (USGS Water-Supply Paper 2357-B) for a retrospective analysis of available water-quality data through 1987; and Parkhurst et al., 1996 (USGS Water-Supply Paper 2357-C) for modeling and analysis of geochemistry and ground-water movement through the aquifer. Recent findings have expanded on the findings of the NAWQA Program studies. Some of these findings on water and rock characteristics, summarized briefly here, formed a foundation for data interpretations in this report.

Major characteristics of water in the Central Oklahoma aquifer:

1. Dissolved arsenic concentrations range from <1 to 110 µg/L and appear to exist almost exclusively as arsenate (Ferree et al., 1992; Schlottmann, 2001);
2. Chromium, selenium, and uranium also exist as oxyanions in the aqueous phase and behave similarly to arsenate. Ranges of chromium, selenium, and uranium concentrations in aquifer water are <1 to 100, <1 to 190, and <1 to 318 µg/L, respectively (Ferree et al., 1992);
3. Neutral to alkaline pH; pH ranges from 6.0 to 9.6 (Schlottmann et al., 1998); pH tends to be greater in deeper wells (Becker, 2006);
4. Dissolved oxygen concentrations greater than 1 milligram per liter (mg/L) are present in most water, indicating oxic conditions (Schlottmann, 2001);
5. Water can contain large concentrations of sulfate and chloride in areas near the base of fresh water, in the confined part of the aquifer, and near the discharge areas of regional flow paths (Schlottmann et al., 1998); and
6. Other constituents that may limit use of water include fluoride and boron.

Major characteristics of rock in the Central Oklahoma aquifer:

1. Predominantly sandstone, siltstone, and mudstone, with some thin, localized conglomerates (Gromadzki, 2004; Abbott, 2005; Kenney, 2005);
2. Dolomite cement is common to the conglomerate and some sandstone and mudstone (Nkoghe-Nze, 2002; S.T. Paxton, Oklahoma State University, written commun., 2005);
3. Cation-exchange capacity of clay-fraction subsamples ranges from 20 to 50 milliequivalents per 100 grams (Parkhurst et al., 1996);
4. Exchangeable sodium, as a percentage of exchangeable cations in clay subsamples, is greater in deeper sandstones than in shallow sandstones; Exchangeable sodium is often less than 1 percent in shallow sandstones (Parkhurst et al., 1996);
5. Iron oxide and iron oxyhydroxide minerals are present as cements and coatings on framework grains in aquifer rocks (Parkhurst et al., 1996);
6. Evidence of paleosols (Sokolic, 2003; S.T. Paxton, Oklahoma State University, written commun., 2005);
7. Presence of reaction fronts indicating mobilization (or dissolution and reprecipitation) of iron oxide (Parkhurst et al., 1996; S.T. Paxton, Oklahoma State University, written commun., 2005);
8. Arsenic concentrations range from <1 to 62 milligrams per kilogram (mg/kg) in drill-core samples, with an average of 7.3 mg/kg (Mosier et al., 1990);

9. Average concentrations of chromium, selenium, and uranium in drill-core samples are 56, 1.4, and 3.6 mg/kg, respectively (Mosier et al., 1990);
10. Elemental composition of drill-core samples ranges from about 0.1 to 14 percent iron (Mosier et al., 1990); and
11. Iron, arsenic, and dolomite concentrations are all elevated in isolated conglomerate layers (S.T. Paxton, Oklahoma State University, written commun., 2005).

Ground-water Flow and Geochemical Processes

Ground water in the Central Oklahoma aquifer originates as infiltration from precipitation, a process known as recharge. Where the aquifer is unconfined, ground-water recharge occurs areally, that is, everywhere the Garber Sandstone, Wellington Formation, and alluvium and terrace deposits are at the land surface. Ground water in the unconfined part of the aquifer mostly follows relatively short flowpaths (on the order of feet to miles) before discharging to streams (Parkhurst et al., 1996). In the confined part of the aquifer, most ground-water recharge originates in a relatively small area near a potentiometric high centered in south-central Oklahoma County (Figure 9). The potentiometric high corresponds to a structural high that is expressed in contour maps of the base of each aquifer unit (Christenson et al., 1992; Parkhurst et al., 1996). The potentiometric high also corresponds to the part of the aquifer with the thickest sequence of sandstone (Christenson, 1998). Ground water in the confined part of the aquifer mostly follows longer flowpaths (on the order of miles to tens of miles) before discharging to streams.

In the area of the Norman well field, short and long flow paths influence ground-water quality. Unconfined ground water with short flow paths and short residence times travels to the southeast through Cleveland County and discharges relatively quickly to the Little River drainage system near Lake Thunderbird (Parkhurst et al., 1996; Figure 8). Estimated ages of water (times since recharge) along short flow paths are on the order of hundreds to thousands of years (Parkhurst et al., 1996). Confined ground water with longer flow paths and greater residence times travels to the southwest, descending under the confining unit before turning back to the east (Figure 8). Estimated ages of water along long flow paths are on the order of tens of thousands of years (Parkhurst et al., 1996).

In the unconfined part of the aquifer, recharge water picks up carbon dioxide from the vadose zone, which can make water mildly acidic. Dolomite, which is present as cement in many aquifer rocks, dissolves to equilibrium in the presence of carbon dioxide (Parkhurst et al., 1996). The dissolution of dolomite causes a small increase in pH, to values near neutral (7.0). The general

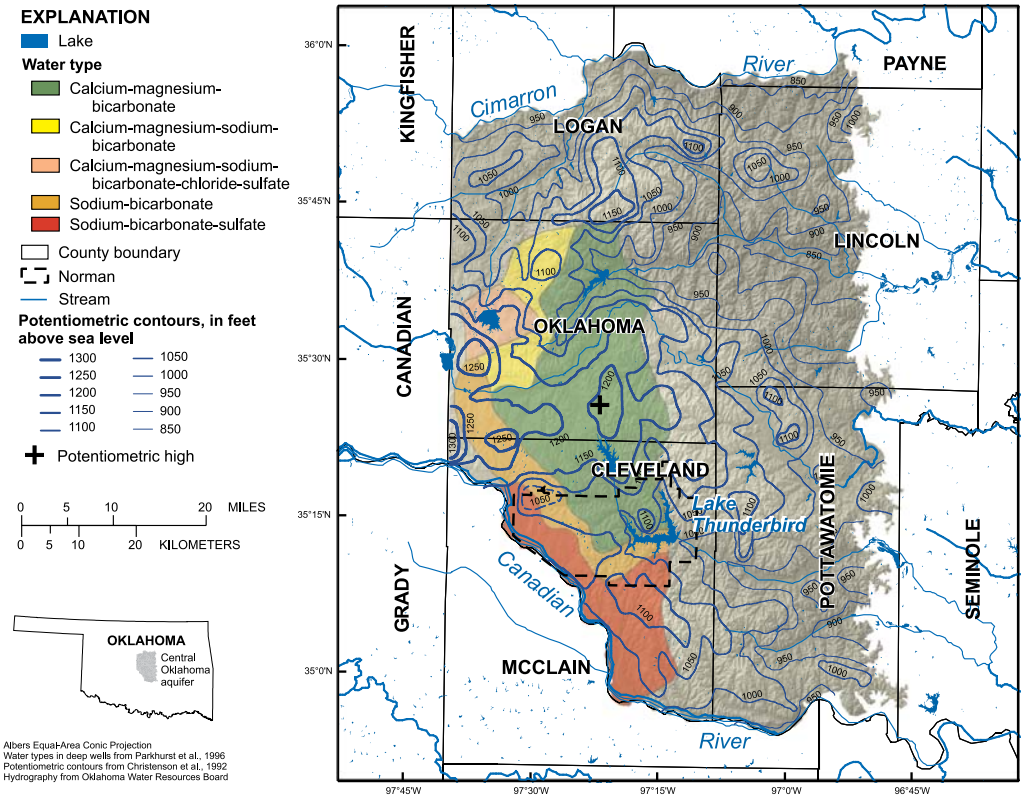


Figure 9. Potentiometric contours and water type from deep wells in the Central Oklahoma aquifer.

water type in the shallow, unconfined part of the aquifer is calcium-magnesium-bicarbonate, identifying dolomite dissolution as the dominant geochemical process (Figure 9).

Deeper ground water comes in contact with aquifer rocks (mudstone) rich in mixed-layer illite-smectite clay minerals with high cation-exchange capacity (Figure 8). Calcium and magnesium ions in the water are exchanged for sodium ions in the clays. When calcium ions leave the solution, the water becomes undersaturated with respect to dolomite. In response, more dolomite dissolves and more dissolved calcium and magnesium are available for ion exchange. As cation exchange and dolomite dissolution continue at depth in the confined aquifer, the pH gradually increases. Along longer flow paths at depth and in the confined part of the aquifer (Figure 8), where carbon dioxide is limited, this process can elevate the pH to greater than 8.5 – the pH value at which arsenic is expected to begin desorbing from iron-oxide mineral coatings (Figure 10).

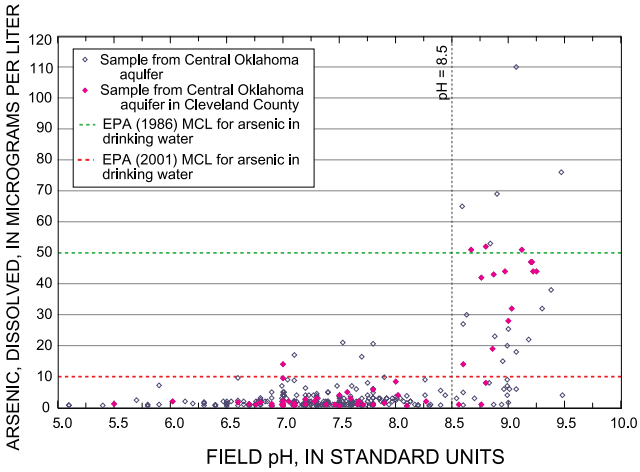


Figure 10. Relation between dissolved arsenic concentration and pH for wells in the Central Oklahoma aquifer [MCL, Maximum Contaminant Level].

Multiple lines of evidence support the Central Oklahoma aquifer conceptual model. Though ground-water pH in the aquifer ranges from about 6.0 to 9.6 (Schlottmann et al., 1998), most elevated arsenic concentrations occur in water with pH greater than 8.5 standard units (Figure 10). Because longer flow paths and greater quantities of clay-rich rocks (which participate in ion-exchange) tend to occur in deeper parts of the aquifer (Figure 8), a strong relation exists between well depth and arsenic concentration (Becker, 2006). Because longer flow paths and clay-rich rocks tend to occur in the confined part of the aquifer (Figure 8), a strong relation exists between well location and arsenic concentration (Figure 2).

Spatial Variability of Water Quality in the Deep Aquifer System

Because water chemistry changes along flow paths, the aquifer area can be divided into different water-type regions that are consistent with the Central Oklahoma aquifer conceptual model. For wells greater than 300 feet deep, Parkhurst et al. (1994) realized that the major-ion composition of water near the potentiometric high (area of greatest recharge) was dominated by the products of dolomite dissolution. The calcium-magnesium-bicarbonate water type in that area is characteristic of relatively recent recharge water that has not been substantially altered by cation-exchange reactions (Figure 9). Water in that area is hard and has a near-neutral pH. Along the southern flow path under the confining unit, the character of water changes to a sodium-bicarbonate water type around northwestern Norman as sodium in clays is exchanged for calcium and magnesium in ground water (Figure 9). Water in this region is soft and has a pH that approaches 8.5 standard units. Farther south and west under the confining unit, the water is dominated by sodium-bicarbonate-sulfate (Figure 9). This change in chemistry is thought to reflect the influence of dissolution of sulfate-bearing rocks in the overlying Hennessey Group. In regions with this water type, cation exchange has removed nearly all calcium and magnesium from the water. This water is soft and is likely to have a pH greater than 8.5 standard units. Sodium-bicarbonate and sodium-bicarbonate-sulfate water types are closely associated with elevated arsenic concentrations in the Central Oklahoma aquifer.

Studies by the USGS have established that arsenic in the aqueous phase is not evenly distributed with depth in some parts of the Central Oklahoma aquifer (Parkhurst et al., 1996; Schlottmann and Funkhouser, 1991; Schlottmann et al., 1998). Aqueous concentrations of arsenic (as well as the geochemically related oxyanions of chromium, selenium, and uranium) are sometimes only elevated in one or two zones in a well. Schlottmann et al. (1998) and Schlottmann and Funkhouser (1991) measured aqueous arsenic (65 µg/L), selenium (380 µg/L), and uranium (318 µg/L) concentrations exceeding the MCLs (10, 50, and 30 µg/L, respectively) in a deep sandstone zone from a test hole near Edmond, Oklahoma (Figure 1); all other sampled zones in the same test hole had concentrations that were less than the MCLs. If zones with elevated trace element concentrations can be identified and sealed off from production, concentrations measured at the well head may be decreased to meet drinking-water regulations.

4.0
Approach and Methods

Selection of Wells for Investigation

The data compiled in Figure 11 represent well-head arsenic concentrations of active Norman wells. Through 2003, nearly two thirds of the wells in the Norman well field had produced water with arsenic concentration greater than 10 µg/L (Table 1; Figure 11). The total number of available water samples from each well is listed in black near the top of Figure 11, and the number of censored (nondetected) analysis values is represented by a blue number at the bottom of Figure 11. These well-head samples were analyzed by several different laboratories with reporting levels ranging from 0.2 to 15 µg/L. Because some censored values had a practical quantitation limit greater than or equal to 10, the median value of detections (excluding the censored values) was selected to represent central tendency of well-head arsenic concentrations.

A consensus decision was made to assess water quality with depth in 11 Norman wells (Table 1, Figures 11-12) in meetings with officials representing the City of Norman, EPA, and USGS. Several factors were considered for well selection including median arsenic concentration, well production rate, well age, and well location in the well field. The best candidates for successful remediation were considered to be those wells that had (1) marginal well-head arsenic concentrations (near 10 µg/L), (2) wide variation in well-head arsenic concentrations, and (3) high water-production rates (greater than 200 gallons per minute). These wells are most likely to benefit from isolation of a single, high-arsenic zone and are the least likely to suffer from loss of production from that zone. Because remediation of older wells nearing life-expectancy was not cost-effective, more recently constructed wells were selected over older wells. Some wells, such as Well 07, were given more

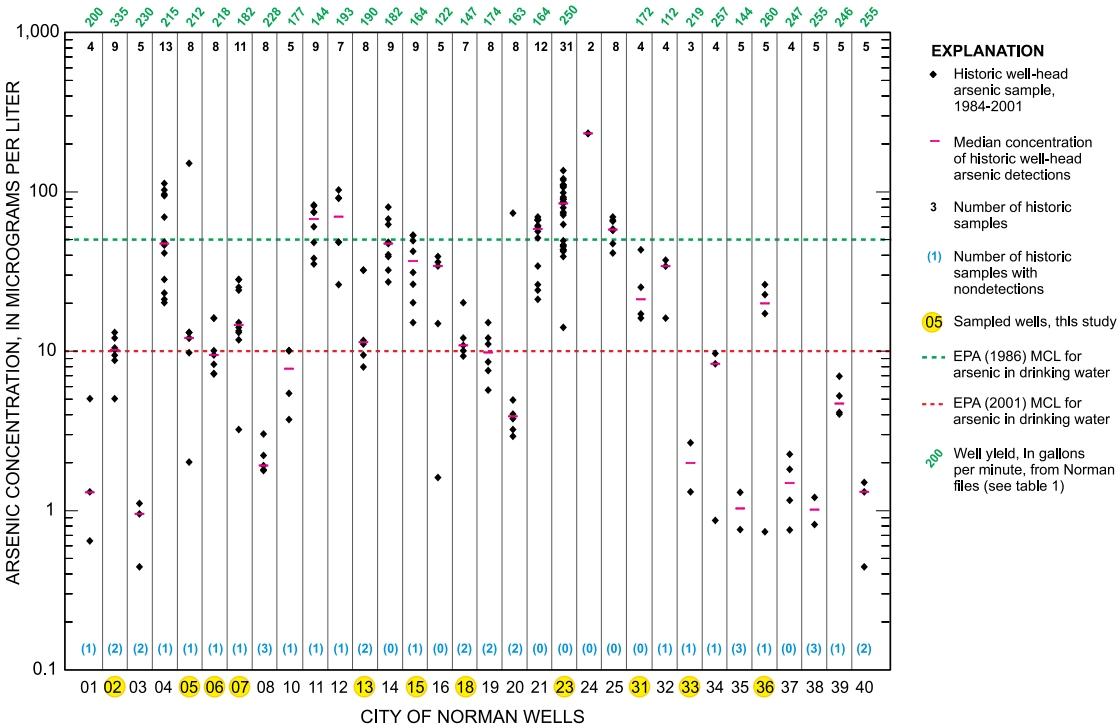


Figure 11. Well-head concentrations, median values, and maximum contaminant levels for arsenic in the Norman well field, 2003.

consideration because those wells were located in parts of the aquifer not represented by other selected wells (Figure 12). One Well (23) with unusually high arsenic concentration was selected for a comparison of well-sampling methods.

Historical well-head arsenic concentrations were useful for identifying wells that might have potential for rehabilitation by well modification. Some wells (for example Well 05, Figure 11) showed large variation in arsenic concentration over time. Variation in reported arsenic concentrations over time indicates that a well has differences in water quality with depth. Assuming that data were accurately reported and that water chemistry of contributing zones has remained relatively constant over the life of the well, there must be at least one zone in the well that produces water with arsenic concentrations equal to or greater than the maximum detected value (for example, 150 µg/L for Well 05). Likewise, there must be at least one zone that produces water with arsenic concentrations equal to or less than the minimum detected value (2 µg/L for Well 05). The difference between the maximum and minimum detected concentrations, therefore, gives an indication of the range in water quality to expect if depth-dependent or zone-specific samples were collected. If all well-head samples are nearly equal in arsenic concentration over time (for example Well 25, Figure 11), there may be little variation in water quality with depth and little potential for successful remediation by well modification.

The temporal variability in well-head arsenic concentrations, from a compliance standpoint, can be problematic for a public water-supply system. Some wells, which may have always tested around 5 µg/L arsenic, may suddenly test slightly greater than 10 µg/L and become noncompliant. In remediation attempts, the only way to increase the likelihood that well-head arsenic concentrations will never exceed 10 µg/L is to exclude from production all zones that contribute arsenic at concentrations greater than 10 µg/L.

Remediation Options

This report examines two potentially reasonable remediation options which involve well modification. The ultimate goal of both options is to decrease the proportion of flow supplied by contaminated zones and increase the proportion of flow supplied by uncontaminated zones.

The simplest and least costly remediation option is to move the pump intake to a new location and/or decrease the capacity of the pump. If the pump intake is moved farther from a contaminated zone, the well may produce a lesser proportion of water from the contaminated zone (Ground Water Protection Council, 2005). This strategy is reversible, minimizes potential loss of production, and involves minimal down time for the well. This method, however, may provide only a temporary remediation solution because well-head contaminant concentrations may fluctuate with seasonal or prolonged changes in

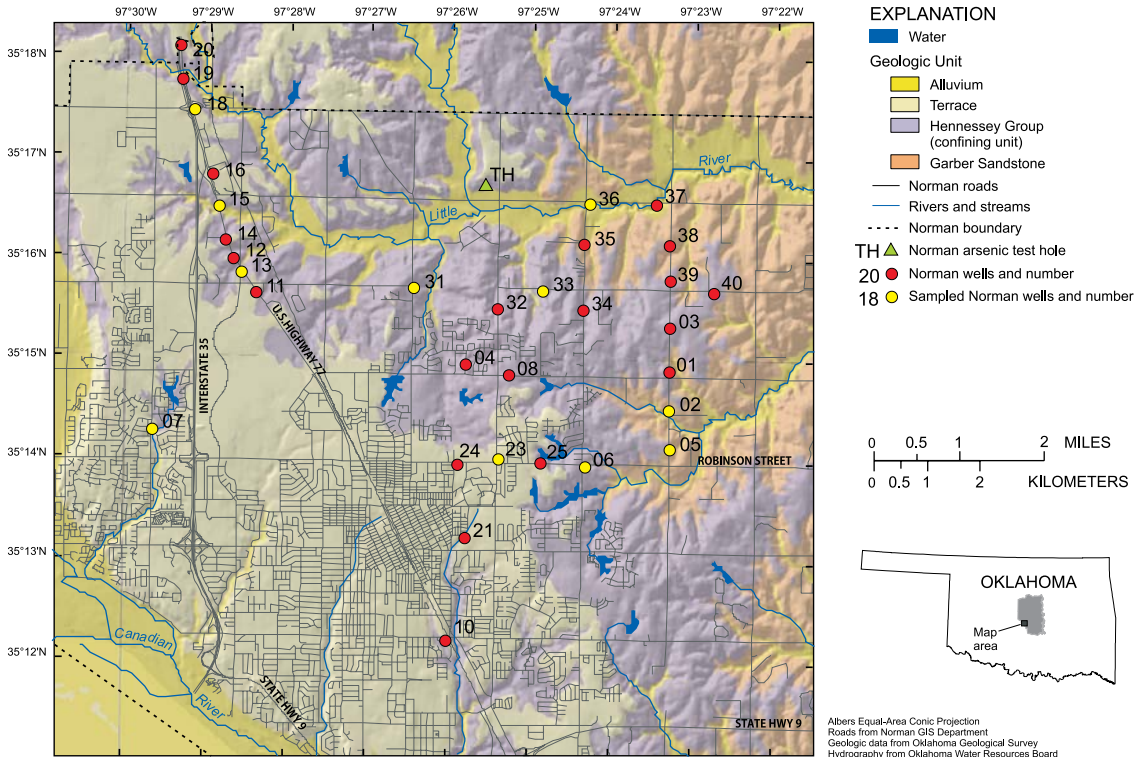


Figure 12. Selected wells and surficial geology of the Norman well field, 2003.

aquifer water levels. Over time, the well could produce an increasingly greater proportion of flow from the contaminated zone and subsequently degrade well-head water quality.

