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HYDROGEOLOGY FIELD MANUAL

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1996

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1. Introduction

This manual describes how wells can be used to define physical and chemical properties of aquifers. Topics include constructing wells, performing aquifer tests, and sampling wells. Those tasks are of major importance in planning the development or remediation of an aquifer. For example, the hydraulic conductivity of an aquifer dictates the rate at which contaminants migrate in groundwater. It also influences the productivity of an aquifer. Drawdown in a pumping well is used to evaluate the potential yield of a water supply well. Field measurements of basic water quality parameters such as dissolved oxygen, electrical conductance, and pH can be used to screen for the possible presence of contamination before samples are submitted to a laboratory for a more detailed analysis. For example, an electrical conductance value can be used to estimate the concentration of total dissolved solids in groundwater, which may be elevated near a leaking waste storage facility. Once collected, groundwater samples can be chemically analyzed to determine the presence or absence of contaminants beneath a site, or to monitor the status of a contaminated aquifer during the implementation of remediation measures. A list of frequently cited references appears at the end of the manual for further reading.

2. Constructing Monitoring Wells

Monitoring wells are used to obtain groundwater samples and water level elevations in aquifers. They can also be used to estimate the hydraulic characteristics of

aquifers. The wells are constructed by advancing a boring with a drilling rig, installing a well casing and screen, and backfilling the annulus between the casing and the wall of the borehole (Figure 2-1).

While a borehole is being drilled, soil and rock samples should be collected at frequent depth intervals. Careful observations of drill cuttings can guide the collection of undisturbed samples. Soil and rock samples are used to define the stratigraphy of a study area. In environmental investigations, these samples can be chemically analyzed to determine the extent of subsurface contamination. All cuttings and samples obtained during drilling should be carefully described in a field log book.

Soil and rock samples are described according to several physical characteristics, including color, texture, degree of compaction, density, and moisture content. The predominant grain size is used to name an unconsolidated sample. Other observable grain sizes may be used as adjectives. For example, a clay containing a minor fraction of sand may be described as a sandy clay.

Drilling Methods

The method chosen for drilling a monitoring well depends on several factors, including subsurface conditions, equipment availability, versatility of the drilling method, drilling cost, site accessibility, installation time, ability to preserve natural conditions, and ability to obtain reliable samples (EPA, 1994). The most common drilling methods include hollow-stem augers, solid-stem augers, cable tool, air rotary, water rotary, mud rotary, dual-wall reverse circulation, sleeve- or hammer-driving, and jet percussion. In environmental investigations involving contaminated soil and rock, the drilling equipment should be steam-cleaned between holes. This requirement pertains to all of the drilling methods described below.

Hollow-Stem

The hollow-stem auger method is frequently used to install monitoring wells in unconsolidated or poorly consolidated materials, but is inappropriate for solid rock (EPA, 1994). A cutting head is attached to the first auger flight, and additional flights are added as the augers are rotated downward. A plug attached to a rod inside the hollow stem prevents soil from entering the interior of the stem. Cuttings are moved upward along the sides of the borehole. Periodically, the plug is removed and a sampling device is lowered down the auger to extract an undisturbed sample. Split-spoon or Shelby tube samplers are commonly used for this purpose. A split-spoon sampler is a hollow tube comprised of two halves, whereas the Shelby sampler is a one-piece hollow tube. With a hammer device, samplers are driven into the soil ahead of the bit.

The hollow stem temporarily cases the borehole, allowing the well screen and casing to be inserted once the desired depth is reached. This function is advantageous because it prevents a collapse of the borehole that might occur if the augers had to be withdrawn before installing the well casing. Potential disadvantages of the hollow-stem method include cross contamination of subsurface material (drill cuttings are conveyed along the length of borehole), sand and gravel heaving into the auger, and

clay smearing along the borehole wall.

Solid-Stem

In contrast to hollow-stem augers, solid-stem augers need to be removed from the borehole to collect samples and install casing. Unconsolidated deposits that lack cohesion are not amenable to the solid-stem method. Most unconsolidated and poorly consolidated deposits have a tendency to collapse when augers are removed from the saturated zone (EPA, 1994). Therefore, with the solid-stem method, undisturbed samples of unconsolidated materials cannot be collected below the water table.