Another well-head remediation strategy is zonal isolation, which involves plugging an arsenic contributing zone. This strategy is designed to ensure that a known contaminated zone will no longer contribute to well production. This technique is well suited for use in open-hole wells and gun-perforated wells with a cement annulus. This strategy also could be applied in screened wells, but a gravel-pack annulus makes complete isolation more difficult. In aquifers composed of incompetent rocks, gun-perforated completions are preferred for this technique because the cement annulus prohibits flow behind the casing. Zonal isolation also is best suited for use in aquifers where contributing zones are hydraulically separated by thick, laterally continuous, and relatively impermeable units. The Central Oklahoma aquifer was well suited for the zonal-isolation strategy because a large number of marginal wells have gun-perforated completions, and producing sandstone zones are commonly separated or compartmentalized by thick mudstones. However, intervening mudstones may not be laterally continuous or impermeable near the well.

Methods of Hydraulic Data Collection

Two types of data collection methods were used in this study to record changes in well yield and water quality as a function of depth in Norman municipal wells. Impeller-flowmeter logging (Leve, 1964) and straddle-packer testing (Hess, 1993) were applied only at Norman Well 23. An alternate method, the USGS combined well-bore flow and depth-dependent water sampler (USGS well profiler; U.S. patent numbers 6,131,451 and 6,164,127; Izbicki et al., 1999), was applied at all Norman wells investigated in this study.

Flowmeter Logging and Straddle-packer Testing

A flowmeter is often used to identify producing zones and quantify production from individual zones in wells; a variety of types of down-hole flowmeters are available to obtain these data. An impeller (spinner) flowmeter is a wire-line tool consisting of a small impeller that is free to rotate around a vertically oriented axis. As the tool is slowly lowered or raised in a pumping well, the flowmeter records the vertical component of water velocity through the cross-sectional area of the well and produces a continuous log of flow contribution with depth. The flow-contribution log describes the proportion of well yield supplied by each producing zone. The impeller flowmeter is capable of detecting velocities as small as 0.2 foot per second (ft/s) (Hess, 1982), but the tool is sensitive to changes in cross-sectional area with depth in the well, which can be caused by scale buildup, centralizers, or damaged casing. When present, these features can cause large fluctuations in velocity, and, along with the effects of turbulent flow in the well, can make the impeller-flowmeter velocity profile noisy and difficult to interpret. When conducting an impeller-flowmeter test, if the pump is set near the bottom of the well, the pump must be raised or replaced with a temporary pump set just below the pumping water level. This configuration reduces the possibility of damaging the tool or getting the tool lodged in the well. If the location of the pump during testing is different from the location of the pump during production, though, the flow dynamics created during testing also can be different from the flow dynamics during production.

Water quality with depth is most commonly recorded using packer tests. Inflatable straddle-packer tests are the traditional method used to obtain depth-dependent samples in cement-annulus, gun-perforated water wells. Straddle packers are high-pressure, rubber bladders separated by a length of slotted pipe (Figure 13). The packers are adjustable to any separation width and can be used to test zones of different thickness. In preparation for packer testing, the pump, column pipe, and all other equipment must be removed from the well. The casing wall should be brushed to create a smooth surface on which to complete the packer seal. When a producing zone has been selected for testing, the packer spacing is adjusted to match the thickness of the target zone and the packers are lowered to the desired depth by adding known lengths of pipe. Using compressed gas or pressurized fluid supplied by a line at the surface, the bottom packer is inflated just below the target zone and the top packer is inflated just above the target zone. When both packers are inflated and properly sealed against the casing, the water level inside the packer string equilibrates to the head of the isolated zone. A low-capacity (5 gallons per minute) submersible pump is connected to a small diameter conductor pipe and placed inside the pipe supporting the packers (Figure 13). The pump is lowered to a depth that is well below the static water level of the isolated zone. When the pump is activated, water from the isolated zone flows into the slotted pipe separating the packers, into the submersible pump, and through the conductor pipe to the surface (Figure 13). After purging for some time, field tests can be made and laboratory water samples can be collected from the discharge hose. For packer testing, a cement-filled and perforated annular space is more desirable than a gravel-filled annular space because the cement-filled annulus prohibits intra-annular communication between permeable zones.

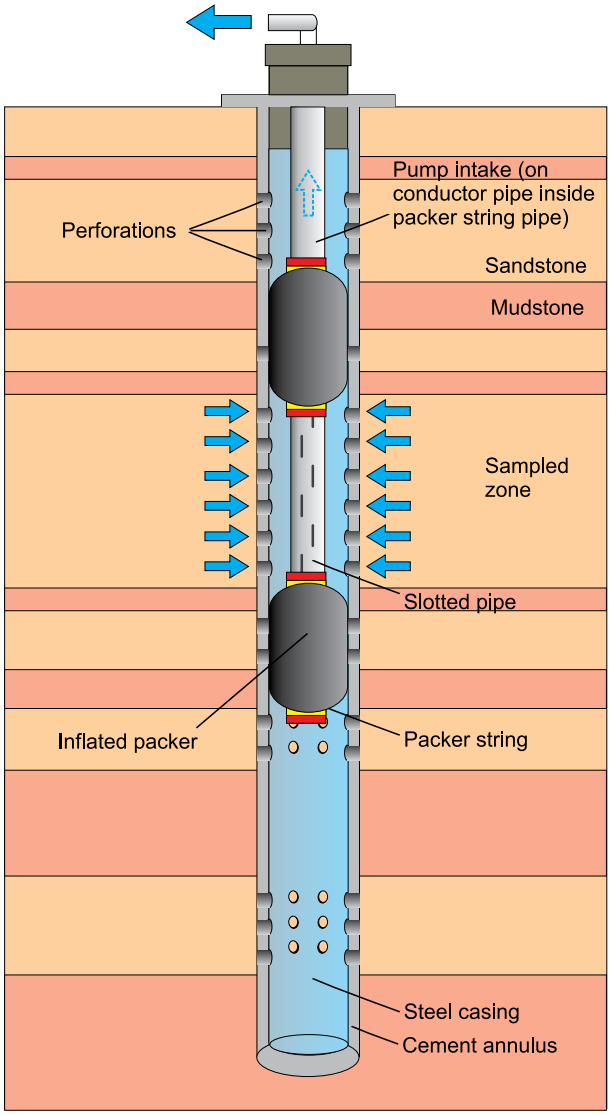


Figure 13. Apparatus used in straddle-packer testing of Norman Well 23.

When the results of impeller-flowmeter logging and packer testing are combined, the data may be used to determine which zones have the greatest influence on water quality at the well head. This information, in turn, guides the selection of appropriate well rehabilitation techniques.

USGS Well Profiler

Though flowmeter logs and packer tests are the traditional methods of depth-dependent flow and water-quality data collection, these methods do not always provide the most useful and representative information on well properties during production. Preparation for using these methods also is time consuming, invasive, and expensive. As an alternate method, the USGS has used a combined well-bore flow and depth-dependent water sampler, referred to as the USGS well profiler, to quantify the contribution of water from perforated or screened zones and to collect samples at various depths in a pumping well (Izbicki et al., 1999). The USGS well-profiler method, as compared to traditional methods, can be considerably less expensive and requires less down-time of the well. In terms of data quality, the most important advantages of the USGS well profiler are that all data collection is performed under true production conditions and that the technique requires minimal modification to the well. The methods described document the adaptation and application of the USGS well profiler to the style of well construction common in the Norman well field.

The USGS well profiler is a slim, high-pressure, multipurpose hose that can be raised and lowered between the pump column and well casing (or borehole wall) by using a motorized hose reel (Figure 14a). A line counter at the surface (Figure 14b) reports the depth of the hose outlet and electrical-tape markings (Figure 14c) on the multipurpose hose are used to confirm counter readings. The hose outlet is equipped with a 0.25-inch pressure-activated, in-line check valve. A 0.25-inch diameter, 35-foot long, stainless-steel cable weight is attached below the check valve to keep the hose hanging straight in the well (Figure 14b). A threaded coupler is welded to the cable weight and small drilled openings in the coupler allow dye tracer solution and sample water to pass in and out of the hose.

A machine-slotted polyvinyl-chloride (PVC) access tube was installed in most selected wells to facilitate access with the USGS well profiler. Obstructions and irregularities in the borehole or casing wall and pump column, such as scale, can impede the movement of the sampling hose in and out of the well. Other obstructions include electrical cables, steel pipe joints, banding material, pump shrouds, pump bowls, centralizers, airlines, and lost tools (Figure 14f). Nearly all obstructions can be bypassed by pulling the pump column and reinstalling the column with a 1.25-inch diameter, 0.375-inch slotted PVC access tube, which is banded to the column pipe every 20 feet (Figure 14g). In a few wells, the access tube was placed next to the electrical cable (Figure 14g), but the access tube usually was placed opposite the electrical cable (Figure 15). The bottom end of the access tube was open, cut at an angle, and attached just above the pump intake. Only the submerged part of the access tube was slotted; the upper part of the access tube was blank 1.25-inch PVC.

Figure 14. Photographs of the USGS well profiler (A) and related equipment including line counter and stainless-steel cable weight (B), multipurpose hose with depth marking (C), Teflon-lined sample hose attachment (D), and field fluorometer and laptop for data logging (E). Photograph F shows the multipurpose hose stuck between a stainless-steel band and the pump column. Photograph G shows the solution to this problem; most well profiler data were collected inside a slotted PVC access tube (arrow), which was banded to the pump column. Photographs by Jerrod Smith, U.S. Geological Survey.

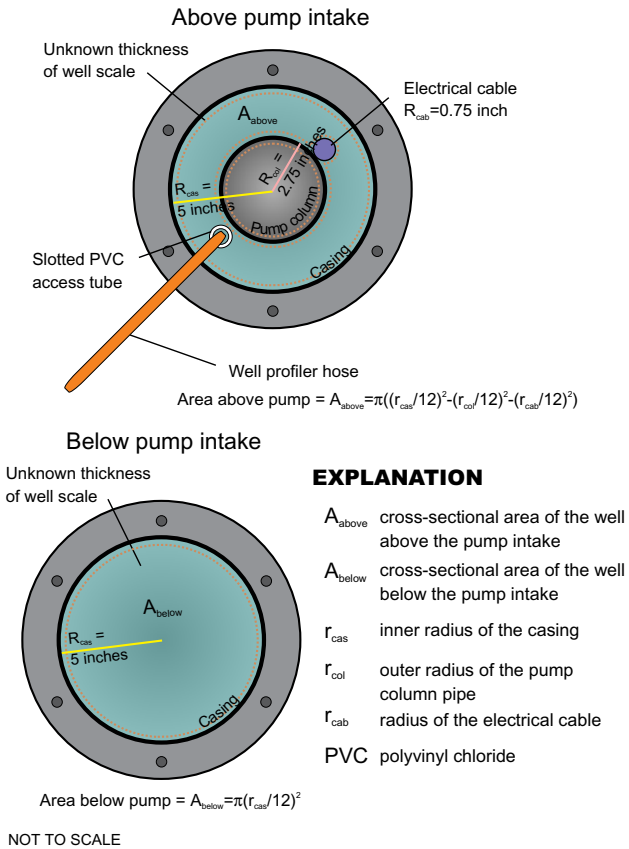


Figure 15. Generalized horizontal well cross sections and unit area computations above and below the pump intake in wells with a 10-inch casing and 4-inch pump column.

With the exception of Wells 23 and 33, wells in this study were equipped with a slotted PVC access tube in which all tests were performed. All tests at Well 23 were conducted without a PVC access tube. In Well 33, only depth-dependent sampling was conducted inside the access tube. The well-profiler hose became stuck in the wells repeatedly at the beginning of this investigation (Figure 14f), but few problems were encountered after installation of the PVC access tube in the sampled wells. Only two problems occurred during tests inside the access tube. One problem was caused by incorrect placement of the tube outlet relative to the pump. The second problem occurred when the cable weight disconnected prematurely, fell down the access tube, and broke through the PVC tube. The weight became lodged in the PVC tube and blocked access into the well.

Tracer-pulse Velocity Logging

Prior to sampling, a velocity profile is constructed to determine the percentage of the total well discharge coming from each contributing depth interval (which may include multiple zones or a part of a zone). To obtain the velocity profile, the multipurpose hose is filled with a dilute, nontoxic, Rhodamine WT dye solution. A brass check valve with a pressure threshold of about 350 pounds per square inch (psi) is attached to the bottom of the multipurpose hose. The check valve keeps dye solution from exiting the hose prematurely. The Rhodamine-filled hose is lowered to the bottom of a pumping well and a pulse of dye solution is injected into the water column using a high-pressure pump at the surface. The pulse travels to the pump intake at the same velocity as water traveling in the well borehole or casing. A small portion of the discharge from the pumping well is routed through a field fluorometer (Turner Designs model 10-AU[1]), which measures dye concentration (in micrograms per liter) at one-second intervals (Figure 14e). When the injected dye pulse is first detected at the surface, the traveltime in seconds is recorded for the given depth. Then the hose is raised 3 to 5 feet and another pulse of dye solution is injected into the well. When the recorded traveltimes are plotted versus depth and combined with ancillary information such as well diameter, the following can be inferred:

1. the depth of the pump intake (minimum traveltime),
2. an estimate of total well yield,
3. changes in water velocity in the well,
4. the approximate depths of contributing intervals,
5. the relative amounts of water produced by these intervals, and
6. the pumping water level.

To interpret the data obtained from a tracer-pulse velocity profile, several assumptions must be made regarding conditions in the pumping well. Velocities (slopes) on the tracer-pulse profile should always increase in the direction of the pump. In addition, laminar pipe flow must be assumed to occur in these wells, but estimates of the dimensionless Reynolds number (Chow, 1959) revealed that laminar flow is disrupted at velocities as small as 0.3 ft/s, and turbulent flow is likely to be present at velocities greater than about 1 ft/s. The velocity resolution of the tracer-pulse technique appears to be about 0.1 ft/s in Norman wells.

Assumptions of the tracer-pulse method:

1. Velocities inside and outside the PVC access tube are equal at a given depth.
2. Velocities (and zonal contribution rates) do not change appreciably with time during testing.
3. No sustained changes occur in well cross-sectional area either above or below the pump intake.

[1] Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. Government

4. Flow in the well is uniform, laminar, and purely vertical.
5. During production, water in the well never flows back out into the formation.
6. Velocity can never decrease in the direction of the pump.
7. Changes in well-bore velocity are proportional to changes in flow rate in the well.

Estimates of Total Well Yield

Tracer-pulse data near the pump intake allow for independent estimates of total well yield, or well-head discharge. This value is necessary to convert estimated zonal flow contribution (in percent) to a zonal discharge (in gallons per minute). Though well-head flowmeters were present at most of the wells in the Norman well field, the flowmeters were installed to read the flow rate only when the well was pumping into the distribution line. All dye-injection tests were performed while the well was pumping to waste (not into the distribution line), therefore, the well yield reported by the well-head flowmeter was not used in any calculations of zonal flow contribution. If the size of the pump column pipe is known, the well yield can be estimated using tracer traveltime data. The depth (in feet) of the minimum recorded traveltime, which marks the pump intake, can be divided by the minimum traveltime (in seconds) to compute the velocity (in feet per second) of water traveling through the pump column to the surface. Well scale thickness, which can reduce the theoretical cross-sectional area inside the pump-column pipe, was not routinely measured but was assumed to be no greater than 0.125 inch -- the maximum scale thickness observed on the outside of any drop pipe. More commonly, well-scale thickness was about 0.1 inch. Estimated total well yields were listed with a confidence interval to reflect uncertainty caused by unknown well-scale thickness. The bounds of the confidence interval were computed as:

$$Q_{min} = v_{col} * \pi(r_{min})^2 * 448.8$$
(with 0.125-inch scale thickness)

$$Q_{max} = v_{col} * \pi(r_{max})^2 * 448.8$$
(with 0.000-inch scale thickness)

where

Q is the well yield in gallons per minute,

v_{col} is the estimated velocity of water in the pump column in feet per second,

r is the inner radius of the column pipe in feet, and

448.8 is the conversion factor from cubic feet per second to gallons per minute.

The difference between the minimum (min) and maximum (max) computed flow rate in Norman wells ranged from about 90 gallons per minute to 40 gallons per minute. The reported yield is the average of Q_{max} and Q_{min}, and the reported confidence interval is equal to one half the difference of Q_{max} and Q_{min}.

Techniques for Determining Flow Contribution by Open Intervals

A graphical approach was used to incrementally determine flow contribution from open (perforated or screened) intervals. This approach was necessary because changes in flow velocities are small compared to noise associated with the tracer-pulse technique. First, the hose depth and raw traveltime data were plotted on a large format graph similar to Figures 16-17. Envelope-fit lines were drawn across sections of the data that expressed linear trends. The slope of these lines (change in depth divided by the change in traveltime) represents the vertical component of water velocity in the well across the given depth interval. Increases in slope (velocity) are proportional to flow contribution by open intervals.

Two examples of traveltime profiles are presented in Figures 16-17. Slope breaks can be identified in Figure 16 by moving a straight edge along the traveltime profile. Changes in slope occur at depths of 430, 450, 500, 530, and 565 feet in data from April 2004 in Well 23 (Figure 16). Slope breaks in Well 15 are more subtle but were identified in the same way (Figure 17). Each change in slope is attributed to the nearest open interval in the direction opposite the pump. The total flow gained over each change in slope was computed using the equation:

$$Q_{gained} = (v_2 - v_1)A$$

where

v_2 is the greater vertical velocity,

v_1 is the lesser vertical velocity, and

A is the cross-sectional area of the well.

For comparison between intervals or zones, flow contribution from each interval or zone is reported as a percentage of the well yield.

The cross-sectional area was assumed to be constant above and below the pump intake, except in Well 23, which had a 5-inch-diameter pump on a 4-inch-diameter pump column. For simplification of calculations, the pump column is assumed to end at the pump intake. In reality, the submersible pump motor extends another 3 to 5 feet below the intake. This simplification is acceptable because the length of the motor is comparable to the method depth resolution of 3 feet. The computation of cross-sectional area below the pump intake (Figure 15) is relatively simple and robust using the equation:

$$A_{below} = \pi(r_{cas}/12)^2$$

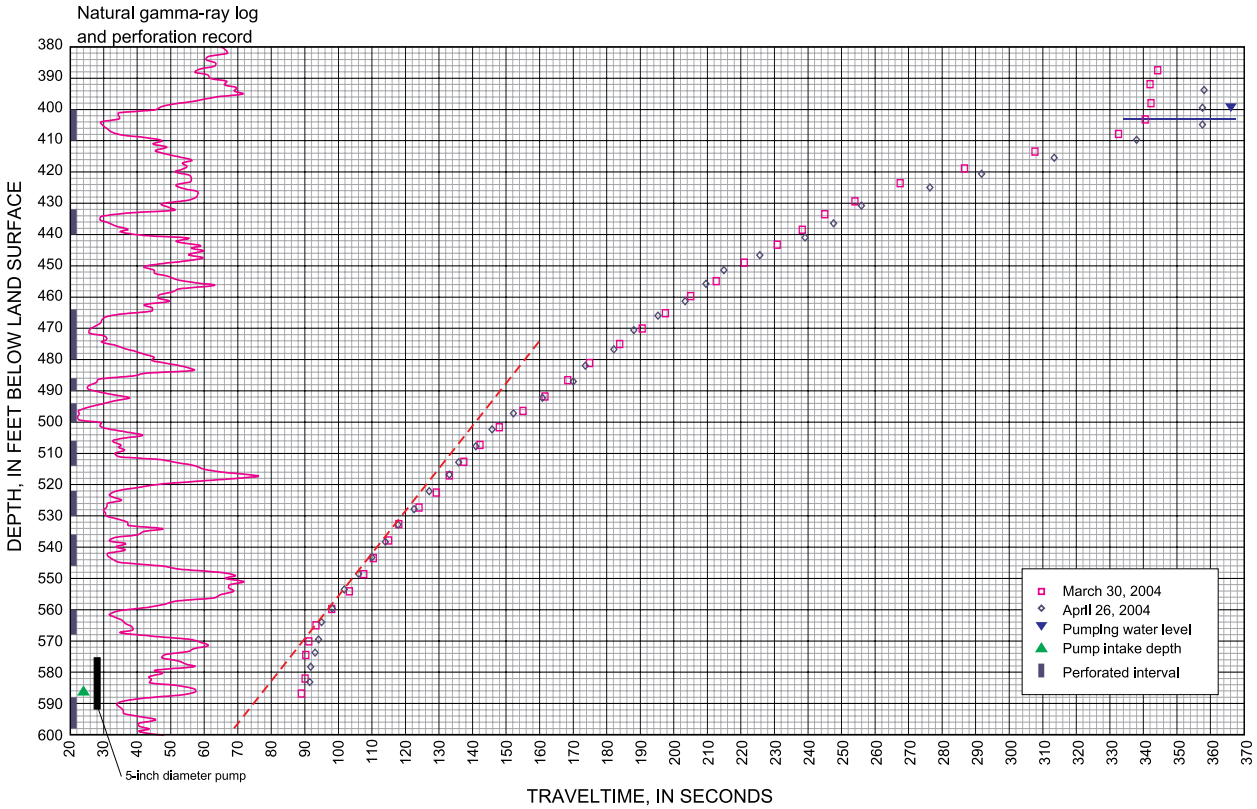


Figure 16. Traveltimes determined by tracer-pulse method in Norman Well 23 in 2004.

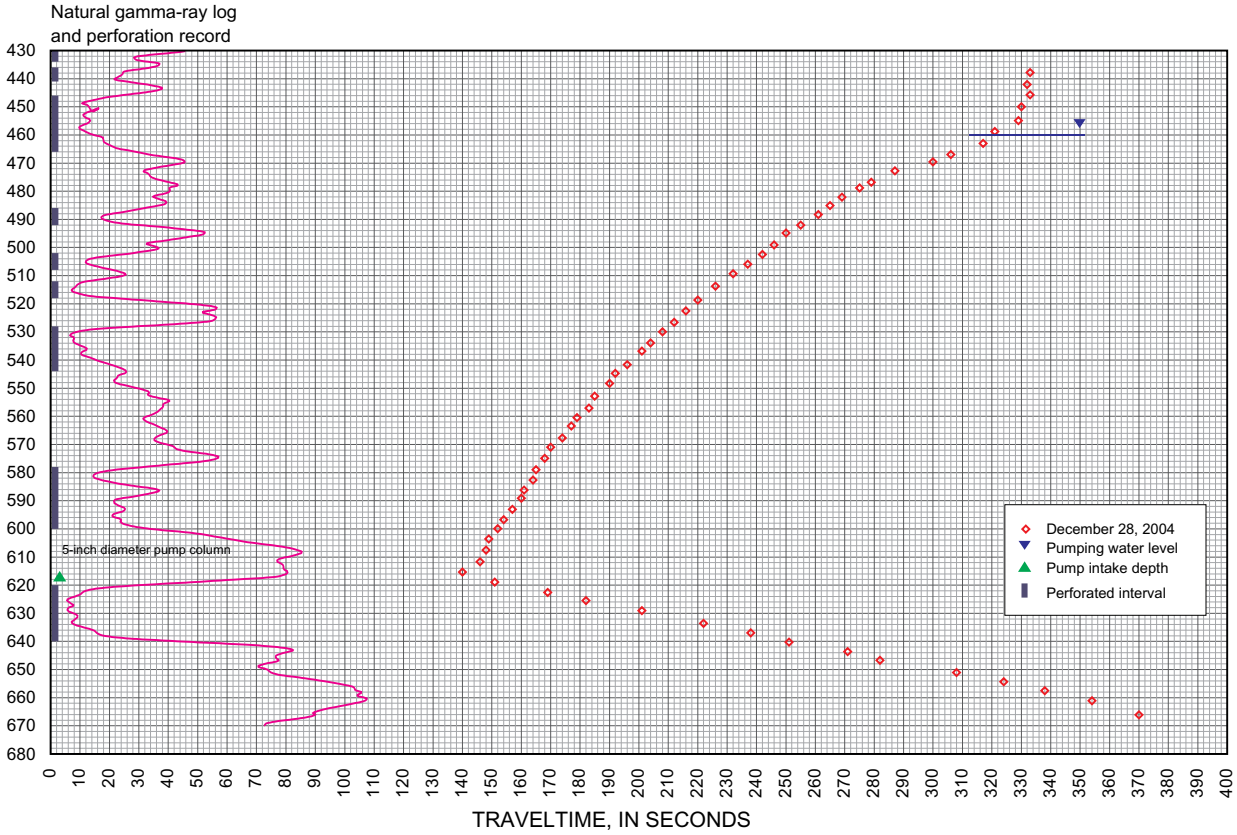


Figure 17. Traveltimes determined by tracer-pulse method in Norman Well 15 in 2004.

where

A_{below} is the cross-sectional area of the well below the pump intake in square feet and

r_{cas} is the inner radius of the casing in inches.

The computation of cross-sectional area above the pump intake is more complex because of the presence of the pump column and attachments (Figure 15). The area above the pump intake can be approximated using the equation:

$$A_{above} = \pi((r_{cas}/12)^2-(r_{col}/12)^2-(r_{cab}/12)^2)$$

where

A_{above} is the cross-sectional area of the well above the pump intake in square feet,

r_{cas} is the inner radius of the casing in inches,

r_{col} is the outer radius of the pump column pipe in inches, and

r_{cab} is the radius of the electrical cable in inches.

When this equation was applied to computations of flow rate, the flow rates were unreasonably high. This discrepancy was probably because the effective cross-sectional area was much less than the true area. The effective cross-sectional area is the remaining area after the theoretical cross-sectional area has been reduced by scale deposits and dead spaces such as eddies.

An empirical approach was used to estimate the effective cross-sectional area above the pump intake in wells with typical construction specifications (4-inch pump column in a 10-inch casing). The effective area was computed using the maximum estimate of well yield, the flow rate below the pump intake, and the tracer-pulse velocity determined just above the pump intake using the equation:

$$A_{above} = (Q_{max} - A_{below} * v_{below})/v_{above}$$

where

A_{above} is the cross-sectional area of the well above the pump,

A_{below} is the cross-sectional area of the well below the pump,

Q_{max} is the maximum estimate of well yield from tracer-pulse data,

v_{below} is the maximum velocity measured below the pump intake, and

v_{above} is the maximum velocity measured above the pump intake.

Calculations of effective area in Norman wells with typical construction (4-inch pump column in a 10-inch casing) ranged from 0.14 to 0.36 square foot, with an average area of 0.21 square foot. The average area of 0.21 square foot was applied to computations for all wells. The theoretical cross-sectional area for a typical Norman well was about 0.42 square foot. The difference between the theoretical and effective cross-sectional areas is likely to diminish as the casing diameter increases, the pump column diameter decreases, or the water velocity decreases.

The average area of 0.21 square foot also was applied to Wells 05 (initial pump depth), 07, 13, 23, 33, and 36 (initial pump depth) in which no traveltime data were collected below the pump. When the flow above the pump exceeded the maximum estimate of well yield, open intervals below the pump were not quantifiable and were labeled (NQ). For some of these calculations, however, the flow above the pump intake was considerably less than the maximum estimate of well yield. In these cases, the difference in flow was assumed to originate from below the pump intake.

Wells 15 and 31 had non-typical construction; Well 15 had a 5-inch pump column and Well 31 had a 12-inch casing. For these wells, the same procedure was used to determine the cross-sectional area above the pump, but the results were not averaged with wells of similar construction. Effective areas computed for Wells 15 and 31 were 0.27 and 0.41 square feet, respectively.

Cross-sectional Area Computations

For wells with traveltime data points above and below the pump intake (see example, Figure 17), computations began with a determination of the proportions of water coming from above and below the pump. The accuracy of these computations depends on the accuracy of computed cross-sectional areas and is, therefore, prone to error. The flow gained over each open interval was computed as a proportion of the total flow coming from above or below the pump. No data points were obtained below the pump in some wells. In these cases, the total flow coming from below the pump could not be determined from velocity measurements. Also, open intervals within a few feet of the pump intake could not be analyzed for flow contribution using the tracer-pulse method. When the pump intake is set in an open interval, water from the interval is pulled into the intake without traveling vertically. When the pump intake is set near an open interval, there may not be enough traveltime data points across the interval to determine a velocity. When flow contribution could not be quantified, the interval was marked NQ. The percentages of flow contribution from other open intervals were determined as if the unquantifiable interval produced no water.

Hose Stretch

Because the multipurpose hose has elastic properties, estimation and compensation for line stretch was necessary. The stretch calibration technique compared an

air-line-determined depth reading with the line-counter reading and computed the difference in terms of percent stretch. The percent stretch was added to each line counter reading to arrive at a corrected depth for the hose intake. The first estimate of stretch was made before the 11-well investigation (Figure 4), when the hose was relatively new. A three percent stretch factor was estimated at that time. As the hose made numerous trips in and out of the wells during preliminary tests, some temporary elastic stretch was converted to permanent inelastic stretch, especially on a few occasions when the hose became stuck in the well. A second estimate of stretch was conducted late in the 11-well investigation and yielded a factor of two percent. A two percent stretch correction factor was applied to all tests and all reported depths are stretch-corrected depths below land surface.

PVC Access Tube

The effect of the access tube on the quality of well-profiler data is unknown because no repeat tests were performed in a well before and after access tube installation. The ability to freely move through the water column certainly increased the precision of depth measurements and, subsequently, water velocities obtained by the tracer-pulse method. A few recorded dye pulses from above the pump in one well, however, were irregular with dual peaks. The separation between peaks increased with distance from the pump intake. These observations indicated that there may be a difference between water velocities inside and outside the PVC access tube.

Depth-dependent Sampling

After contributing intervals are identified, the multipurpose hose is drained and flushed of dye solution, and a Teflon®-lined sample hose with a stainless-steel braided cover is attached to the end of the multipurpose hose (Figure 14d). A stainless-steel check valve with a pressure threshold of 3 psi separates the two hoses and prevents contamination of the sample hose by any residual dye solution in the multipurpose hose. An identical check valve is attached to the bottom of the sample hose (Figure 18). Both hoses are filled with compressed nitrogen gas to a pressure greater than the

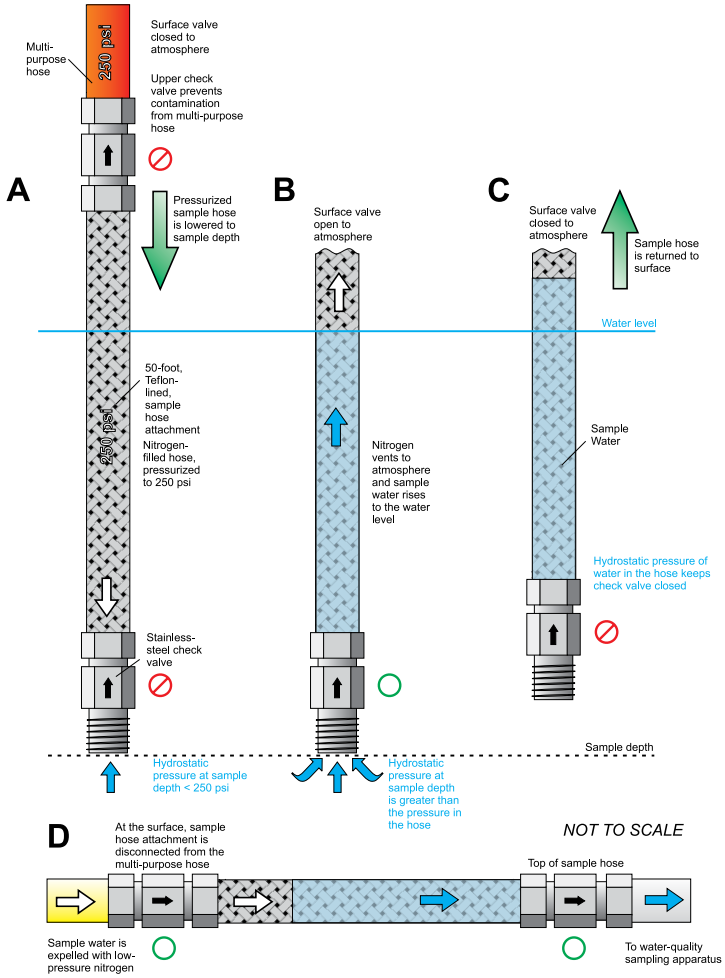


Figure 18. Procedure for depth-dependent sampling with the USGS well profiler. Green, white, and blue arrows indicate movement of the hose, nitrogen pressure, and water pressure, respectively. Green circles indicate open check valves and red crossed circles indicate closed check valves [psi, pounds per square inch].

maximum possible hydrostatic pressure at the bottom of the well (250 psi was used in all Norman wells). Because of the presence of in-line check valves, the system must be pressurized from the bottom of the sample hose. The surface end of the hose is equipped with a pressure gauge for reading the system pressure and a manually controlled release valve for venting the hose. When the system pressure reaches the desired value, the nitrogen pressure source is detached and the pressurized hose is lowered to the desired sample depth (Figure 18a). In cased wells where the depths of openings are known, the sample-hose intake is placed to limit bias by local inputs of water and allow for the most complete mixing of water from contributing intervals.

When the sample depth is reached, samples are collected by opening the manual valve on the surface end of the hose. As the hose depressurizes, the hydrostatic pressure of the water column in the well exceeds the pressure inside the hose (Figure 18b). The in-line check valves open and sample water fills the hose to (about) the pumping water level. When the sample hose stops venting at the surface, the pressures have equilibrated and the hose is full. The manual valve at the surface is closed and the water-filled hose is reeled to the surface (Figure 18c). The pressure of the water column inside the hose is great enough to close the in-line check valves during hose retrieval. Once at the surface, the sample-hose attachment (including check valves) is disconnected from the multipurpose hose and attached to the sampling apparatus (Figure 18d). Compressed nitrogen is used to force the sample water out of the sample hose and into bottles. Though excess sample water may partially fill the multipurpose hose, this water is not suitable for analysis and must be discharged with compressed nitrogen between samples.

Each sample collected with the well profiler represents conditions at a discrete depth in the pumping well, not a specific hydrogeologic zone in the formation. The well profiler sample is a mixture of water from several contributing zones, which can be several feet away from the sample depth. Because the proportion of water produced from each zone was estimated using the tracer-pulse technique, a mass balance approach could be used to estimate constituent concentrations in each zone. However, the depth-dependent water-quality data are more appropriately used to draw qualitative comparisons between zones. This comparison can be performed without knowing how much water is being produced by each zone. For example, the concentration data could be used to determine which zones produce water with elevated concentrations of arsenic and which zones produce water that is relatively free of arsenic contamination. If a sample at depth A has an arsenic concentration of 3 µg/L, and an adjacent sample (in the direction of flow) at depth B has an arsenic concentration of 20 µg/L, there is likely a zone between sample depth A and B that is contributing water with an arsenic concentration greater than 20 µg/L. The likelihood of identifying a single contaminated zone depends on the spacing and locations of depth-dependent samples collected relative to the spacing and locations of producing zones.

The sample-hose attachment used has an inner diameter of 13/32 inch and is 50 feet in length. These dimensions contain a storage volume of about 0.33 gallon (1.25 liters). Because the total sample volume is limited, every effort was made to conserve water and fill bottles in a timely, organized manner. After a field rinse of the sampling apparatus, field measurements of water properties were collected, followed by collection of unfiltered samples and then filtered samples (Table 3). The more sensitive filtered samples were collected last because the last water to enter the sample-hose attachment during sample collection is the last water to be expelled from the sample-hose attachment during bottling. This water has a lesser chance of contamination from the multipurpose hose because any contamination from the multipurpose hose should be flushed from the apparatus by previous steps in the sampling process.

Table 3. Summary of sample water use and analysis types in order of collection

Volume (milli-liters)	Bottle type	Analysis order	Preserva-tive	Filtra-tion
200	--	field water properties*	--	no**
250	--	rinse water	--	no
250	polyethylene	alkalinity	--	no
40	amber glass	major anions	--	no
40	amber glass	nutrients	H_2SO_4	no
60	glass	isotopic ratios	--	no
40	amber glass	carbon	--	yes
40	amber glass	arsenic speciation	--	yes
40	amber glass	metals	HNO_3	yes
960	Sample hose volume = 1,250 milliliters			

* includes specific conductance, pH, water temperature, turbidity, dissolved oxygen concentration, and iron speciation.

**Only samples for iron speciation were filtered; all other field measurements used unfiltered water.

Decontamination Procedures

Laboratory decontamination of sampling equipment was performed using USGS standard methods (Wilde et al., 1998). The same procedures were applied to sample hoses and fittings with one exception. The sample-hose attachment is Teflon-lined and would normally be rinsed with a five percent hydrochloric-acid solution. This step was not applied during the 11-well investigation of water-quality changes with depth (Figure 4) because the rinse would damage the permanently attached stainless-steel fittings. For the two-well investigation of arsenic remediation techniques (Figure 4), the laboratory decontamination for the sample-hose attachment did include a hydrochloric acid rinse that bypassed the stainless-steel fittings. In the field, the sample hose was evacuated with compressed nitrogen between each sample until no water was visible exiting the hose. The entire sampling apparatus was cleaned in the laboratory before moving to another well.

Quality-assurance Procedures

Quality assurance was evaluated using blanks (4), replicates (7), and laboratory duplicates (84). Replicate and duplicate analysis values are listed after each paired environmental sample in the appendix tables. Replicates and laboratory duplicates were mostly consistent with paired environmental samples. About 78 percent of all replicate measurements had relative percent differences less than 5 percent. Replicate types included repeat sampling, in which new sample water was collected from the same depth, and repeat bottling, in which two bottles were filled from the same parent sample volume. The repeat bottling replicate data showed that contamination from the multipurpose hose (which is attached to only one end of the sample hose) was not a problem. The repeat sampling replicate data showed that depth-dependent sampling was repeatable in Norman wells. Major-ion concentrations were consistent in replicate data, but some trace-element concentrations occasionally showed large variation in replicate data. Aluminum, barium, boron, and titanium each had multiple replicate values with relative percent differences greater than ten percent. Among measurements of field water properties, turbidity and field iron speciation each had multiple replicate values with relative percent differences greater than ten percent. Laboratory duplicates, which were created by splitting the environmental samples, were part of the standard procedures of the analyzing laboratory. These duplicates were used to assess repeatability and precision of laboratory methods. No problems were identified in visual comparisons of environmental sample concentrations and laboratory duplicate concentrations.

Several constituents (mostly trace elements) were detected in two field blanks, but concentrations were usually much less than the concentrations detected in typical environmental samples. Nitrate, barium, boron, strontium, and dissolved organic carbon were detected in blanks at concentrations comparable to or exceeding concentrations detected in environmental samples. The contamination is most likely the result of a documented change in the bottling procedures used by the blank water manufacturer. Low-level detections were documented in OmniSolv organic blank water beginning with lot number 44328 (USGS National Water Quality Laboratory lot number 80501), which was used in this study. This lot number and water type was investigated by processing source-water blanks and was discontinued from use as an inorganic blank water in June 2006, after all blanks in this study had been collected (USGS, Office of Water Quality, written commun., 2006). Additional contamination may have resulted from incomplete cleaning of the sample-hose attachment, especially in the case of carbon, which sometimes was detected in greatest concentration in the first sample collected after each laboratory cleaning. Therefore, this particular constituent may have been introduced during the cleaning process or blank sampling process and may be routinely overestimated in the analyses.

Sample Analysis Methods

Specific conductance, pH, water temperature, and dissolved oxygen concentration were determined by USGS staff in the field using methods described in Wilde and Radtke (1998). These water properties were measured with a calibrated YSI 600XL multiparameter probe and, for measurements made at the well-head discharge, using a flow-through cell. For depth-dependent samples collected with the USGS well profiler, water temperature and dissolved oxygen were recorded but were not reported because, in a bailed sample, these properties quickly became unrepresentative of the water in the aquifer. Turbidity was determined with a Hach 2100P turbidimeter. Acid neutralizing capacity, bicarbonate and carbonate concentrations were determined, usually in the lab, using an inflection point titration method (Rounds and Wilde, 2001). For most samples, the titrations were completed within 24 hours of sample collection. Analysis of dissolved iron species was performed in the field using a Hach DR/2000 field spectrophotometer. Ferrous iron concentration was determined using Phenanthroline reagent (Hach method 8146) and total iron concentration was determined using Ferrover reagent (Hach method 8008). Ferric iron concentration was not reported but can be determined by subtracting the ferrous iron concentration from the total iron concentration. Stable isotopic ratios δ^2H and $\delta^{18}O$ of water were analyzed by the USGS Stable Isotope Laboratory in Reston, Virginia (Coplen et al., 1991; Epstein and Mayeda, 1953; Coplen, 1994). One set of arsenic samples collected at Well 36 from 0900 to 0902 on January 18, 2006, was analyzed

at the USGS National Water Quality Laboratory in Lakewood, Colorado, using Inductively Coupled Plasma - Mass Spectrometry (ICP-MS; Garbarino, 1999). The remaining samples were analyzed at the EPA Robert S. Kerr Laboratory facility in Ada, Oklahoma. Major anions were analyzed using capillary ion electrophoresis with indirect UV detection (Waters Quanta 4000 Capillary Ion Analyzer). Iodine was analyzed using Lachat flow injection analysis (FIA). Nitrate, nitrite, ammonia, and orthophosphate also were analyzed using FIA. Trace elements and major cations were analyzed by Inductively Coupled Plasma - Optical Emission Spectrometry (ICP-OES) using a Perkin Elmer Optima 3300 DV system. Arsenic speciation was accomplished using Ion Chromatography - Hydride Generation - Atomic Fluorescence Spectrometry (IC-HG-AFS). The method was adapted from Slejkovec et al. (1998). Total dissolved arsenic was analyzed using ICP-OES or ICP-MS. Samples collected prior to May 25, 2005, were analyzed for arsenic by ICP-OES, and samples collected on or after May 25, 2005, were analyzed for arsenic by ICP-MS (Creed et al., 1994). For arsenic samples collected prior to May 25, 2005, determination of arsenic V by IC-HG-AFS was used to represent dissolved arsenic concentration in this report because the method detection limit was lower. Total carbon, total organic carbon, dissolved carbon, and dissolved organic carbon were analyzed using methods adapted from the EPA (Methods 415.1, 415.2 and 5310C, Standard Methods for the Examination of Water and Wastewater [17th Edition]).

Environmental samples were analyzed at different times, by different methods, and by different laboratories. As a result, the thresholds or limits used for reporting and censoring data changed over the duration of this study. The method detection limit (EPA, 1999) for a particular analysis method is the "minimum concentration of a substance that can be measured and reported with 99 percent confidence that the true value is greater than zero." The practical quantitation limit (EPA, 1999) is the "lowest concentration of an analyte that can be reliably measured within specified limits of precision and accuracy." In this report, some analysis values are reported as less than (<) the practical quantitation limit; these values are considered to be nondetections. Other values are reported as estimated (E) at a value between the practical quantitation limit and the method detection limit. For estimated values there is confidence that the constituent concentration is greater than zero, but there is low confidence in the concentration value. Estimated values are considered to be detections in this report.

Filtered (dissolved) and Unfiltered (total) Samples

Regulatory compliance samples, which comprise most of the data presented in Figure 10, are collected and reported as total (unfiltered) arsenic. In contrast, samples from this study are reported as dissolved (0.45-µm filtered) arsenic. Focazio et al. (2000) have documented large differences between filtered and unfiltered ground-water samples. Some initial concern existed that much of the arsenic in Norman water samples, if sorbed onto small particulates, would be removed from the samples during the filtering process, and filtered and unfiltered samples could yield different arsenic concentrations. Sample turbidity was less than one Nephelometric Turbidity Unit (NTU) for most well-head samples collected in this study, so differences in arsenic concentration in filtered and unfiltered samples were expected to be small. There also was concern that different laboratories could yield different results from the same sample water. To examine these possibilities, two sets of well-head samples were collected at Norman Well 36 on January 18, 2006. These sample sets were sent to Shaw Environmental Laboratories in Ada, Oklahoma, and the USGS National Water Quality Laboratory (NWQL) in Lakewood, Colorado, for arsenic analysis by the ICP-MS method. Each sample set included an unfiltered sample, a 0.45-µm filtered sample, and a 0.20-µm filtered sample. The arsenic concentrations in the Shaw Environmental Laboratory samples were 20.6 (with a laboratory duplicate of 20.3), 20.4, and 20.4, respectively. The arsenic concentrations in the NWQL samples were 17.3, 17.7, and 17.9, respectively. Differences in arsenic concentrations in unfiltered, 0.45-µm filtered, and 0.20-µm filtered well-head samples in this study were within the precision of the ICP-MS technique (2-3 percent; Creed et al., 1994; EPA, 1999). Also, based on differences in arsenic concentrations between the two laboratories, relative percent differences in arsenic concentrations determined by different laboratories may be greater than 17 percent. Though this difference is small relative to some environmental concentrations, any difference attributable to different analyzing laboratories is unsettling to water-supply systems that operate wells with arsenic concentrations near the MCL.