Cable Tool

Cable tool drilling has been in use for about 4,000 years (Driscoll, 1986). However, modern cable tool equipment can drill a well within a few days, whereas the earliest equipment often required several years. Cable tool drilling machines operate by repeatedly lifting and dropping a heavy string of drilling tools into the borehole. The drill bit breaks and crushes consolidated rock into small fragments and loosens unconsolidated material. The reciprocating action of the tools mixes the crushed or loose particles with water to form a slurry at the bottom of the hole. Water is added if none is present in the formation. Periodically, the slurry is removed by a sand pump or bailer.

In situ samples can be obtained by attaching a sampling tube to the drilling string. A hammering action drives the sampling barrel into the ground. This form of sampling does not work in consolidated bedrock, but is applicable to overburden material. Borehole instability can be overcome by driving casing ahead of the sampling zone. Overall, the cable tool method is relatively inexpensive, but slow compared to rotary drilling methods.

Air Rotary

Rotary drilling involves the use of circulating fluids to remove drill cuttings and maintain an open hole as drilling progresses. Borehole caving is a potential problem in unconsolidated formations. However, driving casing as the borehole is advanced can resolve this problem. In the air rotary method, air is forced down the drill pipe and back up the borehole to remove the drill cuttings. Best suited for hard rock, the air rotary method is not often used for environmental investigations because it cannot yield representative samples (EPA, 1994). Injection of air into a borehole may alter the natural properties of the subsurface.

Water and Mud Rotary

Water and mud rotary drilling involves the introduction of fluids through the drill pipe to maintain an open hole, provide lubrication to the drill bit, and remove drill cuttings (EPA, 1994). Water rotary is rapid and effective for most materials, but the fluid tends to mix with the surrounding formations and groundwater. The identification of water-bearing zones is hampered by the addition of water into the borehole. In clay-rich sediments, the water may form a slurry that plugs the formation, making it difficult to develop the well. In fractured rock, it may be difficult

to maintain effective water circulation because of water losses to the subsurface.

Mud rotary is used instead of water when it is extremely difficult to maintain an open borehole. However, the additives create a high potential for affecting aquifer characteristics and groundwater quality. In environmental investigations, the mud should be limited to locally occurring clays. An additional problem with mud rotary is that filter cakes which tend to form on the sides of boreholes reduce porosity and may inhibit groundwater flow.

Dual-Wall Reverse-Circulation

The dual-wall reverse-circulation rotary method employs a double-wall drill pipe. Air or water is forced down the outer casing and circulated up the inner drill pipe. Cuttings are lifted upward through the pipe to the surface. Either a hammer or bit can be used to cut the formation. The primary advantage of the dual-wall method is that it allows continuous sampling of the subsurface and eliminates problems associated with lost circulation and borehole instability (EPA, 1994). However, this method also requires relatively expensive drilling equipment and a large borehole to accommodate the dual-wall pipe.

Driven Wells

Driven wells consist of a steel well screen that is welded or coupled to a steel casing. The well screen and casing are forced into the ground by hand using a weighted drive sleeve, or with a drive head mounted on a hoist (EPA, 1994). As the well is driven, new sections of casing are attached in four- or five-foot sections. Diameters of driven wells are small, most falling in the range of one to four inches (Todd, 1980). The method is better for shallow wells, less than 50 feet in depth.

It is difficult or impossible to drive wells through dense clays or materials that contain cobbles and boulders. If driven, the well may be destroyed in these environments. In addition, silts and clays can clog the well screen. Another disadvantage of the driven-well method is that it cannot yield representative samples of materials that are penetrated.

Jet Percussion

The jet percussion method employs a wedge-shaped drill bit attached to the end of the drill pipe. The bit is lifted and dropped while rotating. Water is forced under pressure down the drill pipe and discharged through ports on the sides of the drill bit. It then moves up the annular space between the drill pipe and borehole wall, carrying cuttings to the surface. The jet percussion method is limited to unconsolidated or soft formations. Disadvantages include disturbing the formation and an inability to obtain representative samples (EPA, 1994).