5.0
Norman Arsenic Test Hole

Schlottmann et al. (1998) found that aqueous concentrations of arsenic (and the geochemically related oxyanions of chromium, selenium, and uranium) were elevated in only one sampled sandstone zone at depth in the Central Oklahoma aquifer near Edmond (Figure 1). As part of the investigation of changes in water quality with depth in the Norman area (southern Central Oklahoma aquifer), an undeveloped site in Norman was selected for drilling, logging, coring, and water sampling in a test hole similar to that of Schlottman et al. (1998). The selected test-hole site (SE1/4, Section 5, Township 09 North, Range 02 West) was in northern Norman near the Little River (Figure 2). The USGS Central Region Drilling Unit, using mud-rotary methods, drilled to a total depth of 728 feet. Caliper, natural gamma-ray, resistivity, and neutron logs (Keys, 1990) were recorded and used to identify major water-bearing sandstones from depths of about 300 to 728 feet (Plate 1). Seven sandstone-dominated units or zones, ranging from 12 feet to 40 feet in thickness, were selected for coring and water sampling (Plate 1, Figure 19). When logging was complete, this reconnaissance hole was plugged, and a second test hole, for collection of core and water samples, was drilled about 20 feet to the west of the logged hole.

Discontinuous core collection and a single-packer sampling method were used to collect rock and water samples from selected sandstones. First, the hole was drilled to about 300 feet. The coring bit was inserted and core samples were collected from 302 feet to the bottom of the first water-quality sample zone (Zone 1, about 350 feet, Plate 1, Figure 19). The hole was then reamed to 6 inches and air developed for several hours, until the water reaching the surface cleared. A single inflatable packer was then installed on 3-inch pipe at the top of the water-quality sample zone (about 320 feet) and inflated. A low-capacity pump was installed on 2-inch, stainless-steel drop pipe inside the packer string at a depth of about 310 feet. The selected sample zone was purged for 1 to 2 hours at 5 gallons per minute, until the discharge was relatively clear (turbidity less than 500 NTU) and field water properties stabilized. The test-hole water sample was then bottled, preserved, and analyzed in the same manner as samples collected from public-supply well heads. After the sample for water-quality Zone 1 was collected, the water-quality sampling apparatus was removed from the well and coring resumed. Rock material was sampled by coring from 302 to 536, 568 to 598, 615 to 636, 640 to 652, and 668 to 686 feet (Plate 1).

Norman Arsenic Test-hole Core

Core recovery in the Norman arsenic test hole was good considering the rock properties of the Garber-Wellington (mudrock that slacks upon wetting). Some parts of the core were rubblized by the coring process. One example was observed in the images of core boxes 5 and 6 at a depth of 375 feet where rounded clasts of sandstone set in a disrupted mudstone matrix occurred above an underlying well-cemented conglomerate (Appendix 1).

Most of the red sandstones in the core were made up of very-fine to fine-grained sand that was moderately well to well sorted with respect to framework grains (Appendix 2). The sandstones in the core were friable and small fragments broken from the core could be crushed with the fingers. The sandstones also contained red mud (matrix composed of clay and silt-sized mineral matter) between the framework grains. The sandstones were thin to moderately bedded. The upper and lower bounding surfaces of sandstone beds were straight (or relatively horizontal). Some of the lower-bounding surfaces were curved and indicative of erosion of underlying units in the Permian. Internal to the sandstones, some of the beds contained horizontal laminations and ripple laminations. Irregular or curved internal bounding surfaces are present in some beds. Other features included planar and trough cross-bedding. Evidence for trough cross-bedding was that the angle of cross-bedding inclination increased upward in individual cross-bedded units. Some of the sandstone beds contained small spherical carbonate concretions, which were thought to be diagenetic.

Thin to moderately bedded layers of conglomerate were present in the core (Appendix 2). The conglomerates tended to be massively bedded (structureless) and contained clasts of dolomite and mudstone. The dolomite clasts appeared to be a product of Permian soil forming processes that were liberated from the mudstone during Permian erosion. Most of the mud clasts in the conglomerates were derived from the underlying mudstones as well (mudstone rip-up clasts). The conglomerates were well-cemented by dolomite and could be broken only through hard impacts with a heavy hammer.

Some of the sandstone beds in the core were above thin conglomerate zones. Collectively, these zones fined upward as expected for units associated with a fluvial depositional setting (Visher, 1965, Reading, 1987).

Most of the silt- and clay-rich layers (mudrock) in the core were red mudstone (mudrock without

laminations). The mudstones were relatively massive (or structureless). The mudstone slacked when immersed in water. Locally, the mudstone contained features that appeared to be paleo-root traces, rhizocretions (carbonate accumulations associated with ancient roots), irregular-shaped carbonate nodules, some weak horizonation, and local chemical reduction of the red mudstone (or locally, sandstone) to a greenish color (redoximorphic processes). These features were indicative of occurrence of soil forming episodes during the Permian period (Permian paleosol formation).

Evidence for secondary iron mobilization (hematite based on visual inspection) was present in the sandstone, conglomerate, and mudrock preserved in the core (Appendix 2). The iron occurred as clots and as crenulated crusts. The clots were concretion-like and occurred in some of the sandstone and mudrock. The crusts appeared similar to, or were compatible with, the movement of a local chemical oxidation front through the Garber-Wellington sequence (similar in appearance to liesegang banding, p. 123, Pettijohn et al., 1987). The crusts occurred in the sandstone and mudrock. Iron mobilization in the mudstones of the core was dramatic. Iron would not be expected to mobilize out of mudrock late in the burial history of the unit (that is, iron mobilization in the mudrock that is occurring in the present-day time frame) because the volume of water in the mudrock today is low relative to the volume of water moving through the aquifer sandstone on a daily basis. Consequently, present-day iron mobilization would be expected to occur in sandstone rather than in mudstone. Textural evidence in core indicates that the iron in the mudrock was probably mobilized during the early phase of burial (water squeezed from the mud by mechanical compaction soon after deposition and shallow burial during the Permian period).

Norman Arsenic Test-hole Water Quality

Interpretation of the geophysical logs in Plate 1 indicates that the test hole penetrated about 50 feet of the Hennessey Group, and nearly half of the total thickness of the Garber Sandstone and Wellington Formation (Figure 19). According to elevation-contour maps of the base of the Hennessey Formation and base of the Wellington Formation (Christenson et al., 1992), the combined thickness of the Garber Sandstone and Wellington Formation is about 1,500 feet at the test hole location.

Ground-water-quality samples were analyzed from seven predominantly sandstone zones. The zones were labeled in order of increasing depth below land surface as Zone 1 (320-350 feet), Zone 2 (416-456 feet), Zone 3

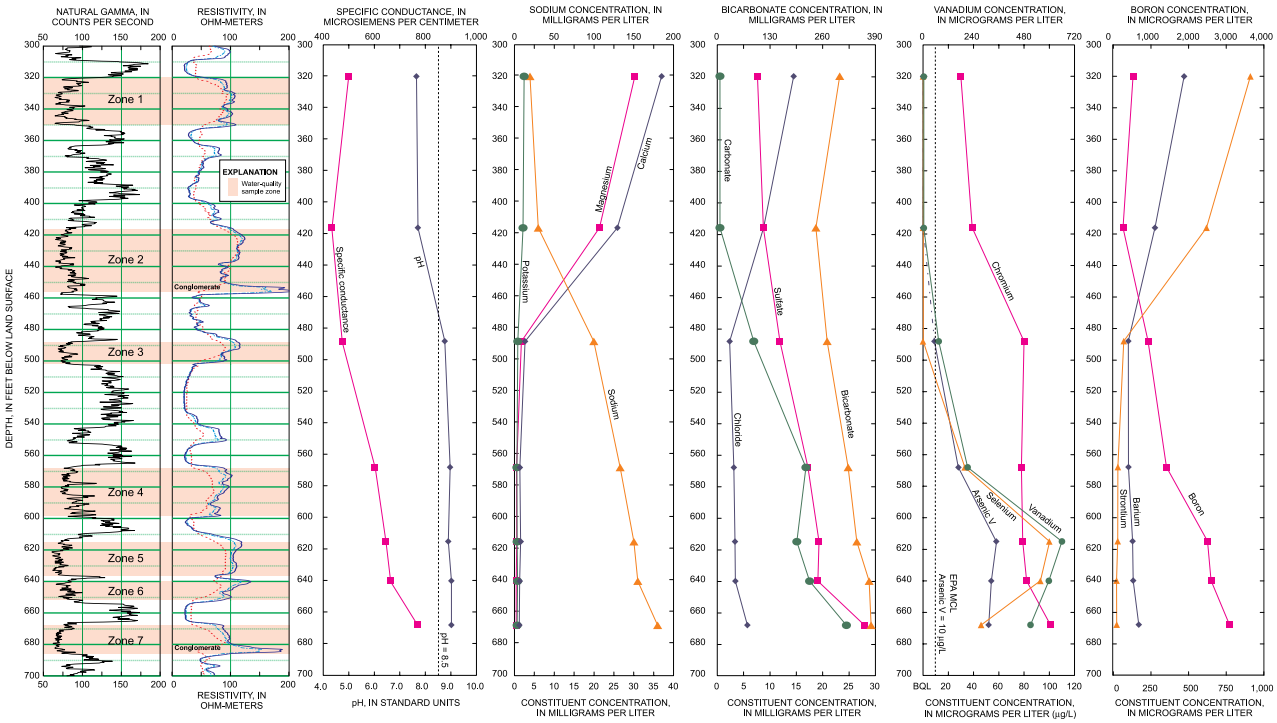


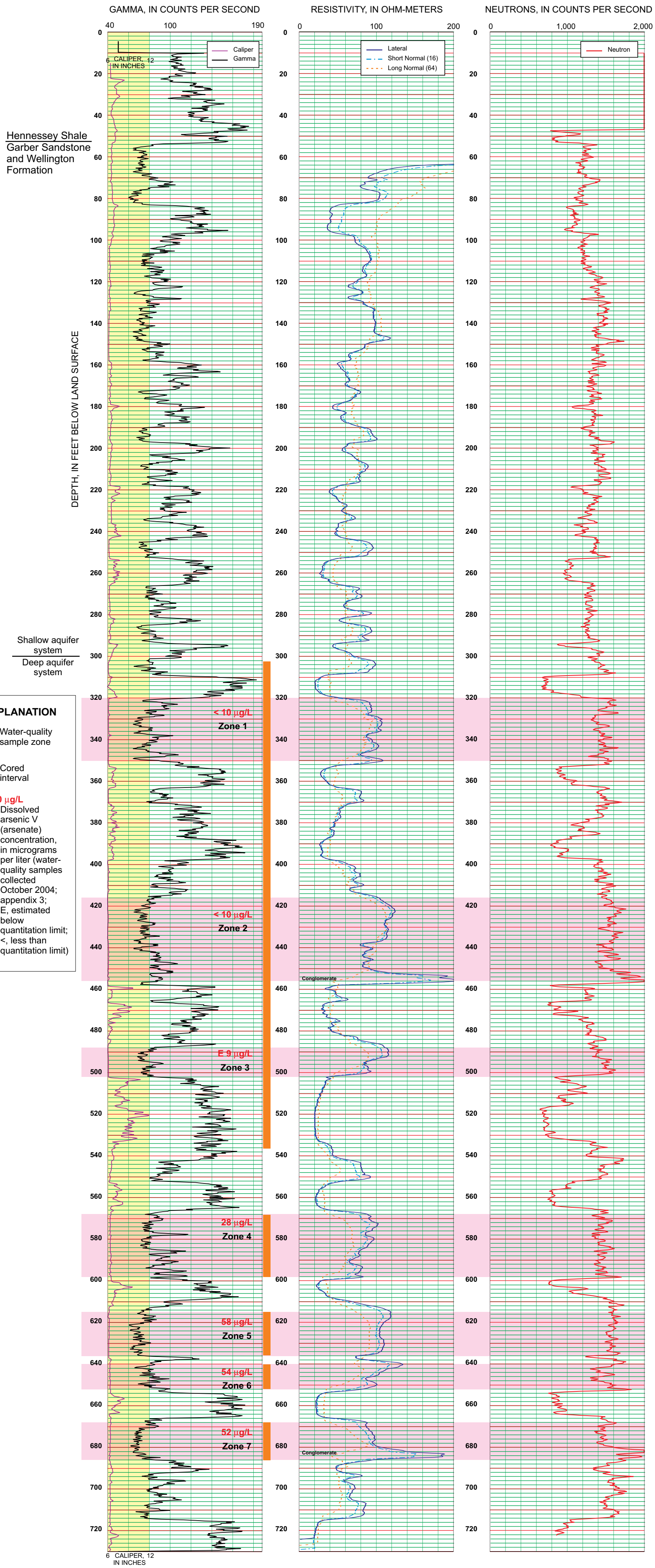
Figure 19. Selected geophysical logs and water-quality samples from the deep aquifer system at the Norman arsenic test hole, October 2004 [µg/L, micrograms per liter; BQL, below practical quantitation limit; EPA MCL, Environmental Protection Agency maximum contaminant level for arsenic].

(488-502 feet), Zone 4 (568-598 feet), Zone 5 (615-636 feet), Zone 6 (640-652 feet), and Zone 7 (668-686 feet) (Figure 19, Plate 1, Appendix 3).

Sandstone Zones 2 and 3 were separated by only 32 feet but had different water types (Figure 19). In the sampled Zones 1 and 2, the water type was calcium-magnesium bicarbonate with a pH about 8.0 and specific conductance less than 500 microsiemens per centimeter (µS/cm) (Figure 19). Dissolved arsenic concentrations were less than 10 µg/L in the two shallowest sampled zones, Zones 1-2 (Figure 19). In sampled Zones 3 through 7, the water type was sodium bicarbonate with a pH about 9.0 and specific conductance usually greater than 500 µS/cm. Arsenic concentrations were considerably greater in Zones 3 through 7 (as compared to Zones 1 and 2), ranging from estimated 9 µg/L to nearly 60 µg/L. Arsenic concentrations exceeded the MCL of 10 µg/L in all zones where pH was greater than 8.5 and specific conductance was greater than 600 µS/cm. Selenium and chromium concentrations mostly were greater in the sampled Zones 3 through 7 than in the sampled Zones 1 and 2. Selenium concentrations exceeded the MCL of 50 µg/L in Zones 5 and 6, and chromium concentration exceeded the MCL of 100 µg/L in Zone 7. Also, concentrations of carbonate, sulfate, fluoride, orthophosphate, boron, and vanadium were greater in Zones 3 through 7 than in Zones 1 and 2.

The contrast in water quality between Zones 2 and 3 could be an indication that the intervening mudstone is a regional barrier to the vertical flow of ground water and, therefore, the flow paths supplying water to Zone 2 are much different than the flow paths supplying water to Zone 3. Alternatively, the contrast could mean that zones below a depth of 460 feet are richer in exchangeable clays. The contrast in water quality between Zones 2 and 3 also could be related to the presence of a basal carbonate-clast conglomerate (represented by a zone of increased resistivity, Figure 19 and plate 1) in Zone 2. Conglomerates in the Central Oklahoma aquifer contain large concentrations of dolomite, arsenic, and iron in the solid phase relative to the other lithofacies in the aquifer system (S.T. Paxton, Oklahoma State University, written commun., 2005). If dolomite supply or arsenic availability are limiting factors in the chemical evolution of water at this depth, the presence of the conglomerate could accelerate arsenic release.

Plate 1. Caliper, natural gamma-ray, resistivity, and neutron logs from the Norman arsenic test hole, SE1/4, Section 5, Township 09 North, Range 02 West.


GAMMA, IN COUNTS PER SECOND
40
100
190
Caliper
Gamma
6 CALIPER, 12 IN INCHES
RESISTIVITY, IN OHM-METERS
0
100
200
Lateral
Short Normal (16)
Long Normal (64)
NEUTRONS, IN COUNTS PER SECOND
0
1,000
2,000
Neutron
DEPTH, IN FEET BELOW LAND SURFACE
Hennessey Shale
Garber Sandstone and Wellington Formation
Shallow aquifer system
Deep aquifer system
< 10 µg/L
Zone 1
< 10 µg/L
Zone 2
E 9 µg/L
Zone 3
28 µg/L
Zone 4
58 µg/L
Zone 5
54 µg/L
Zone 6
52 µg/L
Zone 7
Conglomerate
Conglomerate
EXPLANATION
Water-quality sample zone
Cored interval
< 10 µg/L Dissolved arsenic V (arsenate) concentration, in micrograms per liter (water-quality samples collected October 2004; appendix 3; E, estimated below quantitation limit; <, less than quantitation limit)

6.0
Flow Contribution and Water Quality with Depth from an Eleven-well Investigation

Ground-water Flow and Particle-tracking Models

The ground-water flow and particle-tracking models of Parkhurst et al. (1996) were used to project theoretical flow paths from perforated zones in Norman wells back to likely recharge source areas. The theoretical flow paths are useful to visualize the movement of ground water and to explain general differences in water chemistry between wells.

The ground-water flow model was designed to simulate the flow system in the entire Central Oklahoma aquifer as described in the late 1980s. The ground-water flow model used to simulate the aquifer was the USGS modular ground-water flow model (McDonald and Harbaugh, 1988), now commonly called MODFLOW. This model uses a block-centered, finite-difference approach to simulate flow in three dimensions. The Central Oklahoma aquifer simulations contained 40 columns, 60 rows, and, because vertical flow is substantial, 12 layers. In the horizontal dimensions, cells are 6,562 feet (2,000 meters) on a side, and cell spacing is constant for the region of simulation. All layers are 100 feet thick. Unlike many flow models, the Central Oklahoma aquifer model layers do not correspond to particular geologic units. Instead, the layers are horizontal, and each cell is assigned properties that represent the geohydrologic unit that is the thickest within the cell.

Parkhurst et al. (1996) used a particle-tracking model in conjunction with the ground-water flow model to calibrate the flow model, assist in visualizing flowpaths in the flow system, and integrate the results of the flow model simulations with the analysis of the geochemistry of the Central Oklahoma aquifer. The particle-tracking model generates pathlines, which are the paths of hypothetical particles of water moving through the aquifer as simulated by the numerical flow model. The pathlines correspond to flowpaths in the Central Oklahoma aquifer. The particle-tracking model, MODPATH, was developed by Pollock (1989) and designed to be used with the MODFLOW ground-water flow model. In general terms, MODPATH takes the cell-by-cell flow terms (volumetric fluxes) computed by the flow model and computes the pathlines through each model cell. MODPATH assumes that each directional velocity component varies linearly with each grid cell. Particles can be placed anywhere in the model flow field and tracked forward or backward, and traveltimes can be computed.

The Central Oklahoma aquifer was modeled only as a steady-state system without withdrawals because at the time the Central Oklahoma aquifer model was developed (in the late 1980s), MODPATH only simulated steady-state flow. Parkhurst et al. (1996) stated that because the 1986-87 water table they simulated did not show substantial effects from withdrawals, simulation of transient conditions was not necessary.

The current investigation uses the same ground-water flow field generated by the Parkhurst et al. (1996) MODFLOW simulation, but the current MODPATH particle-tracking model is different. Calibrated MODFLOW model parameters from Parkhurst et al. (1996) were not changed for simulations done for this report. The MODPATH particles for the current analysis were generated at the locations of individual City of Norman supply wells. Particles were generated along the well bore at 5-foot intervals between the highest and lowest known perforations. If the top of the perforations was unknown, perforations were assumed to start 300 feet below the land surface. If the bottom of perforations was not known, perforations were assumed to continue to the bottom of the well. Particles were tracked backwards to the recharge locations.

For all wells in the Norman well field, the dominant modeled source area is south-central Oklahoma County immediately north of Lake Stanley Draper (Figure 20). This source area corresponds to the potentiometric high mapped by Christenson et al. (1992), and is the part of the aquifer that contains the thickest sequences of sandstone (Parkhurst et al., 1996). Only flow paths for wells 05, 07, 20, 23, 33, and 36 are shown in Figure 20 for clarity. Flow path sets for other wells are similar to those shown and can be visually interpolated on Figure 20. The modeled flow paths for each Norman well mostly travel southwest from the source area into the confined part of the aquifer, turn south within the city, and finally arc back to the east before terminating at the well (Figure 20).