Well Construction

Once a hole is drilled, casing and annular material are added to complete the well. The casing consists of segments of solid pipe, whereas the screen is a slotted section of pipe. Segments of casing and screen are usually threaded, glued, or welded

together. Threaded joints require a rubber O-ring to prevent leakage. Glue is less preferable than other coupling alternatives because it can leach into the groundwater.

The well screen should be designed to allow water but not sediment to enter the well. Screen lengths vary depending on the purpose of a well, but generally should not exceed 10 feet (EPA, 1994). Long well screens provide vertically composite rather than discrete samples. However, the screened portion of a well must allow sufficient water to enter for sampling or piezometer tests.

Casing material is chosen on the basis of cost, durability, and reactivity with water. Teflon is most costly, least durable, and most inert. Stainless steel is most durable, moderate in cost, and essentially inert. PVC, which is available in most hardware stores, is least costly and most frequently used.

The annular space between the borehole wall and casing should be filled to prevent passage of formation materials into the well. Many bedrock wells do not require screens and thus do not require filter packs. However, most wells do require filter packs and a screened length of casing. Filter pack material should be chemically inert. Industrial grade silica sand or glass beads are appropriate. The Environmental Protection Agency (EPA) recommends that filter pack constituents be well rounded and of uniform size. The filter pack should extend at least two feet above the top of the well screen. A tremie pipe can be used to install the filter pack and other annulus materials to prevent bridging between the casing and borehole wall.

Above the filter pack, proper sealing of the annular space between the well casing and the borehole wall is required to prevent a hydraulic connection between surface water and groundwater near the well. The sealants should be chemically inert and essentially impermeable. A two- to three-foot plug of bentonite is usually placed directly over the filter pack. The bentonite can be emplaced by dropping pellets down the annulus of the borehole. When properly emplaced and hydrated, the pellets form a clay plug. A mixture of cement and bentonite is often used to fill the annulus above the bentonite seal.

Monitoring wells are commonly completed above-ground or flush with the ground. In either case, measures should be taken to prevent infiltration of surface runoff into the well annulus and prevent damage or vandalism of the well. Concrete should be installed at the ground surface to hold the well in place. Anchored in concrete, a locking protective casing can be placed around the well casing to prevent damage or unauthorized entry.

After a well is installed, it should be developed to remove any sediment blocking the well screen. This procedure improves yield and creates a well capable of producing low-turbidity samples. Turbid samples may interfere with the chemical analysis of groundwater. Well development is usually performed with a cylindrical plunging device or swab that is repeatedly lowered and raised through the water column. The plunging action forces water into and out of the well screen, thereby removing lodged silt and clay particles. Loose particles are transported into the well and removed with a bailer or pump (Bouwer, 1978). Other methods for developing a well include jetting with air or water and over pumping. Over pumping involves pumping at a higher rate

than the design rate.

Typically, a borehole contains a single monitoring well. However, it is also possible to place multilevel sampling devices in a single borehole. These devices may consist of a series of flexible tubes tapping the sidewall of a pipe at different depths. Alternatively, multiple pipes can be placed in a single borehole, but screened at different depths.

Exercises

1. Examine the cuttings from boreholes 1 through 4. Describe the color, texture, and degree of compaction of each sample.
2. Describe the probable environment of deposition for the samples in problem 1.
3. What do the samples indicate about aquifer yield? Would the formation be suitable for high-capacity production wells?
4. Examine the well completion forms attached to the end of this section. Describe the hydrogeologic setting.
5. Using a bailer, collect a sample from each well and describe its turbidity. Use a swab to develop the well. Collect another sample and compare its turbidity to that of the first sample.

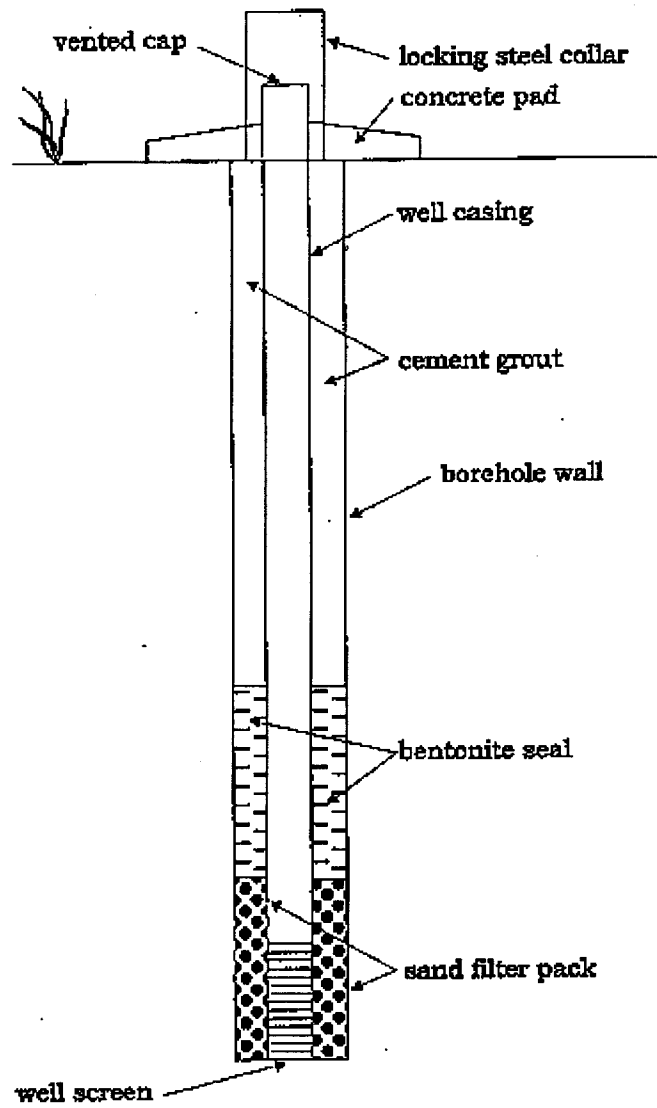


Figure 2-1. Groundwater monitoring well (modified from Hudak, 1994).

Well Completion Form

SOP #2150

PAGE OF

MONITOR WELL INSTALLATION		
Client:	Job No:	Date Drilled: Well No: <u>1</u>
Site: <u>WRS</u>	Elevation: Pad:	Top of Steel Casing:
Total Depth: <u>16.5 ft</u>	Casing Size & Type: <u>2 in PVC</u>	Screen Size:
Comments:		
Depth	Sample Description	Completion Data
0	dark brown clayey loam	casing: 0-12 ft blank 12-16 ft screened 16-16.5 ft blank
0-12	gravel and clay	
5	dense sandy clay; brown; moist	
10		
15	water @ 15.5 ft	
15-16.5	gravel and clay	
20		
25		

Well Completion Form

SOP #2150

PAGE OF

MONITOR WELL INSTALLATION			2
Client		Job No.	Date Drilled
Site: <u>WRS</u>		Excavator	Top of Steel Casing
Total Depth	<u>10 ft</u>	Casing Size & Type	<u>2 in PVC</u>
Comments			
Depth	Stratigraphic	Sample Description	Geologic Data
		dark brown clay; moist	casing:
		dense sandy clay; brown; dry	0-7.5 ft blank
5		gravel and clay	7.5-10 ft screened
		clay with gravel; angular pebbles; dry	drilled to 12 ft;
10		sandy clay; brown; dry	borehole caved
		gravel and clay	
15			
20			
25			

Well Completion Form

SOP #2150

PAGE OF

MONITOR WELL INSTALLATION			
Client	Job No.	Date Drilled	Well No. 3
Site: WRS	Elevation	Pad	Top of Steel Casing
Total Depth: 20 ft	Casing Size & Type: 2 in PVC	Screen Size:	
Comments			
Depth	Symbol Stratigraphy	Sample Description	Completion Data
0		dark brown clay loam; moist	casing: 0-16 ft blank 16-20 ft screened
1		brown clayey gravel	
5		dense brown clay with minor sand content; dry	
10		sand content increases downward	
15		mottling @ 14.5 ft	
20		dense gray clay	
25		gray shale	

3. Measuring Water Levels in Wells

Once constructed, groundwater monitoring wells should be surveyed to determine their X-Y coordinates and elevation. A surveyed reference mark should be placed on the top of the well casing. This reference point should be used for all depth-to-water measurements. Its height should be determined within 0.01 feet in relation to mean sea level.