For Well 23, which has unusually high concentrations of arsenic in produced water, the majority of modeled flow paths begin near the potentiometric high north of Lake Stanley Draper (Figure 20). The flow paths

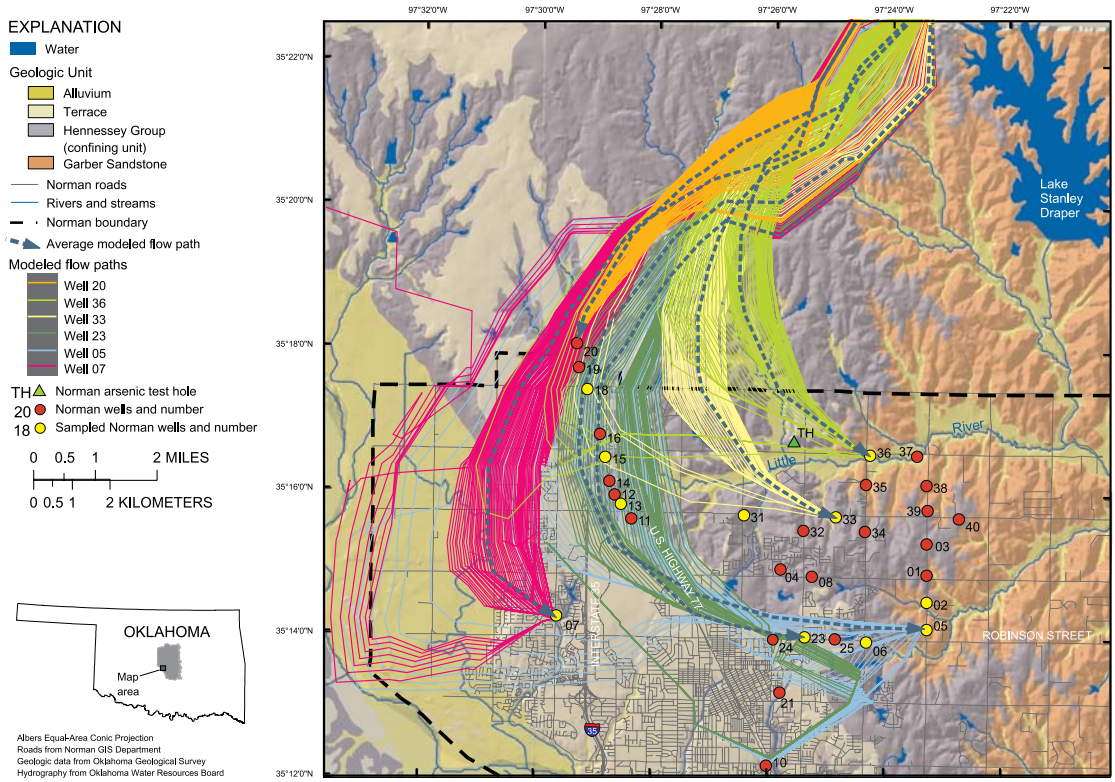


Figure 20. Modeled flow paths for selected wells in the Norman well field. Dashed blue lines represent a typical or average modeled flow path to each selected well.

trend southwest, descending under the confining unit and progressing into Cleveland County, where the flow paths turn to the south near U.S. Highway 77 and then to the east near Robinson Street (Figure 20). If a typical flow path is selected to represent flow to Well 23, this path passes within one mile of several wells in the Norman well field before reaching Well 23 (Figure 20). When plotted in order from most proximal to most distal (from the recharge area), the historical arsenic data for these wells express mostly upward trends in minimum, median, and maximum detected arsenic concentrations with distance along the flow path (Figure 21), validating the aquifer conceptual model. Median arsenic concentration ranges from 3.9 µg/L in Well 20 to 231 µg/L in abandoned Well 24 (Figure 21). In the four wells east of Well 24 (23, 25, 06, and 05), this trend is reversed, probably because the aquifer materials quickly become coarser-grained, more conductive, and better flushed to the east (Figure 21).

The MODFLOW simulation of Parkhurst et al. (1996) was designed to simulate the ground-water flow at an aquifer scale. The model grid was relatively coarse (2,000 meters by 2,000 meters) because of limitations of computer memory at the time the simulation was performed, and the model simulates an assumed steady-state condition not reflective of changes due to ground-water withdrawals. Thus, the pathlines generated by the MODFLOW simulation are indicative of the general direction of ground-water flow but should not be considered to exactly represent the path of ground-water movement.

Traditional Methods of Hydraulic Testing – Well 23

In preparation for hydraulic testing in Well 23, a video log was run to inspect the integrity of the casing, examine the degree of scale buildup, and verify the perforated depths listed on the original perforation record. Perforated depths on the perforation log were about 1 to 2 percent less than the depths noted on the video log (Figure 22). Differences in the recorded depths are most likely caused by differences in line-counter calibration. Errors of this type can result in the accidental perforation of low-permeability zones at greater depths. Consequently, some zones identified as perforated may not produce substantial quantities of water. Also, the video log showed several sets of perforations that were completely plugged by scale buildup. This is another scenario in which a zone could be perforated but nonproducing. Water was observed flowing into the well casing at several depths, including cascading water at 296 and 343 feet and jetting water at several depths. At three depths (479, 506, and 569 feet), water was observed flowing from the well into the aquifer rock.

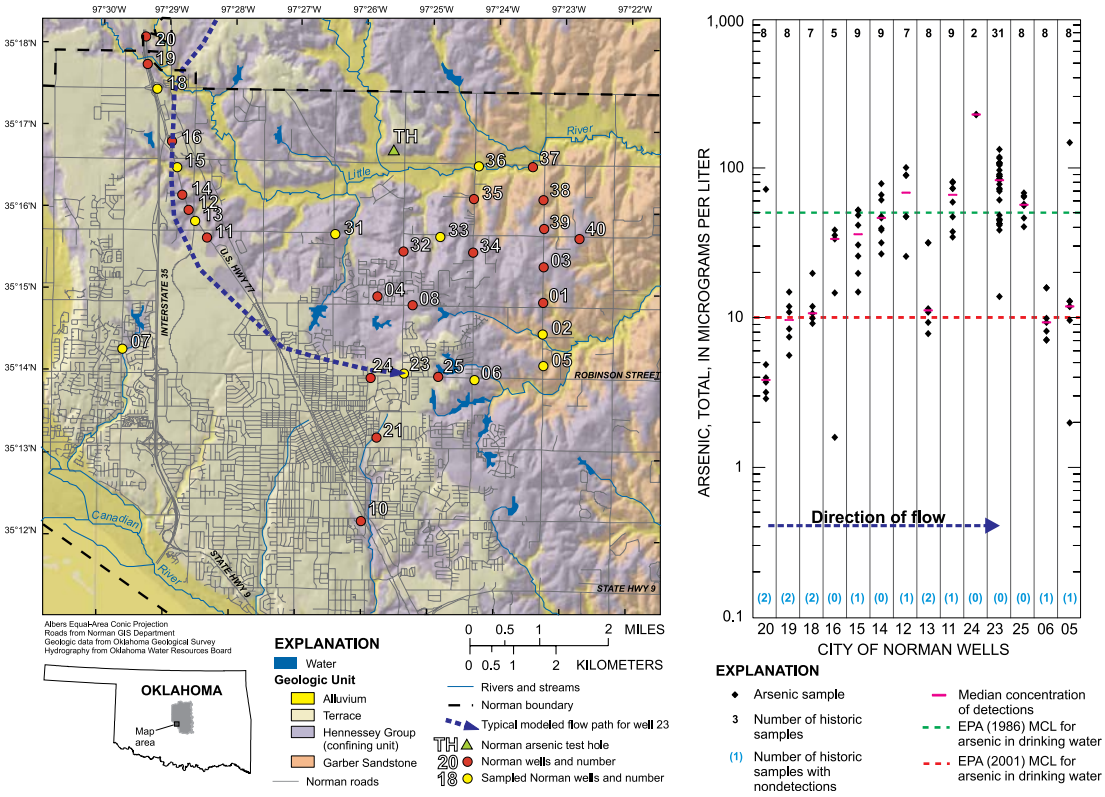


Figure 21. Typical modeled flow path for Norman Well 23 and trend in well-head arsenic concentrations [MCL, Maximum Contaminant Level].

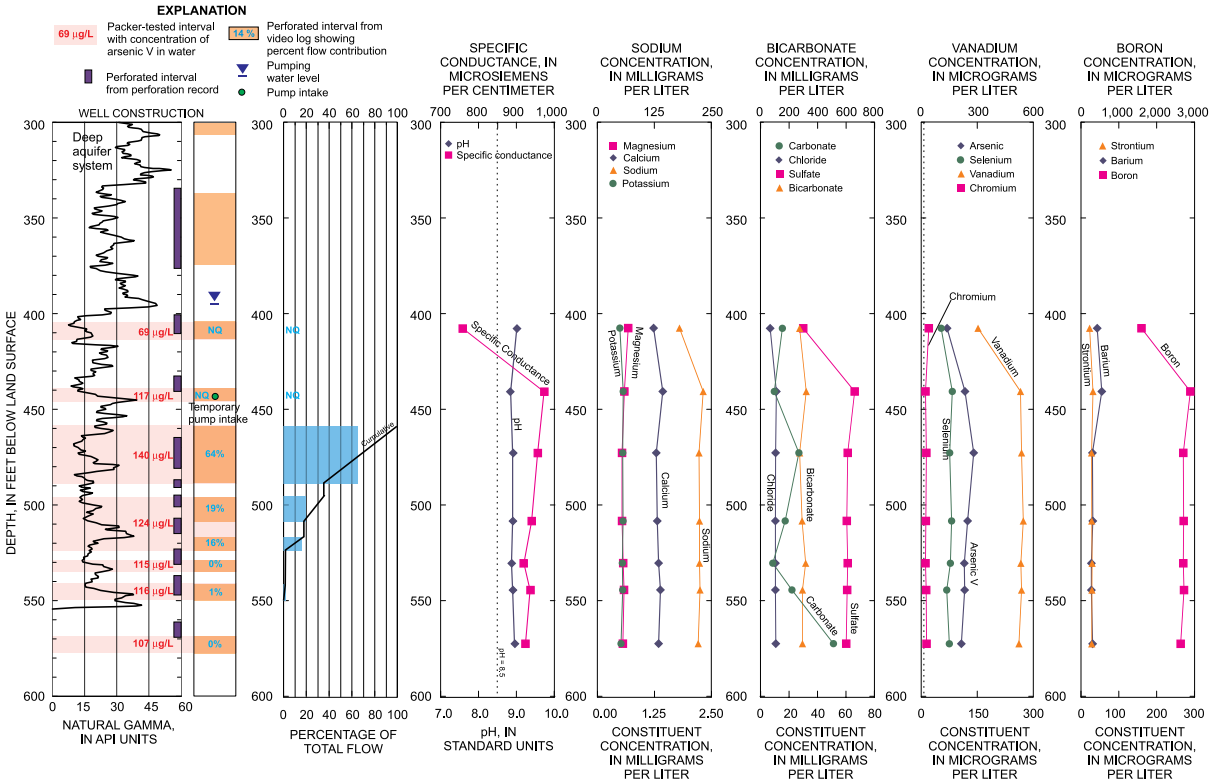


Figure 22. Results of wireline logging, impeller-flowmeter, and packer tests in Norman Well 23, September 2004. [µg/L, micrograms per liter; API, American Petroleum Institute; NQ, not quantifiable; %, percent].

An impeller-flowmeter log was run in Well 23 to determine zonal flow contributions during production. A temporary pump was installed at 446 feet below land surface, about 50 feet below the pumping water level and about 140 feet above the depth of the pump during normal production. Throughout flowmeter logging, the production rate was maintained at 200 gallons per minute. The impeller-flowmeter log identified a 30-foot perforated interval near the temporary pump intake as the greatest contributor of water to the well. This interval (458-488 feet below land surface) supplied nearly two thirds (64 percent) of the well discharge (Figure 22). The next two deepest zones, with perforated intervals of 495 to 508 and 516 to 523, produced 19 and 16 percent of the well discharge, respectively (Figure 22). The zones below 525 feet contributed only 1 percent of the well discharge according to the impeller-flowmeter log (Figure 22). The quantity of water being contributed from zones above the temporary pump intake could not be determined because of limitations of the impeller-flowmeter tool.

Seven perforated zones or groups of zones identified in the video log were selected for packer testing. Five zones were sampled using a packer spacing of 10 feet and two zones were sampled using a packer spacing of 30 feet. Each zone was tested at a production rate of about 5 gallons per minute for 1 to 2 hours using a submersible pump. No water-level measurements were made outside the packer string to verify that the packers were sealed against the casing. One targeted zone failed to produce water, however, because the packers were placed at the wrong depth. When the problem was corrected, the targeted zone produced water. Because the wrong placement resulted in no production, the packers are believed to have sealed properly against the casing wall.

Samples were collected in the same manner as test-hole and well-head samples, and a sealed flow-through cell was used for measurement of field water properties. Samples were acidified, if necessary, and placed on ice immediately after collection. Alkalinity titrations were completed on site after all other samples were collected and preserved.

With the exception of data from the shallowest zone (403-412 feet below land surface), the data indicated relatively little variation in water quality between sampled zones in Well 23 (Figure 22, Table 4). Dissolved arsenic concentrations (as arsenic V) exceeded 10 µg/L in all zonal samples from Well 23 (Figure 22). At 140 µg/L, arsenic concentration was greatest in the water produced from 458 to 488 feet below land surface (Figure 22). The arsenic concentration mostly decreased with distance from this interval, measuring 69 µg/L in the shallowest tested interval (404-413 feet) and 107 µg/L in the deepest tested interval (568-577 feet; Figure 22). Selenium concentrations greater than 50 µg/L also were measured in all zonal samples from Well 23 (Figure 22).

USGS Well Profiler

The results of water-quality sampling and velocity profiling in 11 selected Norman wells are presented in Appendix 4 and 5. Each figure in Appendix 5 (5A-5K) includes a natural gamma-ray or spontaneous-potential log for general lithologic determination. To the right of each log is a well construction record showing open (perforated or screened) intervals, where known, the estimated depth of the pump intake, and the pumping water level at the time of testing. To the right of the well construction information are estimates of flow contribution percentages from open intervals. This graph also includes an estimate of percentages of flow coming from above and below the pump, where applicable. Estimates of flow contribution from intervals above the pumping water level are marked “cascading.” The estimate of well yield (Q) appears at the top of the flow contribution graph.

The remaining graphs show water-quality samples from various sample depths in the well. The samples collected at each well include about six depth-dependent samples and one well-head sample. Some wells (02, 05, and 36) were sampled twice. These wells were resampled to conduct more precise analysis of dissolved arsenic by the ICP-MS method; results of the second sampling are presented. From left to right the water-quality graphs on each figure show selected field water properties, major cations, major anions, and trace elements (including arsenic) that were routinely detected. For some constituents, an additional scale was placed at the top of the graph so related constituents could be displayed together. Depth-dependent samples are represented by filled markers at the appropriate sample depth, and well-head samples are represented by hollow markers at the approximate depth of the pump. Constituents that were not detected in depth-dependent samples were graphed as having a zero concentration. Also, two vertical reference lines appear on each figure to indicate where pH exceeds 8.5 standard units and arsenic exceeds 10 µg/L.

Tracer-pulse Velocity Profiles

Because disposal of large volumes of well discharge was problematic, the tracer-pulse velocity profiles in this study were conducted after less than 8 hours pumping duration. Flow contribution percentages varied widely between wells. In many wells, greater than 50 percent of the well yield originated from one or two contributing zones near the pump intake. Whether this phenomenon is real or simply an artifact of the zonal flow computation

Table 4. Well-head values and maximum and minimum values of constituents measured in depth-dependent samples from 11 selected wells, Norman, Oklahoma, 2003-2006.

Well	Station number	Specific conductance, field (mS/cm at 25 °C)	pH, field, standard units	Calcium, filtered (mg/L)	Magnesium, filtered (mg/L)	Sodium, filtered (mg/L)	Sodium/Calcium ratio (milliequivalent ratio)	Bicarbonate, unfiltered (mg/L)	Carbonate, unfiltered (mg/L)	Sulfate, unfiltered (mg/L)	Chloride, unfiltered (mg/L)	Arsenic, filtered (mg/L), shaded values by ICP-MS	Boron, filtered (mg/L)	Chromium, filtered (mg/L)	Selenium, filtered (mg/L)	Uranium, filtered (mg/L)
02	351426097232201	580 434 1,006	8.51 7.46 8.79	6.91 6.27 16.7	5.18 4.63 13.8	118 67.0 136	14.9 9.31 7.10	305.2 247.2 407.2	3.2 0.9 E 15.2	18.4 12.3 39.2	15.2 3.88 59.9	11.0 0.757 13.1	1,860 934 2,270	83 29 85	20 E 7 E 31	< 20 ND ND
05	351409097231801	559 435 1,116	8.46 6.92 8.92	5.60 2.35 9.53	4.08 1.13 7.56	114 81.5 245	17.7 30.2 22.4	288.1 256.7 414.6	5.0 1.5 24.8	15.7 9.99 42.9	13.2 2.09 86.9	10.9 1.73 63.1	1,370 703 3,620	73 64 114	14 E 7 40	< 20 ND ND
06	351357097242001	699 680 951	8.35 8.35 8.77	6.67 4.68 7.25	4.12 2.30 4.42	150 149 211	19.6 27.8 25.4	404.1 404.1 437.3	6.7 4.7 15.6	20.4 19.5 35.4	15.4 11.3 60.4	9.55 8.99 24.5	1,590 1,570 2,250	23 16 34	< 100 ND E 43	< 13 ND ND
07	351414097293901	1,252 1,252 1,369	8.45 8.35 8.45	3.33 3.33 3.86	1.18 1.18 1.38	278 278 304	72.8 72.8 68.7	426.4 426.4 444.6	8.5 6.9 8.5	223 223 292	20.3 20.3 22.7	22 15 22	4,460 4,460 5,080	14 ND 14	< 100 ND ND	16 16 44
13	351550097283801	571 534 577	8.90 8.59 8.99	1.98 1.61 2.24	0.959 0.766 1.15	126 125 132	55.5 67.7 51.4	272.8 272.8 291.9	12.5 3.1 12.5	13.1 12.4 13.7	3.79 3.73 4.53	27 27 34	1,470 1,340 1,590	24 22 26	< 100 ND ND	< 13 ND ND
15	351648097285101	632 533 693	9.11 9.02 9.22	2.00 0.949 2.00	0.712 0.320 0.995	147 124 163	64.1 114 71.0	327.0 300.0 329.7	20.9 14.3 28.6	26.4 11.8 35.8	7.79 4.43 9.99	41 27 51	1,210 981 1,260	21 19 23	< 100 ND ND	< 13 ND ND
18	351726097290901	576 567 582	8.89 8.89 8.96	1.49 1.39 1.49	0.56 0.529 0.560	134 134 136	78.4 84.0 79.6	299.8 298.5 312.6	14.3 13.5 14.9	19.9 19.1 21.7	10.7 10.5 11.0	18 ND 19	1,460 905 1,480	16 15 17	< 100 ND ND	< 13 ND ND
23p	351401097252301	-- 759 973	-- 8.85 9.01	-- 1.23 1.43	-- 0.537 0.672	-- 180 232	-- 128 141	-- 342.9 401.5	-- 10.8 E 64.1	-- 37.5 82.6	-- 8.35 13.9	-- 69 140	-- 1,600 2,890	-- 11 20	-- E 53 E 83	-- 44 85
23	351401097252301	912 801 912	8.95 8.87 9.04	1.29 1.26 1.29	0.605 0.588 0.669	210 179 213	142 124 144	413.5 301.9 413.5	21.3 3.3 23.2	68.6 48.3 68.6	12.9 9.59 12.9	120 71 120	2,640 1,890 2,640	14 14 19	E 65 ND E 65	85 61 85
31	351542097262801	652 649 668	8.84 8.84 8.99	1.85 1.75 2.29	0.685 0.659 0.733	149 145 151	70.2 72.2 57.5	306.9 303.1 312.6	14.3 14.3 18.0	39.2 38.5 39.9	10.7 10.5 13.7	41.8 40.2 44.3	1,690 1,520 1,700	27 26 54	E 34 ND E 40	< 13 ND ND
33	351541097245301	449 428 450	7.68 7.68 8.13	22.3 21.2 28.2	19.4 18.3 24.2	38.2 27.8 40.9	1.49 1.14 1.26	223.6 219.0 231.3	0.4 ND 0.4	9.70 9.07 9.88	8.30 7.84 10.4	0.423 0.262 0.600	821 795 875	57 49 63	E 9 E 7 E 10	< 20 ND ND
36	351633097241901	539 440 539	8.24 7.54 8.48	13.8 13.8 26.1	10.9 10.9 21.3	93.1 34.4 93.1	5.88 2.17 3.11	314.3 174.7 364.8	4.2 0.5 4.2	12.0 9.06 12.0	11.1 4.74 11.1	16.5 0.429 16.5	1,760 347 1,760	84 48 84	E 40 E 7 E 40	< 13 ND ND

method is unknown. That the majority of well discharge can originate from one zone is surprising, but is consistent with an impeller-flowmeter log from Gossell et al. (1999) who found that 48 percent of the discharge from a high capacity well was produced by only 6 percent of the total screened interval. Computations of percentages originating from above and below the pump intake do not routinely show greater production from either side of the pump intake. Production is often evenly (about 50 percent and 50 percent) or nearly evenly split (about 40 percent and 60 percent) between either side of the pump intake in Norman wells. Lithology and zone thickness, as determined from the natural gamma-ray or spontaneous-potential log, do not appear to be reliable indicators of the quantity of flow contribution. In some wells, though, thicker and cleaner sandstones do appear to contribute more flow than thinner and finer-grained sandstones.

In terms of repeatability, tracer-pulse velocity profiling is sensitive to the water-level and pumping duration prior to testing. A replicate tracer-pulse velocity profile was completed in Well 23, about 1 month after the completion of the initial profile. Well 23 was unused at the time of testing, and the water level and pumping duration were nearly identical between tests. Except for the uppermost zone, which is influenced greatly by cascading water, well velocities and zonal contribution percentages were nearly identical. The tracer-pulse traveltimes were repeatable to within 2 seconds at most depths and to within 5 seconds at depths farthest from the pump (Figure 16). These results translate to about a 2 percent difference in traveltimes for all depths in the well. Repeated graphical analysis of the tracer-pulse traveltime data showed that velocities and zonal contribution percentages are mostly repeatable to within 5 percent. No repeat measurements were completed to determine the effect of the PVC access tube on repeatability of down-hole tests used in this study.

The changes in well-bore velocity caused by zonal flow contribution tend to occur as slope breaks, not as gradual changes in slope (curves). Changes in velocity across thick contributing intervals (30 – 40 feet) sometimes occur as multiple distinct slope breaks, not curves. This phenomenon indicates that contributed water may be entering the access tube only at discrete depths, and that the water inside the access tube may not be identical to water outside the access tube at a given depth. If contributed water is entering the access tube at discrete depths and not continuously, this could be caused by constrictions (such as pump-column pipe connections) that may force locally contributed water into the access tube. However, abrupt changes in slope of tracer-pulse traveltimes also were observed in Wells 23 and 33, which were not equipped with access tubes at the time of the tracer-pulse velocity profile.