The depth to water in a well can be measured with various instruments, including a cloth tape and popper, a steel tape coated with chalk, an acoustic well probe, electrical sensors, pressure transducers, air lines, and float devices. Regardless of the equipment used, the depth to water should be measured to the nearest 0.01 feet. The measuring devices should be chemically inert and not prone to sorption or desorption. They should be thoroughly decontaminated after each measurement to prevent cross-contamination of the groundwater. It is good practice to take the first measurement at the least-contaminated well. Subsequent measurements should be made at progressively more contaminated wells.

Water levels measured in monitoring wells or piezometers can be used to construct a water table or potentiometric surface map showing the distribution of hydraulic head in an aquifer. Water level measurements used to define a single potentiometric surface should be made within a time interval less than 24 hours. A shorter time interval may be necessary if water levels are changing due to natural or artificial stresses, such as recharge from precipitation or pumping.

At least three wells are required to establish the local direction of groundwater flow in an aquifer. A rapid determination of the flow direction can be made using the three-point method. On a scaled map, construct a triangle using the three wells as vertices. Next, identify the side of the triangle connecting the highest and lowest water level elevations. Find the point on this line having an elevation equal to that measured in the third well. This can be accomplished by subtracting the lowest water level elevation from the intermediate value, and dividing the result by the difference between the highest and lowest water level elevations. Multiply that quotient by the length of the line. The result is the distance along the line, measured from the well with the lowest water level elevation, at which the intermediate elevation is inferred. Connect that point with the third well. The direction of groundwater flow is perpendicular to that line, in the direction of decreasing hydraulic head.

By drawing the perpendicular to the point with the lowest water level elevation, the magnitude of the horizontal hydraulic gradient can be estimated. Measured in the direction of groundwater flow, the hydraulic gradient is the difference in hydraulic head among two points divided by the distance between the points. Subtract the lowest water level elevation from the intermediate value, and divide that difference by the length of the perpendicular. The result is the hydraulic gradient, expressed as a dimensionless quantity.

Groundwater flow directions should be determined from water levels measured in wells screened in the same hydrostratigraphic position. In heterogeneous geologic settings, long well screens can intercept stratigraphic intervals with different groundwater flow directions and different hydraulic head values.

The hydraulic gradient, in conjunction with the hydraulic conductivity and effective porosity, can be used to estimate the groundwater velocity in an aquifer. Hydraulic conductivity quantifies the ease with which a fluid is conducted through a porous medium. It ranges from approximately 10-8 feet per day for unfractured shale and igneous rocks to 105 feet per day for gravel or cavernous limestones.

Effective porosity is the percentage of a rock or soil sample that consists of interconnected pores through which water can flow. Typical values range from zero for granite to 50 percent for clay. Although clay has a high porosity, the pore spaces available for flow are small. Consequently, the hydraulic conductivity of clay is rather low.

Groundwater velocity, \bar{v} , is defined as

$$\bar{v} = \frac{K\bar{i}}{n_e}$$

where K is the hydraulic conductivity, \bar{i} is the hydraulic gradient, and n_e is the effective porosity of the aquifer.

Exercises

1. Survey monitoring wells 1 through 4. Construct a scaled map showing the positions of the wells. Also record the elevation of the surveyed reference points. Place a north arrow on your map.
2. Measure the depth to water in each well. Record your data on the attached groundwater level data form. Calculate the water level elevation using the elevation of the surveyed reference point. Describe the occurrence of groundwater beneath the site and the probable direction(s) of groundwater flow.
3. Solve a three-point problem for the monitoring wells in Figure 3-1. Draw an arrow showing the inferred direction of groundwater flow. Express this direction in degrees (0° = N; 90° = E; 180° = S; 270° = W). Calculate the horizontal hydraulic gradient. Assuming a hydraulic conductivity of 1.5 feet per day and an effective porosity of 0.30, what is the groundwater velocity?

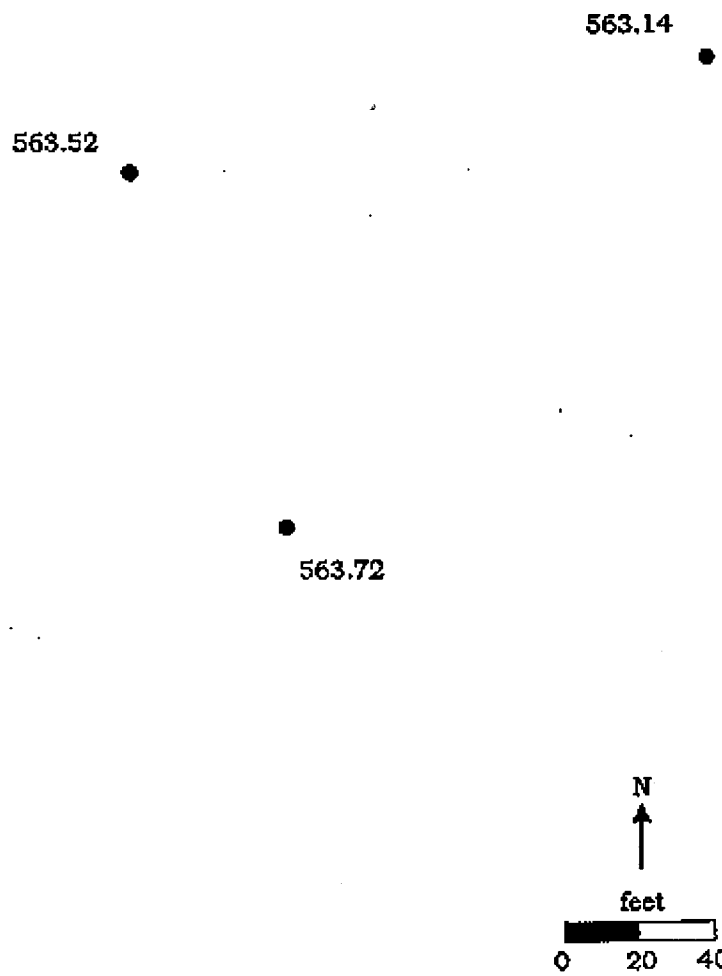


Figure 3-1. Three-point problem; water table elevations are in feet above mean sea level.

4. Slug Tests

Slug tests are used to determine the horizontal hydraulic conductivity of a distinct geologic horizon under in situ conditions. Compared with pumping tests, slug tests can be performed quickly and at relatively low cost (because pumping and observation wells are not required). In a slug test, a known volume (slug) of water is suddenly added to or removed from a well, and the decline or recovery of the water level is measured at closely-spaced time intervals.

Hvorslev (1951) devised a method for computing the hydraulic conductivity of a formation from slug test data. In that method, the unrecovered head difference, normalized to the initial head difference, is plotted on a logarithmic scale against time, which is plotted on a standard arithmetic scale. A line is fitted to the data, and the time at which the normalized unrecovered head difference = 0.37 is determined. This time, T_0 , is then used with the following equation to estimate hydraulic conductivity, K :

$$K = \frac{r^2 \ln(L/R)}{2LT_0}$$

where r is the radius of the piezometer, L is the length of the intake (screened interval), and R is the radius of the intake. The analysis assumes an isotropic, homogeneous medium.

The hydraulic conductivity value obtained from a slug test applies only to the aquifer deposits in the vicinity of the well screen. In contrast, a pumping test gives a hydraulic conductivity value indicative of broader portion of the aquifer.

Exercises

1. Calculate a hydraulic conductivity value using the slug test data in Table 4-1.
2. Perform a slug test on each well at the Water Research Station by submerging a bailer in the water column, allowing the water level to stabilize, and suddenly removing the bailer. Collect water level data at approximately 30-second time intervals until the water level has nearly risen to the initial level. Record your data on the attached slug test data form. Use the Hvorslev method to calculate hydraulic conductivity.

Table 4-1. Slug Test Data¹

Time (seconds)	Depth to Water (feet)
0	5.00
13	4.96
19	4.93
25	4.91
29	4.84
41	4.75
52	4.62
70	4.54
99	4.46
108	4.37
132	4.27
151	4.05

¹ Rising-head test; static depth to water = 4.05 feet.