Estimates of Well Yield

The tracer-pulse log data and available well-construction details for each selected well were used to estimate well yield. Estimated well yields (Table 5) were mostly within 20 percent of those in City of Norman records (Table 1). The tracer-pulse estimates tended to be greater than the reported well yields. The estimates are probably slightly overestimated because first arrivals of dye were used to compute traveltimes. The time of first arrival corresponds to the maximum velocity of water flowing through the center of the column pipe. Small, positive differences in well yield estimates also could be caused by the difference in backpressure between pumping to the distribution pipe and pumping to waste or by differences in pumping duration between tracer-pulse profiling and actual production. The estimates of yield for Wells 15 and 31 were more than 20 percent different from values reported in City of Norman records. The reason for the abnormally large disagreement in tracer-pulse-determined estimates and reported yield values in these two wells is unknown. Larger differences could result from underestimating scale thickness inside the column pipe or from seasonal changes in aquifer water levels and well output since the Norman records were created.

Depth-specific Water Quality

Well-head arsenic concentrations measured in this study compare favorably with historical well-head samples at each well (Figure 23). All well-head arsenic samples collected in this study were within the range of historical well-head concentrations (Figure 23). Most depth-dependent arsenic samples also were within the range of historical well-head concentrations. Only Wells 02 and 36 had depth-dependent arsenic samples that were outside the range of historical well-head concentrations (Figure 23).

Most of the selected wells (06, 07, 13, 15, 18, 23, and 31) showed elevated or near-elevated arsenic concentrations at all depths in the well (Appendix 4, Figure 23). For these wells, well-modification techniques would be ineffective in lowering well-head arsenic concentrations to less than 10 micrograms per liter. Wells 02, 05, 33, and 36, however, showed some potential for successful application of well modification techniques for arsenic remediation. All these wells lie near the Hennessey-Garber contact (Figure 2). Wells 02 and 05 are on the eastern margin of the well field; Wells 33 and 36 are on the northern margin of the well field (Figure 2).

Elevated arsenic concentrations in Wells 02, 05, and 36 only were detected in depth-dependent samples from the bottom of the well. In Well 33, elevated arsenic concentrations were not detected at any depth. In fact, no arsenic concentrations greater than 1 µg/L were detected in the depth-dependent samples or the well-head samples

Table 5. Estimated well yields determined from tracer-pulse velocity profiling for 11 selected wells in Norman, Oklahoma, 2003-2006

Well	USGS station number	depth of t_{min}	t_{min}	v_{col}	r_{col}	A_{col}		Estimated well yield, in gallons per minute			Well yield from Norman records, in gallons per minute	Relative percent difference
						max	min	max	min	mean		
02	351426097232201	560	55	10.2	2.0	0.0873	0.0767	400	351	**376 ± 24**	335	11.4
05	351409097231801	640	95	6.7	2.0	0.0873	0.0767	265	233	**249 ± 16**	212	15.9
06	351357097242001	610	87	7.0	2.0	0.0873	0.0767	275	242	**259 ± 17**	218	17.1
07	351414097293901	660	154	4.3	2.0	0.0873	0.0767	168	148	**158 ± 10**	182	-14.0
13	351550097283801	630	125	5.0	2.0	0.0873	0.0767	198	174	**186 ± 12**	190	-2.2
15	351648097285101	615	140	4.4	2.5	0.1364	0.1231	270	243	**256 ± 13**	164	44.0
18	351726097290901	600	139	4.3	2.0	0.0873	0.0767	170	149	**159 ± 10**	147	8.0
23	351401097252301	585	89	6.6	2.0	0.0873	0.0767	258	227	**242 ± 16**	250	-3.0
31	351542097262801	550	84	6.5	2.0	0.0873	0.0767	257	226	**242 ± 16**	172	33.6
33	351541097245301	610	108	5.6	2.0	0.0873	0.0767	222	195	**208 ± 13**	219	-5.0
36	351633097241901	670	103	6.5	2.0	0.0873	0.0767	255	225	**240 ± 15**	260	-8.0

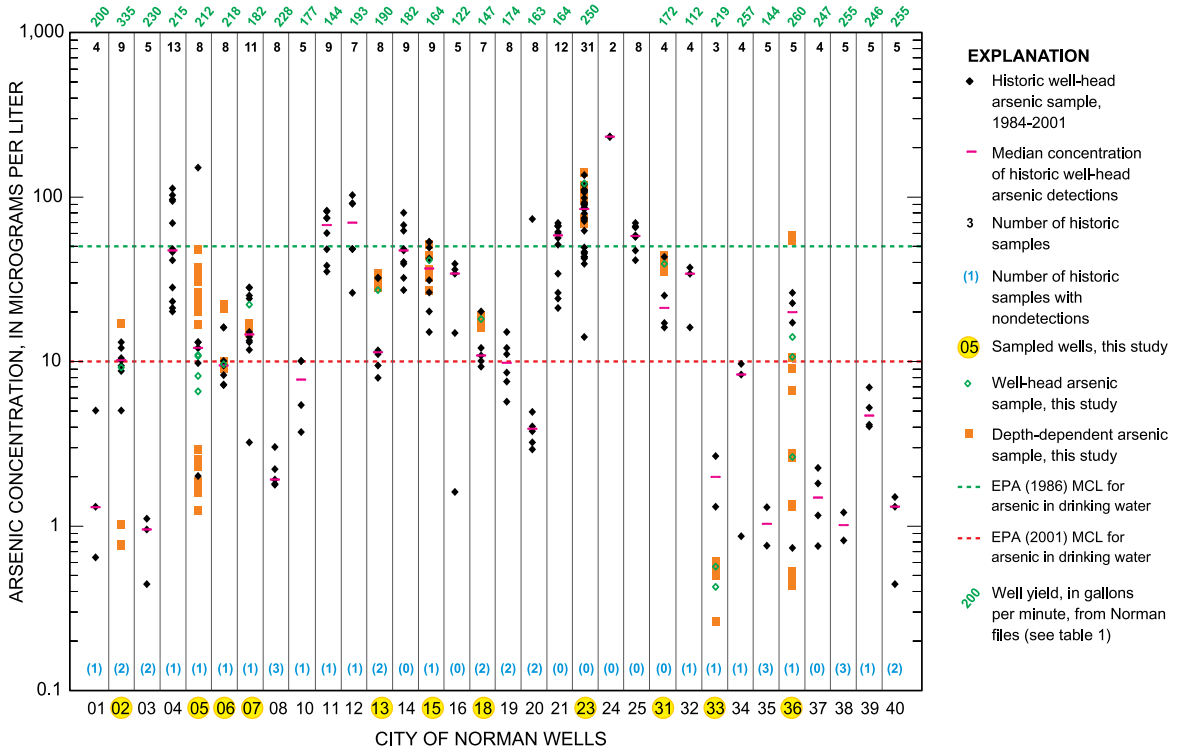


Figure 23. Well-head arsenic concentrations (1984-2001), median values, and maximum contaminant levels (MCL) for arsenic in the Norman well field, 2003, with depth-dependent and well-head arsenic concentrations from 11 selected wells, 2003-2006.

at Well 33 (Table 4, Appendix 4). The sampler, however, could not access the bottom of Well 33 because of the presence of a pump shroud. In 2005, Well 33 produced one well-head water sample (not included in Table 1 or Figure 11) which exceeded the arsenic MCL. The water responsible for elevated well-head arsenic concentrations is believed to be originating from the perforated zones below the pump intake (606-622 feet). Also, because the well-head sample from this investigation had an arsenic concentration less than 1 µg/L (Table 4), these deep perforated zones are likely to cause deterioration of well-head water quality only after the well has been pumped for periods longer than 8 hours.

Isotopic Composition of Water from Wells

Stable isotopes of water can serve as approximate surrogates for water age or time since recharge in some aquifer systems. To investigate the utility of this relation, as defined by Parkhurst et al. (1996), samples from the selected Norman wells (except Well 33) and the arsenic test hole were analyzed for the stable isotopes of water (Figure 24). Oxygen-18 ($\delta^{18}O$) and deuterium ($\delta^{2}H$) abundances in water are expressed as ratios as a per mil (parts per thousand) difference relative to the Vienna Standard Mean Ocean Water (VSMOW) (Gonfiantini, 1978). Usually, the top and bottom depth-dependent samples from each well were submitted for isotopic analysis because these samples are representative of a single zone (not mixtures). A well-head sample also was submitted for isotopic analysis.

Water sampled from 10 selected Norman wells had $\delta^{18}O$ values that ranged from -5.98 to -6.82 per mil, with a median value of -6.39 per mil (Figure 24); the $\delta^{2}H$ values ranged from -34.8 to -42.4 per mil with median value of -38.4 per mil (Figure 24). Water sampled from the Norman arsenic test hole had $\delta^{18}O$ values that ranged from -5.70 to -6.79 per mil, with a median value of -6.68 per mil (Figure 24); the $\delta^{2}H$ values ranged from -34.0 to -42.8 per mil with median value of -41.6 per mil (Figure 24). These ranges are consistent with published values of isotopic ratios in continental precipitation at mid-latitudes (Kendall and Coplen, 2001) as well as previously measured isotopic ratios in ground water of the Central Oklahoma aquifer (Parkhurst et al., 1996). The ground-water samples mostly plot along a line that is parallel to and about 2.1 per mil ($\delta^{2}H$) greater than the global meteoric water line (Craig, 1961; Figure 24). This deuterium excess value of 2.1 per mil is consistent with the value reported by Parkhurst et al. (1996) for ground water in the Central Oklahoma aquifer (Figure 24).

Parkhurst et al. (1996) found a relation between $\delta^{2}H$ and water age determined from tritium and radiocarbon dating in the Central Oklahoma aquifer. Though not a perfect relation, water age increases with decreasing $\delta^{2}H$. According to this relation, the youngest water was found in Wells 18, 02, and the shallowest sampled zones of the arsenic test hole (320-350 feet below land surface) (Figure 24). The oldest water sampled was found in Well 07 and intermediate sandstone zones of

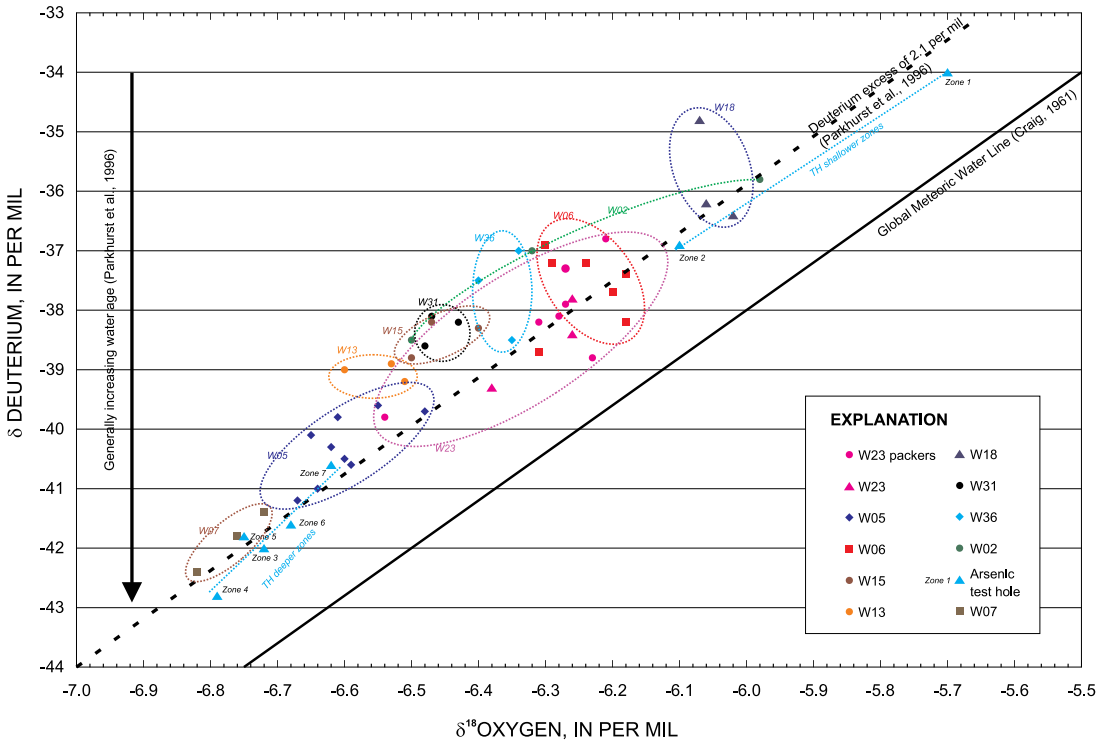


Figure 24. Plot of δ^{18}Oxygen and δDeuterium for selected wells in the Norman well field. W indicates samples from public-supply wells and TH indicates samples from the Norman arsenic test hole.

the arsenic test hole (568-598 feet below land surface). Well 07, which is farthest west in the confined part of the aquifer, draws some of the oldest water in the well field according to the NAWQA conceptual model and the δ^2H – age relation.

According to the age relation (Parkhurst et al., 1996), the ages of water in the test hole could range from only a few hundred years before present to more than 30,000 years before present. This wide range in age, as compared to well samples, is contradictory to ages expected from the modeled flow paths in Figure 20 and could be characteristic of the natural, undeveloped flow system. Completed wells, after years of seasonal pumping stresses, could develop a narrower age signature due to mixing by intra-borehole flow between zones when the well is not being used (Zinn and Konikow, 2007). The range in estimated ages of test-hole water also could be a result of the imperfect relation between age and δ^2H, or the interception of units in the test hole that are hydrologically isolated from regional flowpaths.

The δ^2H of samples from the test hole indicates a substantial change in estimated water age from the two shallowest sampled zones (320-350 and 416-456 feet below land surface) to the next lowest sampled zone (488-502 feet below land surface, Figure 24). This gap between younger and older water (Figure 24) corresponds to the gap between detected and non-detected arsenic in zonal water samples (Figure 19). Coincidently, the test-hole sample with the greatest apparent age (Zone 4, Figure 24) appears, at least qualitatively, to be one of the least permeable (shaliest) sandstone units in the test-hole gamma-ray log (Figure 19).

Major-ion Water-quality Trends with Depth

Piper (1944) diagrams were used in this report to illustrate water-type in selected wells and characterize common trends in water quality with depth. In Piper diagrams, anion and cation compositions are plotted in milliequivalents on separate ternary diagrams and are projected into another, diamond-shaped diagram. The location of the data point in the diamond reveals the general composition of water and may indicate a water source. If several analyses trend in a line or curve, the position and direction of the trend are indicative of active chemical processes or mixing of water from different sources. If all depth and well-head samples from the 11 Norman wells are plotted on the diagram, three major processes are evident – cation exchange, influence of sulfate-rich rocks (Hennessey Group), and the influence of brine (Figure 25). The pattern of data points in this study is consistent with the pattern for all wells in the Central Oklahoma aquifer (Figure 25) and the pattern for high-arsenic wells in the Central Oklahoma aquifer (Schlottmann et al., 1998).

All analyses in the ternary diagram for cations plotted along a line representing a magnesium to calcium ratio of about 1.4. The sodium plus potassium proportion ranged from about 20 percent to about 100 percent, usually increasing with depth in individual wells (Figure 25). This trend represents the process of cation exchange and shows a continuous transition from a calcium-magnesium dominated water type to a sodium dominated water type. This transition can occur rapidly with depth as in Well 36 and the Norman arsenic test hole (Figure 25). In the test hole, the process was most pronounced. The increase in sodium plus potassium proportion was nearly 65 percent from the sandstone Zone 2 (ending at 456 feet below land surface) to the sandstone Zone 3 (beginning at 488 feet below land surface).

The proportions of sulfate and chloride in the ternary diagram for anions tended to increase with depth in individual wells (Appendix 4). This trend indicates the influence of brine on wells that are drilled too deep or wells that are in the confined aquifer system. Wells in the confined aquifer system, such as Well 07 and Well 23 (Figure 25), showed the possible influence of dissolution of sulfate-rich rocks (Parkhurst et al., 1994). The data from these wells showed the beginnings of a transition from a sodium-bicarbonate water type to a sodium-chloride-sulfate water type. An increase in the sulfate or chloride proportion also could indicate connate water that has not been sufficiently flushed from the aquifer by recharge.

In the composite diamond-shaped part of the Piper diagram, the general evolution of water with depth is from a calcium-magnesium-bicarbonate water type, to a sodium-bicarbonate water type, to a sodium-chloride-sulfate water type (Figure 25). This is the same trend seen in regional maps of well-head analyses in the Norman area (Figure 9; Parkhurst et al., 1994). Wells that plot further along this trend are more likely to produce water with arsenic concentrations exceeding the MCL. Analyses from different depths in some wells, such as Well 36 and Well 05, are spread along the cation-exchange trend (Figure 25). These wells are the most likely to benefit from the remediation techniques examined in this report because these wells will have greater contrast in water quality between individual zones. From the results of this study, wells producing water with sodium plus potassium greater than 90 percent of milliequivalent cations tend to be poor candidates for remediation by well modification because the cation-exchange process (and release of arsenic) is too far advanced.

The transition from calcium-magnesium-bicarbonate water to sodium bicarbonate water occurred abruptly in Wells 02, 05, and 36, as did the transition from low-arsenic water to high-arsenic water (Appendix 5).

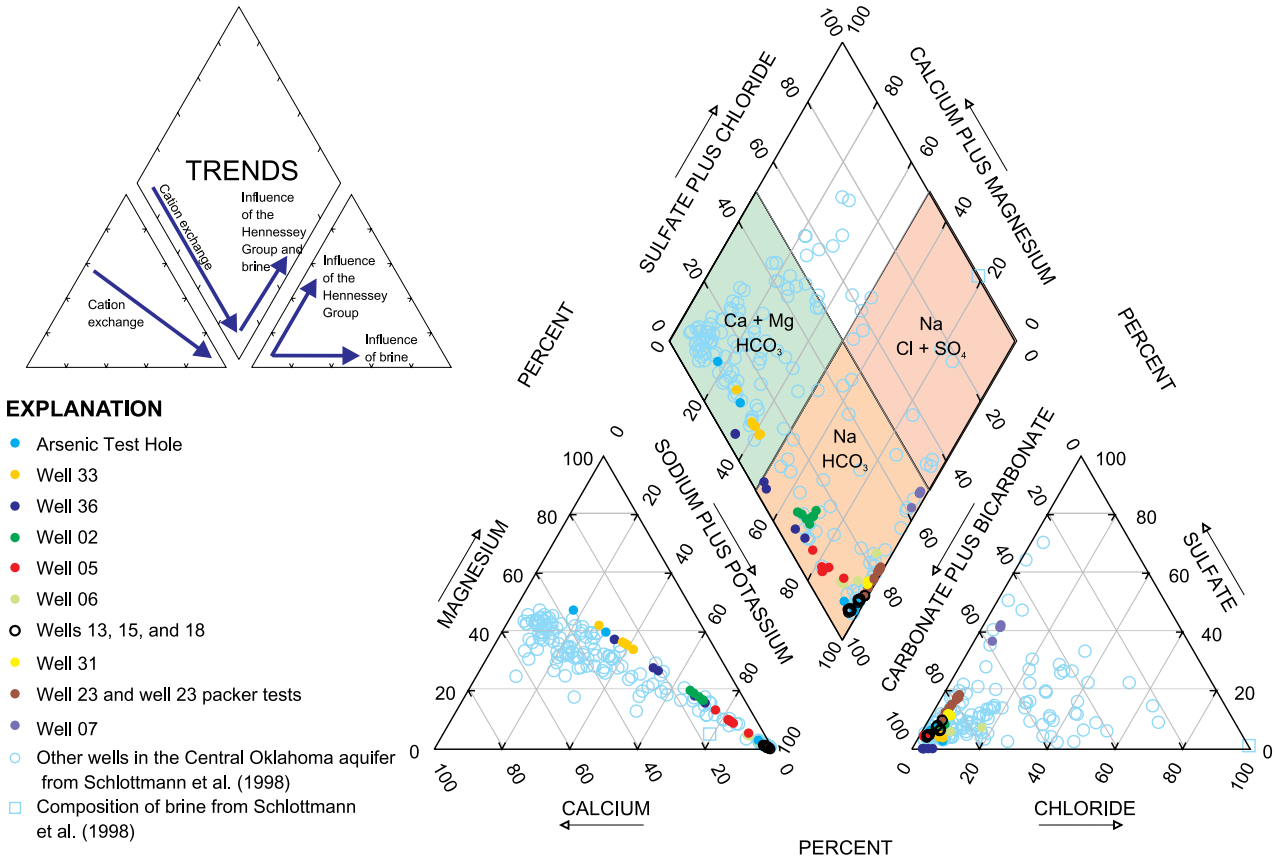


Figure 25. Piper diagram showing water types in the arsenic test hole and selected wells (colored circles) in the Norman well field, with data from Schlottmann et al. (1998) (empty faded circles). The data from this study (Norman well field) show trends very similar to the Schlottmann data (wells from the Central Oklahoma aquifer).

Abrupt changes in water quality with depth may be good indicators that intervening low-permeability units may be laterally pervasive.

Iron Species

Because dissolved iron species and arsenic concentrations are related in some aquifer systems, water samples were analyzed for ferrous and total dissolved iron using field methods. The maximum ferrous iron (Fe II) concentration detected in Norman wells was 0.82 milligrams per liter. Ferrous iron concentrations, as a percentage of total iron concentrations, ranged from 0 to 100 percent, with a median of 43 percent. Ferrous iron accounted for about 30 to 60 percent of total iron in most samples. About 14 percent of samples had no detection of ferrous iron and about 6 percent of samples had ferrous iron concentrations that equaled total iron concentrations (Appendix 4).

Other Characteristics Related to Arsenic Release in the Central Oklahoma Aquifer

Because the cation-exchange process (by increasing pH) indirectly causes desorption of arsenate in the Central Oklahoma aquifer, indexes such as the sodium/calcium milliequivalent ratio (Figure 26, Table 4) or sodium adsorption ratio (Appendix 4) are related with dissolved arsenic concentrations. Also, vanadium concentrations are related with arsenic concentrations as the mechanism for vanadate release is similar to that of arsenate (Figure 26). A pH greater than 8.5 and conductance greater than 500 µS/cm are indicators of elevated arsenic concentrations in water from Norman wells (Figure 26). Fortunately, measurement of these properties is easy, fast, and inexpensive. Relations between these properties and dissolved arsenic are not perfect, though, and could not be used to accurately predict arsenic concentrations in the Norman well field.

Orthophosphate and Sulfate

Arsenic concentration is related with orthophosphate concentration, but orthophosphate concentrations measured in Norman wells were relatively small (Figure 26). In Well 23, which had the greatest dissolved arsenic concentrations, orthophosphate was measured from 0.049 to 0.064 milligrams per liter (Figure 26). Arsenic concentration also is related with sulfate

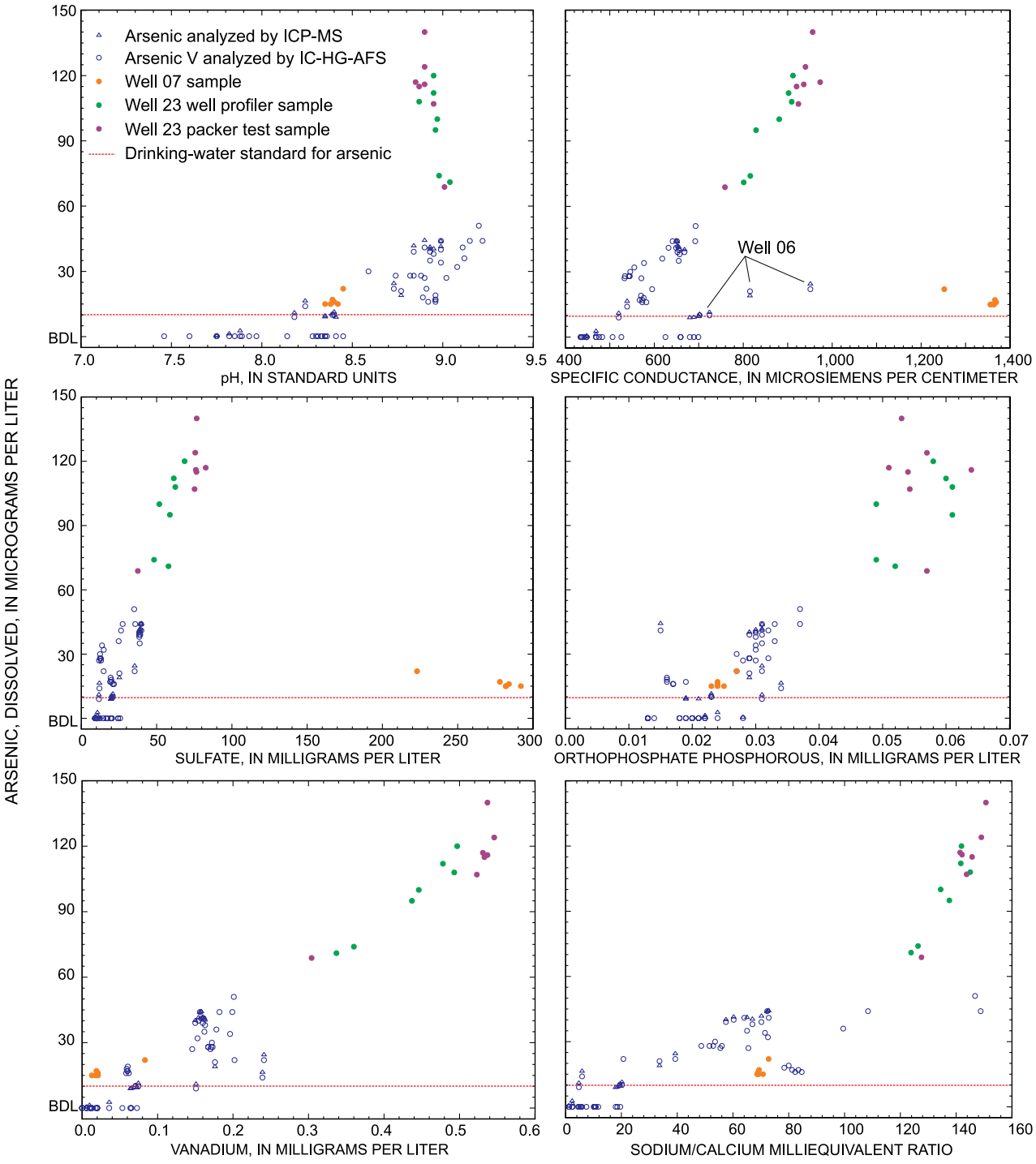


Figure 26. Graphs of selected constituent concentrations that are related with arsenic concentrations in Norman wells. Samples from Well 07, which is in western Norman, had much greater specific conductance and sulfate concentration than samples from other selected wells [ICP-MS, inductively-coupled plasma–mass spectrometry; IC-HG-AFS, ion chromatography–hydride generation–atomic fluorescence spectrometry; BDL, below practical quantitation limit].

concentration, another ion known to compete with arsenic for sorption sites (Figure 26) (Stollenwerk, 2003). However, Well 07, which had the greatest sulfate concentrations (three times greater than any other well), had dissolved arsenic concentrations around 20 µg/L (Figure 26). Arsenic competition with orthophosphate and sulfate for sorption sites may be an active process in the Central Oklahoma aquifer, but it is probably not the dominant process causing arsenic desorption in the Norman area.

Barium and Strontium

A few constituents, especially barium and strontium, are inversely related with arsenic (Appendix 4 and 5). Barium and strontium concentrations are usually greatest in the shallower depth-dependent samples from Norman wells and often tend to decrease with depth (Appendix 4 and 5).

Boron

Though boron concentrations are not regulated in drinking water, boron is problematic in irrigation water because elevated concentrations can be harmful to vegetation. According to Hem (1989), concentrations as low as 1 mg/L can be toxic to some plants. Boron concentrations exceeded 1 mg/L in every sampled well except Well 33 (Table 4). Boron concentrations greater than 2 mg/L were measured in Wells 02, 05, 06, 07, and 23 (Table 4). The concentrations measured in Well 07 (4.5 to 5.1 mg/L) are comparable to those measured in ocean water (Hem, 1989). For the data collected in this investigation, boron concentration is strongly related to sodium concentration (Appendix 4 and 5).

Chromium, Selenium, and Uranium

Arsenic is not the only contaminant of concern for drinking-water use in the Norman well field. In some of the wells sampled in this study, dissolved chromium concentrations are near the MCL of 100 µg/L (Table 4). Chromium concentrations greater than 70 µg/L were found in depth-dependent samples from Wells 02, 05, and 36 (Table 4). One depth-dependent sample from the bottom of Well 05 contained a chromium concentration of 114 µg/L and was the only sample that exceeded the MCL (Table 4). The wells that appeared to be the best candidates for arsenic remediation also were those with the greatest chromium concentrations.

Chromium concentration decreased sharply with increasing sodium concentration in most wells (Appendix 5). This relationship contradicts findings of Schlottmann et al. (1998), who found a weak increase in chromium concentrations with increasing sodium in wells from the entire Central Oklahoma aquifer.

The MCL for selenium is 50 µg/L, but the practical quantitation limit for some analyses in this report is 100 µg/L. Therefore, some wells with no detection of selenium could exceed the MCL. Though selenium was detected in water from most sampled wells, concentrations exceeding the MCL were detected only in Well 23 (Table 4).

The MCL for uranium is 30 µg/L. Uranium was detected in water from Wells 07 and 23, which had well-head concentrations of 16 µg/L and 85 µg/L, respectively (Table 4). Concentrations exceeded the MCL in the two deepest samples from Well 07 and in all depth-dependent samples from Well 23 (Appendix 4, Table 4).

7.0
Comparison of Traditional Methods and USGS Well-profiler Methods

Norman Well 23 (Table 1) was selected for a comparison of depth-dependent water-quality data collected using traditional methods (packer tests and impeller-flowmeter logs) and the USGS well profiler (depth-dependent sampling and tracer-pulse logs). This well produced water with relatively high dissolved constituent concentrations and was believed to have good contrast in arsenic concentrations between zones (Figures 11 and 22). The purpose of the comparison was to evaluate whether the traditional methods and USGS well profiler methods yielded comparable data and to determine if one of the methods yielded more useful results than the other. The comparison showed that the two methods of velocity logging are different, and neither is perfectly suited to the problem addressed in this study. Impeller-flowmeter velocity data are collected at conditions that may differ greatly from production conditions, especially when the pump intake is located near the bottom of the well during normal production. Many impeller-flowmeters, like the one used in this study, cannot quantify contribution from open intervals above the pump intake because of a lack of space for the tool to fully deploy. Likewise, tracer-pulse velocity logs cannot quantify contribution from open intervals within a few feet of the pump intake. Tracer-pulse velocity logs are capable of estimating flow contribution from cascading water, though. The tracer-pulse velocity profile also requires many assumptions and simplifications about cross-sectional areas and velocities in the well.

With inflatable straddle packers, a water sample originates from only one zone (if the packers seal against the casing), but the sample is not collected under normal pumping conditions. A sampled zone, in reality, may produce water during packer tests but be nonproducing during normal pumping conditions. The USGS well profiler, in comparison, collects a sample during normal pumping conditions, but the sample represents a mixture of water from multiple producing zones. Also, the sample is collected at a single point in the well; if the sampling device is located next to a perforation at the time of sampling, the sample water retrieved may not be truly representative of the mixture of water in the well bore at a given sample depth. The USGS well profiler may not be well suited for determining arsenic concentrations in individual zones, but the well profiler is useful as a qualitative tool for assessing remediation possibility. The depth-dependent sampling method can identify the depth in the well bore at which water becomes unsuitable for public supply (exceeds the MCL), even without any data on flow contribution. The USGS well profiler does offer considerable savings in terms of cost and well down-time.

Differences in Flow Data

The flow-contribution data generated by the impeller-flowmeter method (Figure 22) and the tracer-pulse method (Appendix 5H) compare poorly. The poor relation is probably a result of the difference in pump placement during the two tests. Ideally, the intake would be in the same location for both tests, but the impeller-flowmeter tool used in this study could only log below the pump. To maximize the amount of data collected with the impeller-flowmeter, the pump was raised to a location just below the pumping water level. The tracer-pulse log was completed with the pump intake at the bottom of the well because this is the configuration used during production. Both flow-logging methods identified the zone nearest the pump intake as the greatest producer of water to the well. However, the greatest producing zone identified by the tracer-pulse method was shown to produce almost no water in the impeller-flowmeter test. This result indicates that pump placement can have profound influence over zonal contribution percentages. If correct, this finding indicates that in some cases water-quality problems could be solved simply by moving the pump intake farther away from contaminated zones in the well.

Differences in Water-quality Data

Because packer testing samples produce zonal water-quality data and USGS well-profiler samples collect depth-dependent water-quality data, comparisons of water quality using the two methods are problematic. Changes in depth-dependent water quality will be more subtle because of mixing and will spatially lag changes in zonal water quality determined from packer tests. Also, the magnitude of influence of zonal water quality on depth-dependent samples is related to the proportion of flow coming from the zone as a percentage of total flow at that sample depth.

Unfortunately, not much contrast in zonal water quality existed in Well 23. The pH of samples in both tests was relatively comparable, remaining around 9.0 for all depths and zones (Figure 22, Appendix 4 and 5). Specific

conductance, however, was different for many depths. Most major ions were similar between the packer testing and depth-dependent samples (Figure 22, Appendix 4). The major-ion data generated by the two methods plot in the same position on a Piper diagram (Figure 25). Results of trace-element analysis were less consistent, but were still similar (Appendix 4).

Using the USGS well profiler, mass-balance calculations of zonal water quality are possible, but in practice do not always yield meaningful results in Norman wells. Mass-balance calculations of zonal concentrations of low-level constituents such as arsenic are not meaningful, especially when constituent concentrations approach the method detection limit (or the limit of analytical precision). Therefore, the USGS well profiler is not a good choice for collection of data to be used in applications such as geochemical modeling. The depth-dependent water-quality data collected by the well profiler are useful, however, as a qualitative tool for identification of zones that may degrade water quality in the Norman wells. The depth-dependent water-quality data, even without flow contribution data, show the depth at which the water mixture in the well becomes unsuitable for public supply.

8.0
Implications for Arsenic Remediation: Well Modification and Well Response

Wells 06, 07, 13, 15, 18, 23, and 31 were determined to be poor candidates for remediation by well modification, either because all zones produced water with elevated arsenic concentrations or because one or two high-arsenic zones supplied most of the water to the well (Table 4, Appendix 5). Wells 02, 05, 33 and 36 were better suited for remediation by well modification. Elevated arsenic concentrations were identified at depth in each of these four wells except Well 33 (Table 4). All depth-dependent samples from Well 33, as well as the well-head sample, had arsenic concentrations less than 1 µg/L (Appendix 4). A pump shroud, however, prevented collection of samples below the pump intake. Well 33 was not selected for attempts at arsenic remediation because elevated arsenic concentrations were not detected. In Wells 02, 05, and 36, elevated arsenic concentrations were detected in water from individual zones near the bottom of the wells (Appendix 4 and 5). Well 02 was excluded from attempts at arsenic remediation because the pumping water level was too deep to allow relocation of the pump intake. Data collected during a two-well (Wells 05 and 36) investigation of arsenic remediation by well modification are presented in Appendix 6.

Pump Relocation

Two of the 11 selected wells were selected for repeated sampling to determine the effects of pump intake relocation on water quality and well yield. In Wells 05 and 36, the pump was moved up to all positions in blank sections of casing that were at least 100 feet below the pumping water level. The pump was moved three times in Well 05 and two times in Well 36. In Well 05, the initial pump intake setting (d0) was at 640 feet below land surface. The pump intake was moved to (d1) 610 feet, (d2) 560 feet, and (d3) 505 feet below land surface (Figure 27). In Well 36, the initial pump intake setting (d0) was at 670 feet. The pump intake was moved to (d1) 650 feet and (d2) 590 feet below land surface (Figure 28). For each pump location, determinations of flow contribution and depth-dependent water quality were repeated to evaluate changes in well dynamics.

For Well 05, the well-head arsenic concentration at intake location d0 was 10.9 µg/L (Figure 27). At intake locations d1, d2, and d3, well-head arsenic concentrations were less than 10 µg/L (8.1 µg/L, 6.5 µg/L, and 7.4 µg/L, respectively; Figure 27). For Well 36, the well-head arsenic concentration at intake location d0 was 16.5 µg/L (Figure 28). At intake location d1, the well-head arsenic concentration increased slightly to 18.8 µg/L (Figure 28). At intake location d2, well-head water quality showed much improvement, with arsenic concentrations of only 2.6 µg/L (Figure 28). Both wells showed short-term improvements in water quality as the pump was moved to the highest locations in the well. In Well 05, arsenic concentration decreased by about 32 percent and well yield decreased by 12 percent (Figure 27). In Well 36, the arsenic concentration at the well head decreased by 84 percent and well yield increased by 13 percent (Figure 28).

Arsenic concentrations at the well head mostly decreased as the pump intake was moved farther from the contaminated zone. However, all well-head measurements described in this report were collected after short periods (hours) of continuous pumping. At Well 36, well-head samples at intake location d2 were recollected after several days of continuous pumping to determine if improvements in water quality persisted. Analysis of these samples revealed that the improvement in well-head water quality was only temporary; the well-head arsenic concentration increased almost a full order of magnitude from 2.61 µg/L to 20.6 µg/L (Figure 28). Well-head water-quality samples, as well as field measurements of specific conductance and pH, indicate that the well was producing a greater proportion of water from the deepest perforated zones when the well-head sample tested at 20.6 µg/L than when the sample tested at 2.6 µg/L (Figure 28). These results are similar to those of Bexfield and Anderholm (2002), who discovered that prolonged pumping of some public-supply wells in parts of the Middle Rio Grande Basin resulted in increased contribution of deeper water. Arsenic concentrations in produced water also may fluctuate with seasonal or prolonged changes in aquifer or zonal water levels (Focazio et al., 2000). Izbicki et al. (2005) found that increases in well-head chloride concentrations can result from increased production from deeper zones over time. This evidence confirms that zonal contributions and well-head conditions can fluctuate substantially over relatively short time periods (months) and long time periods (decades) as a result of increased pumping duration or frequency in a well field. The effect of

extensive and seasonal pumping on well-head arsenic concentrations is a subject that requires further study.

Zonal Isolation

The main objective of zonal isolation in this study was to eliminate production from all zones that contribute water of elevated arsenic concentrations. If this well-modification method is successful, the concentration at the well head should not exceed 10 µg/L (assuming that arsenic concentrations in water from producing zones remain constant over time). Variability in well-head samples over time also should be lessened by eliminating production from zones that contribute elevated arsenic concentrations.

Elevated arsenic concentrations were detected at the bottom of Wells 05 and 36 after every relocation of the pump intake (Figures. 27-28). The depth-dependent water-quality and flow-contribution data were integrated to estimate the arsenic concentrations coming from each perforated interval in Wells 05 and 36 (Figures. 27-28). These estimated zonal arsenic concentrations, which are listed on the gamma-ray logs in Figures 27-28, are not true mass-balance calculations but semiquantitative interpretations of zonal water quality; the precision of data collection and analysis methods used in this study was not sufficient for mass-balance calculations on constituents measured in very small concentrations (µg/L).

The two deepest zones in Well 05 (620-677 feet below land surface) were suspected of contributing elevated arsenic concentrations to the well (Figure 27). The shallower of the two zones was about 10 feet thick and was suspected of producing water with arsenic concentrations of about 20 µg/L (Figure 27). The deeper zone was about 40 feet thick and was suspected of producing water with arsenic concentrations greater than 60 µg/L (Figure 27). Only the deepest zone in Well 36 (648-658 feet, Figure 28) was suspected of contributing elevated arsenic concentrations to the well. This 10-foot sandstone was suspected of contributing water with arsenic concentration greater than 60 µg/L (Figure 28). Though Wells 05 and 36 were considered good candidates for attempts at zonal isolation, Well 36 was selected because of maximum potential for successful

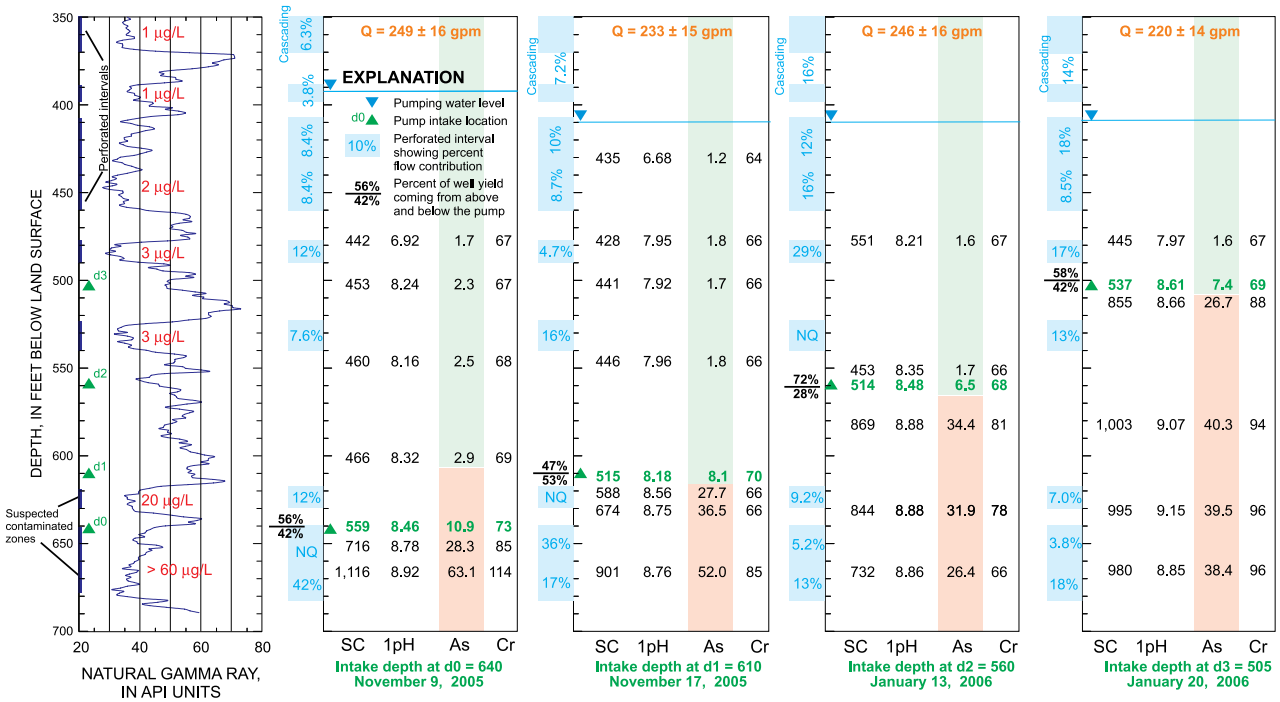


Figure 27. Results of pump intake relocation in Norman Well 05. The final location (d3) resulted in a 32 percent decrease in well-head arsenic concentration and a 12 percent decrease in production rate. Concentration and field water property data in green are from well-head samples; those in black are from depth-dependent samples at the indicated depth. Concentrations in red on the natural gamma ray log are estimated arsenic concentrations in produced water from perforated intervals [Q, production rate in gallons per minute; SC, specific conductance in microsiemens per centimeter; As, dissolved arsenic in micrograms per liter; Cr, dissolved chromium in micrograms per liter; µg/L, micrograms per liter; API, American Petroleum Institute; gpm, gallons per minute; NQ, not quantifiable; %, percent].