5. Pumping Tests

Pumping tests are among the most reliable methods of determining aquifer transmissivity and storativity (EPA, 1991). However, such tests require a greater degree of activity and expense than alternative methods and are not always justified for all levels of investigation. These tests involve pumping a well at a constant rate and measuring drawdown in nearby wells. Pumping tests allow hydrogeologists to directly observe the response of an aquifer to pumping.

Prior to starting the test, water levels should be carefully monitored in the pumping and observation wells. The purpose of this preliminary monitoring is to determine how water levels are fluctuating over time under pre-pumping conditions. For example, water level fluctuations can be caused by changes in barometric pressure or recharge from precipitation. The pre-pumping water level trend must be used to correct drawdowns measured during the pumping test. The corrected values should reflect only the effects of the pumping well.

During a pumping test, all depth-to-water measurements should be made from the surveyed reference points. The rate of drawdown is normally rapid at first, and then slower as time progresses. Therefore, readings should be taken more frequently at first, for example every 30 seconds. For a confined aquifer, a pumping test may last 24 hours or less (Driscoll, 1986). Pumping tests in unconfined aquifers normally require more time, to account for delayed gravity drainage. Observation wells should be located close enough to the pumping well to be impacted during the test, and screened in the same aquifer as the pumping well. Ideally, both the pumping and observation wells should fully penetrate the aquifer. Depth-to-water measurements in the observation wells and pumping well should be made on the same time schedule.

The pump and accessories must be reliable, capable of drawing water at a constant rate through the duration of the test. If a pump fails, the data may be insufficient to obtain reliable estimates of aquifer properties. Throughout the test, discharge must be measured from the pump, and the flow rate must be adjusted to maintain a constant rate. There must also be a means for conveying water into a holding tank or away from the test site. This is especially important for shallow unconfined aquifers that could be recharged by discharge from the pumping well. Such recharge would interfere with the interpretation of the test results.

The Theis method is used to measure transmissivity and storativity for non steady flow without vertical movement. It involves plotting the time-drawdown data recorded at an observation well on logarithmic paper. A type curve is superimposed on the plotted field data, and any point in the field of overlap between the two graphs is used to obtain values for $W(u)$, $1/u$, s , and t (defined below). The Theis method involves the following assumptions (Lohman, 1979; Heath, 1984): (1) the aquifer is homogeneous and isotropic, (2) the boundaries of the aquifer are beyond the effects of the pumping well, (3) the well penetrates the thickness of the aquifer, (4) the well has a small diameter, (5) the aquifer has a constant storativity, and (6) water withdrawn from the aquifer is derived entirely from storage and is discharged instantaneously with the decline in hydraulic head. The equations used to relate drawdown to transmissivity and storativity are:

$$s = \frac{Q}{4\pi T} W(u) \quad ; \text{ and}$$

$$u = \frac{r^2 S}{4Tt}$$

where s is drawdown, Q is discharge from the pumping well, T is aquifer transmissivity, $W(u)$ is the well function, r is the distance between the pumping well and observation well, S is aquifer storativity, and t is time since pumping began. Values for $W(u)$ are tabulated in most hydrogeology textbooks (for example, Driscoll, 1986; Fetter, 1994). The Theis equations cannot be solved directly, but are amenable to solution with manual curve matching techniques or computer software (Fetter, 1994).

The Theis method cannot be applied directly to time-drawdown data measured in unconfined aquifers. If the aquifer is fine-grained, water is released slowly over a period of hours or days, not instantaneously with the decline in head (Heath, 1984). As a result, the value of S determined by the Theis method for a short test in an unconfined aquifer may be too small. If the pumping rate is large and the observation well isn't near the pumping well, dewatering of the aquifer may be significant, and the assumption that the transmissivity of the aquifer is constant is invalid. However, the effect of dewatering the aquifer can be removed with the following equation (Heath, 1984):

$$s' = s - \frac{s^2}{2b}$$

where s is the observed drawdown in the unconfined aquifer, b is the aquifer thickness, and s' is the drawdown that would have occurred if there was no dewatering.

After the end of a pumping test, the pump is shut off and recovery measurements are made in the test wells. In theory, the water levels will recover at the same rate they fall. Recovery data can be used to derive an additional estimate of aquifer transmissivity, which may be better than the first estimate. In some cases, there are uncontrolled variations in the pumping rate during the pumping test which affect the drawdown. However, such variations do not affect the recovery rate. The flow rate used for the recovery data is the mean discharge for the entire pumping period (Todd, 1980). The differences between water levels at the end of pumping and after various times since pumping stopped are plotted as a function of the time since pumping stopped.