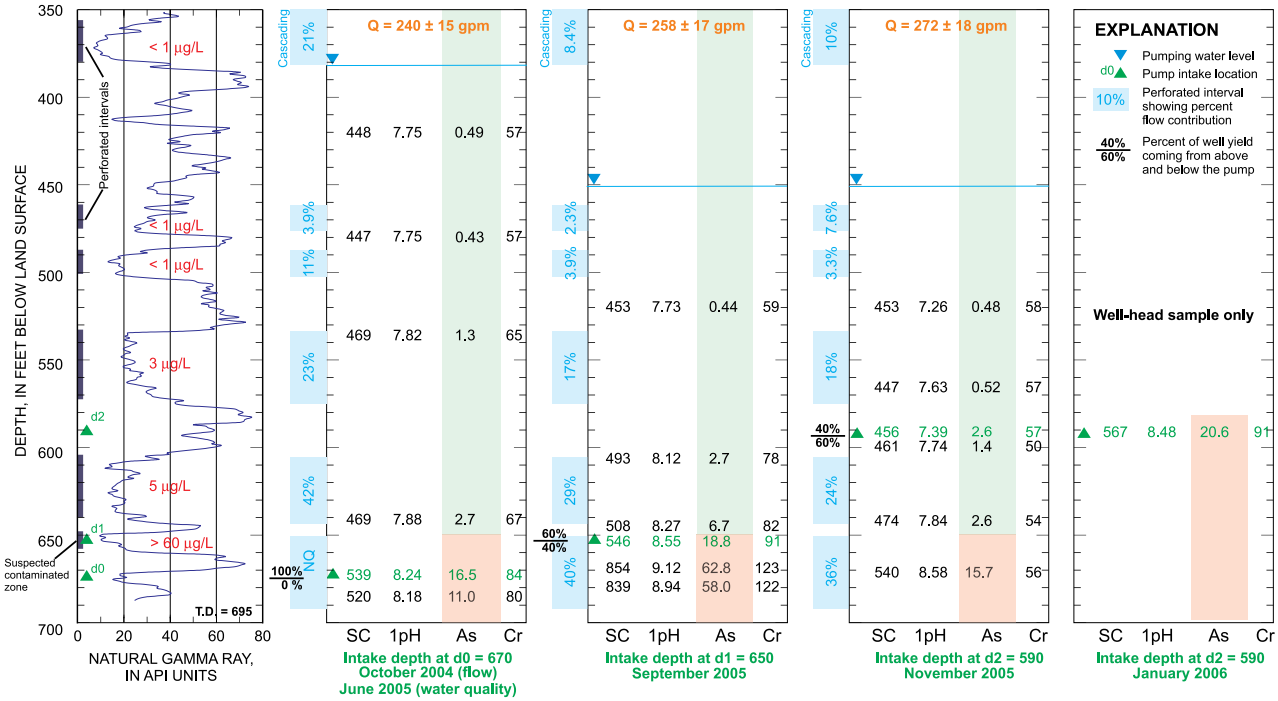


Figure 28. Results of pump intake relocation in Norman Well 36. Concentration and field water property data in green are from well-head samples; those in black are from depth-dependent samples at the indicated depth. The final location (d2) resulted in an 84 percent decrease in well-head arsenic concentration and a 13 percent increase in production rate. However, the improvement in water quality was only temporary and the well-head arsenic concentration increased to 20.6 mg/L after several days of production. Concentrations in red on the natural gamma ray log are estimated arsenic concentrations in produced water from perforated intervals [Q, production rate in gallons per minute; SC, specific conductance in microsiemens per centimeter; As, dissolved arsenic in micrograms per liter; Cr, dissolved chromium in micrograms per liter; µg/L, micrograms per liter; API, American Petroleum Institute; gpm, gallons per minute; NQ, not quantifiable; %, percent].

remediation and the minimum potential for substantial loss of production.

A retrievable bridge plug was installed in Well 36 in June 2006 to isolate the suspect zone from production (Figure 29). A retrievable bridge plug is an inflatable packer with a valve head attached to the mandrel to allow for setting and retrieving from a fixed position in the well. The bridge plug was lowered to a depth of 640 feet from land surface and hydraulic pressure was used to inflate the packer element and seal the bridge plug against the well casing. After the packer inflation was confirmed, the bridge plug and valve head were released from the pipe string by a specialized overshot tool and the pipe was removed from the well. With the plug in place, the bottom 50 feet of the well was sealed off and no obstructions were present in the upper part of the well.

The pump was installed with the intake at about 600 feet below land surface, or about 40 feet above the bridge plug. The well was pumped continuously for about 96 hours while a YSI 600XLM water-quality sonde recorded specific conductance, pH, water temperature, and dissolved oxygen concentration at the well head (Figure 30). Well yield also was recorded (Figure 30). Well-head trace elements (including arsenic by ICP-MS) and major-ion samples were collected daily.

Well yield was about 220 gallons per minute, arsenic concentration was 18.3 µg/L; pH was about 8.2 standard units; and specific conductance was about 510 µs/cm in the first hour of pumping (Figure 30). After about one day (1440 minutes) of pumping, the well yield (210 gallons per minute), arsenic concentration (20 µg/L), and pH (8.42 standard units) had stabilized (Figure 30). Specific conductance fluctuated from 550 to 510 µs/cm (Figure 30) for the following 3 days. Dissolved oxygen concentration stabilized at 5.6 mg/L (60.0 percent saturation) and water temperature stabilized at about 18.5 degrees Celsius. The results of this brief test indicate that, to obtain results most

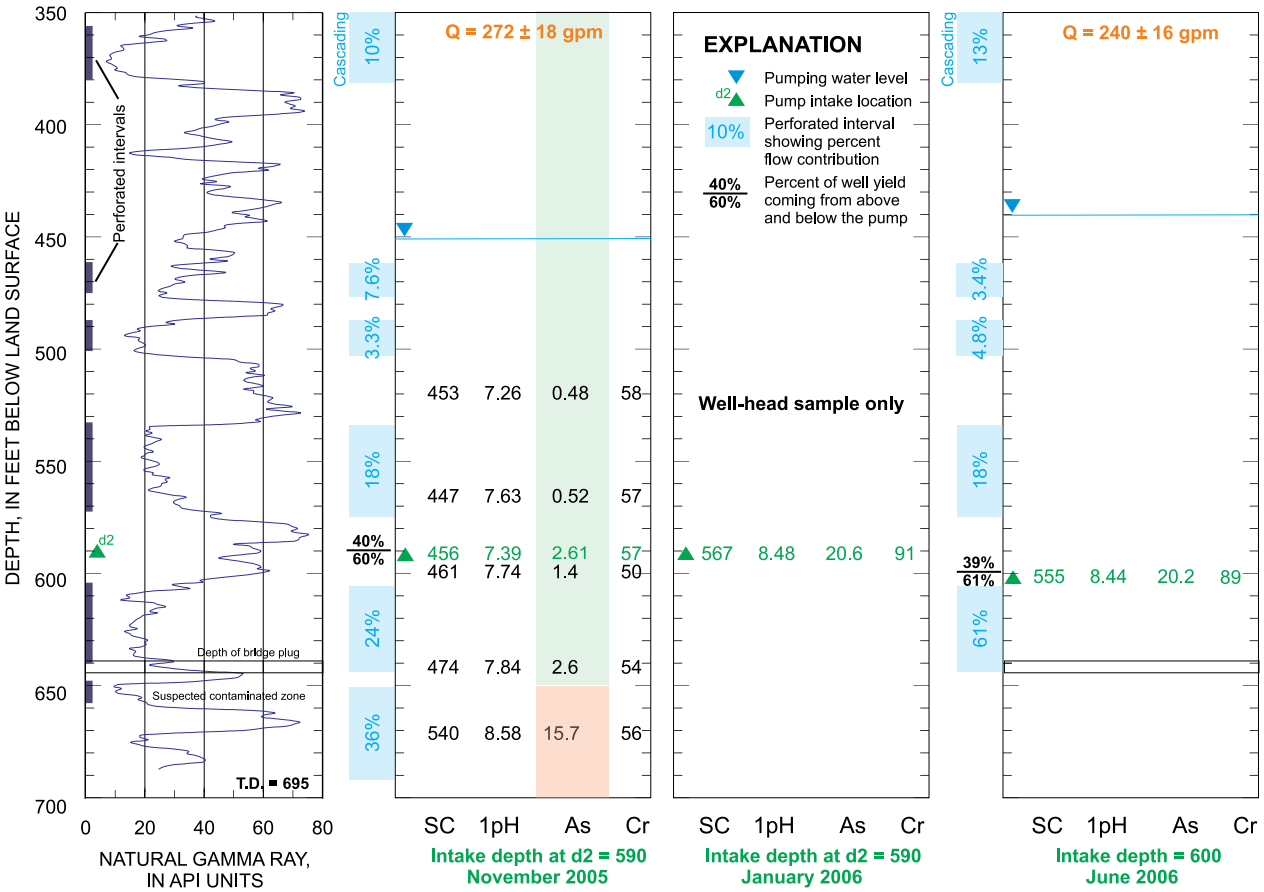


Figure 29. Results of attempted zonal isolation in Norman Well 36 by using a retrievable bridge plug. Concentration and field water property data in green are from well-head samples; those in black are from depth-dependent samples at the indicated depth. The placement of the plug had little effect on well-head water quality or well yield [Q, production rate in gallons per minute; SC, specific conductance in microsiemens per centimeter; As, dissolved arsenic in micrograms per liter; Cr, dissolved chromium in micrograms per liter; µg/L, micrograms per liter; API, American Petroleum Institute; gpm, gallons per minute; NQ, not quantifiable; %, percent].

representative of true production conditions, the Norman wells should be pumped for at least 24 hours prior to investigations of water quality and flow contribution with depth.

The water-quality logging ended after nearly 4 days, when the well had to be shut down after weekend rainfall lowered water demand. The well was rested over the weekend and restarted on Monday, June 19, 2006. After 3 days of continuous pumping, the well discharge was rerouted from the distribution pipe to a waste blow-off prior to tracer-pulse velocity profiling on Thursday, June 22, 2006. Depth-dependent samples were not collected.

Unfortunately, the installation of the bridge plug in Well 36 had no effect on well-head water quality. Compared to the well-head sample collected in January 2006, specific conductance and concentrations of arsenic and chromium each decreased by only 2 percent after installation of the bridge plug (Figure 29). The well yield estimate decreased by about 12 percent (Figure 29). Flow contribution percentages for each perforated interval changed little from measurements made in November 2005. The only substantial change in flow contribution was measured in the perforated interval (605-640 feet) just above the bridge plug (Figure 29). Apparently, this zone began producing more water to compensate for the production lost from the perforated interval below the pump intake. Cement-bond at the depth of the plug was good, so annular communication between zones above and below the plug is unlikely.

The placement of the bridge plug assumed that the isolated zone was not in close hydraulic connection with the producing zone just above the bridge plug. Although this assumption may be valid for the period of testing (Figure 30), this assumption is probably not valid over periods of years and decades. If the mudstone (640-645 feet, Figure 29) above the suspected contaminated zone is not laterally continuous or impermeable near the well, water from the suspected contaminated zone may be entering the well through the perforated zone just above the bridge plug. However, if the zones were hydraulically connected, mixing with relatively uncontaminated water in the upper zone should have had some mitigative effect on well-head arsenic concentration.

A more probable explanation of why well-head water quality remained the same is that the stretch factor applied to the depth-dependent samples may have been too great. If in reality minimal hose stretch occurred during sample runs, the depth of the suspected arsenic-contaminated water could have been overestimated by as much as 13 feet. The arsenic-contaminated water responsible for degradation of well-head water quality may originate from the perforated sandstone just above the bridge plug (605 to 640 feet, Figure 29). Therefore, the bridge plug may have been placed too deep to exclude the arsenic-contaminated water from production.

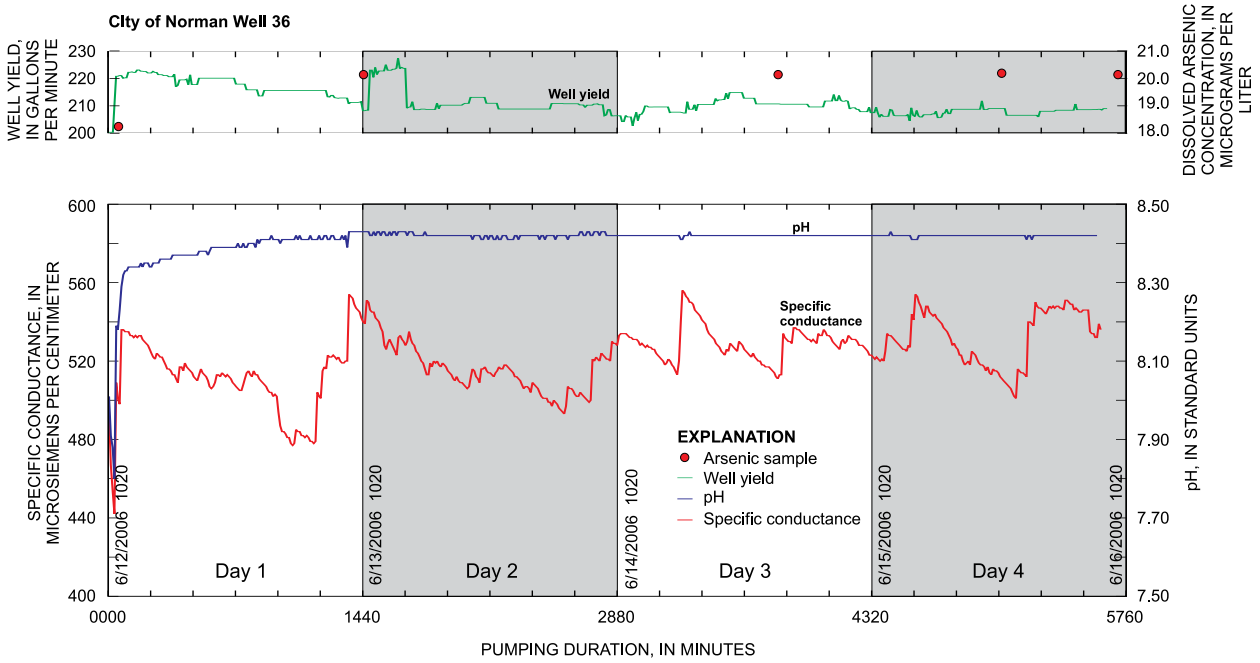


Figure 30. Log of well yield and selected well-head water-quality constituents at Norman Well 36 during an attempt at zonal isolation by using a retrievable bridge plug.

9.0 Summary

The City of Norman, Oklahoma, is one of many municipalities in the United States that is affected by a change in the EPA's National Primary Drinking Water Regulations that reduced the arsenic MCL from 50 µg/L to 10 µg/L in 2006. The City of Norman depends on ground-water from the Central Oklahoma (Garber-Wellington) aquifer, a multilayered sandstone, siltstone, and mudstone aquifer. Arsenic (and the associated metals chromium, selenium, and uranium) has been identified as a naturally occurring contaminant in the aquifer.

Historical arsenic concentrations of produced water from 32 active Norman public-supply wells ranged from less than 1 µg/L to 232 µg/L. Based on maximum detected arsenic concentrations in well-head samples, 11 of these wells could be deemed noncompliant under the old MCL of 50 µg/L arsenic. Of the 21 remaining wells, 10 additional wells, which account for about one-third of the total well-field production capacity, likely will be deemed noncompliant under the new arsenic MCL. Through 2003, two thirds of the wells in the Norman well field had produced at least one well-head sample with arsenic concentration greater than 10 µg/L.

The Norman well field was thought to be well-suited for the zonal-isolation strategy because (1) most Norman public-supply wells have a cement-annulus and gun-perforated openings, and (2) producing sandstone zones are commonly separated or compartmentalized by thick mudstones. The best candidates for successful remediation by zonal isolation are those wells that have (1) marginal well-head arsenic concentrations (near 10 µg/L), (2) wide variation in well-head arsenic concentrations, and (3) high water-production rates (greater than 200 gallons per minute). These wells are most likely to benefit from isolation of a single, high-arsenic zone and are the least likely to suffer from loss of production from that zone.

Based on historical well-head samples, some Norman wells with marginal arsenic concentrations were suspected of producing water from some zones with acceptably low arsenic concentrations and some zones with unacceptably high arsenic concentrations. If zones with elevated trace-element concentrations can be identified and sealed off from production, concentrations measured at the well head may be decreased to meet drinking-water regulations. To determine which wells were possible candidates for arsenic remediation by well rehabilitation, though, the flow contribution and water quality of each producing zone was measured in individual wells.

The water-quality data collected by the USGS well profiler were extremely useful as a qualitative tool for identification of zones that may degrade water quality in the Norman wells. The depth-dependent sampling method can identify the depth in the well bore at which water becomes unsuitable for public supply (exceeds the MCL), even without any data on flow contribution. The USGS well-profiler method, as compared to traditional methods, can be considerably less expensive and requires less down-time of the well. In terms of data quality, the most important advantage of the USGS well profiler is that all data collection is performed under true production conditions.

As part of the investigation of changes in water quality with depth in the Norman area (southern Central Oklahoma aquifer), an undeveloped site in Norman was selected for drilling, logging, coring, and water sampling in a test hole similar to that of earlier studies. The selected test-hole site was in northern Norman near the Little River. The test hole, referred to as the arsenic test hole, was 728 feet deep and penetrated about 50 feet of the Hennessey Group, and nearly half of the total thickness of the Garber Sandstone and Wellington Formation.

Rock material was sampled by coring from 302 to 536, 568 to 598, 615 to 636, 640 to 652, and 668 to 686 feet. Most of the red sandstones in the core were made up of very-fine to fine-grained sand that was moderately well to well sorted with respect to framework grains. The sandstones also contained red mud (matrix composed of clay and silt-sized mineral matter) between the framework grains. Thin to moderately bedded layers of conglomerate were present in the core and contained clasts of dolomite and mudstone. Locally, mudstone layers contained features that were indicative of soil forming episodes during the Permian period (Permian paleosol formation). Evidence for secondary iron mobilization was present in the sandstone, conglomerate, and mudrock preserved in the core.

Ground-water quality samples were analyzed from seven predominantly sandstone zones (ranging from 12 feet to 40 feet in thickness) in the test hole. Arsenic concentrations exceeded the MCL of 10 µg/L in all zones where pH was greater than 8.5 and specific conductance was greater than 600 µS/cm. Sandstone Zones 2 (416-456 feet) and 3 (488-502 feet) were separated by only 32 feet but had different water types. The δ^2H of samples from the test hole indicated a substantial change in supposed water age from Zone 2 (younger) to Zone

3 (older). This gap between younger and older water corresponds to the gap between detected and non-detected arsenic in zonal water samples.

The transition in water quality and supposed water age from Zone 2 to Zone 3 could be an indication that the intervening mudstone is a regional barrier to the vertical flow of ground water. Alternatively, the contrast could mean that zones below a depth of 460 feet are richer in exchangeable clays. The contrast in water quality between Zones 2 and 3 also could be related to the presence of a basal carbonate-clast conglomerate (represented by a zone of increased resistivity) in Zone 2. According to one study, conglomerates in the Central Oklahoma aquifer contain large concentrations of dolomite, arsenic, and iron in the solid phase relative to the other lithofacies in the aquifer system. If dolomite supply or arsenic availability are limiting factors in the chemical evolution of water when it reaches this depth, the presence of the conglomerate could accelerate arsenic release.

Eleven wells and one test hole were sampled to describe changes in water quality and estimated water age with depth in the Norman well field. Earlier studies determined a relation between δ^2H and water age determined from tritium and radiocarbon dating in the Central Oklahoma aquifer. Though the relation is not perfect, water age increases with decreasing δ^2H. According to this relation, the youngest water was found in Wells 18, 02, and the shallowest sampled zones of the arsenic test hole. The oldest water sampled was found in Well 07 and intermediate sandstone zones of the arsenic test hole. The ages of water in the test hole could range from only a few hundred years before present to more than 30,000 years before present. This wide range in age, as compared to well samples, is contradictory to ages expected from review of modeled flow paths and could be more characteristic of the natural, undeveloped flow system. Completed wells, after years of seasonal pumping stresses, could develop a narrower age signature due to mixing by intra-borehole flow between zones when the well is not being used.

Approximate ages of water (times since recharge) in wells are on the order of hundreds to tens of thousands of years. When plotted in order from most proximal to most distal (from the recharge area) along a typical flow path, the historical arsenic data for these wells express upward trends in minimum, median, and maximum detected arsenic concentrations with distance along the flow path. This finding validates the aquifer conceptual model developed by the USGS NAWQA Program.

Well-head arsenic concentrations measured in this study compare favorably with historical well-head samples at each well. All well-head arsenic samples collected in this study were within the range of historical well-head concentrations. Most depth-dependent arsenic samples also were within the range of historical well-head concentrations. Only Wells 02 and 36 had depth-dependent arsenic samples that were outside the range of historical well-head concentrations.

Most of the selected Wells (06, 07, 13, 15, 18, 23, and 31) showed elevated or near-elevated arsenic concentrations at all depths in the well. For these wells, well-modification techniques would be ineffective in lowering well-head arsenic concentrations to less than 10 µg/L. Wells 02, 05, 33, and 36, however, showed some potential for successful application of well modification techniques for arsenic remediation. In Wells 02, 05, and 36, elevated arsenic concentrations were only detected in depth-dependent samples from the bottom of the well. In Well 33, elevated arsenic concentrations were not detected at any depth.

Wells 05 and 36 were selected for repeated depth-dependent sampling to determine the effects of pump intake relocation on water quality and well yield. In Well 05, the initial pump intake setting was at 640 feet below land surface. The pump intake was moved to 610, 560, and 505 feet below land surface. In Well 36, the initial pump intake setting was at 670 feet below land surface. The pump intake was moved to 650 and 590 feet below land surface. Both wells showed improvements in water quality as the pump was moved to the highest locations in the well. In Well 05, arsenic concentration decreased by about 32 percent and well yield decreased by 12 percent. In Well 36, the arsenic concentration at the well head decreased by 84 percent and well yield increased by 13 percent. However, after several days of continuous pumping, additional samples from well 36 indicated that the improvement in well-head water quality was only temporary. To obtain results most representative of true production conditions, the Norman wells should be pumped for at least 24 hours prior to investigations of water quality and flow contribution with depth.

The main objective of zonal isolation in this study was to eliminate production from all zones that contribute water of elevated arsenic concentrations. Only the deepest zone in Well 36 (648-658 feet) was suspected of contributing elevated arsenic concentrations (greater than 60 µg/L) to the well. A retrievable bridge plug was installed in Well 36 to isolate the suspect zone from production. Unfortunately, the installation of the bridge plug in Well 36 had no effect on well-head water quality. The bridge plug may have been placed too deep to exclude the arsenic-contaminated water from production.

10.0
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