Ideally, pumping tests should involve at least one observation well. Using only a

pumping well can be problematic because drawdown in the well will likely exceed that in the adjacent aquifer. The difference between drawdowns in the well and aquifer is the well loss. The efficiency of a well is defined as the ratio of the drawdown outside the well to the drawdown inside the well. (Observation wells are required to estimate drawdown in the aquifer.) In practice, a 70 to 80 percent efficiency is considered good.

Heath (1984) reports an approximate solution for transmissivity in an aquifer test involving only a pumping well. It assumes that well loss increases total drawdown in the pumping well, but does not affect the rate of change in the drawdown with time. The solution is

$$T = \frac{2.3Q}{4\pi\Delta s}$$

where Δs is the drawdown that occurs over one log cycle on a semilogarithmic plot of time (log scale) and drawdown (arithmetic scale). A single-well pumping test does not yield information on the storativity of the aquifer.

Another parameter that can be obtained from a single-well test is the specific capacity of a pumping well. Specific capacity is a basic measure of the performance of a well, with higher values signifying a greater yield capability. The specific capacity of a pumping well is defined as

$$S_c = \frac{Q}{s}$$

where s is the stabilized drawdown in the well.

Exercises

1. Use manual curve matching or the AQTESOLV computer program to estimate hydraulic conductivity and storativity for the data in Table 5-1.
2. Conduct a pumping test by pumping water from one of the wells at the Water Research Station and obtaining water level measurements from all four wells. After the pump is shut off, obtain recovery data from each well impacted by the pumping test. Record your data on the attached pumping/recovery test data sheets. Use an appropriate method to estimate hydraulic conductivity and storativity, if possible.
3. Calculate the specific capacity of the well that was pumped in problem 2.

Table 5-1. Pumping Test Data¹

Time (minutes)	Drawdown (feet)
0	0.00
1	0.66
1.5	0.87
2	0.99
2.5	1.11
3	1.21
4	1.36
5	1.49
6	1.59
8	1.75
10	1.86
12	1.97
14	2.08
18	2.20
24	2.36
30	2.49
40	2.65
50	2.78
60	2.88
80	3.04
100	3.16
120	3.28
150	3.42
180	3.51
210	3.61
240	3.67

¹ Confined aquifer; pumping rate = 26.8 cubic feet per second; distance to observation well = 250 feet.

6. Groundwater Sampling

The chemical composition of groundwater depends on the composition of precipitation, biological and chemical reactions occurring on the land surface and in the soil zone, and the mineral composition of the aquifer and confining beds through which the water moves (Heath, 1984).

The most abundant dissolved solids in natural groundwater are Ca^{2+} , Na^{+} , Mg^{2+} , K^{+} , HCO_3^{-} , SO_4^{2-} , Cl^{-} , and SiO_2 (Fetter, 1994). All of these ions (and SiO_2) are usually present at concentrations greater than 1 mg/l. Routine chemical analyses of water samples report these constituents among others. Secondary ions include iron, nitrate, fluoride and boron. The major gases dissolved in groundwater are oxygen and carbon dioxide. Nitrogen is also present. Minor gases include hydrogen sulfide and methane.

Once a monitoring well is installed and developed, it can be used to sample groundwater. Two steps should be taken before collecting a sample. First, the depth to water and total depth of a well should be measured from the surveyed reference point. Second, the well should be evacuated to remove stagnant water, which may not be representative of flowing groundwater in the aquifer. The standing water may contain foreign material that was inadvertently or deliberately introduced from the land surface. In a high-yielding ground water formation, and where there is no stagnant water in the well above the screened section, evacuation prior to sample withdrawal is not as critical (EPA, 1991). However, the EPA recommends that all wells be evacuated prior to obtaining a groundwater sample. Well evacuation is performed by removing three times the volume of water standing in the well casing. The evacuation procedure can be performed with a low speed pump or bailing device. The water level in an evacuated well should be allowed to recover to the original level before sampling.

The three broad categories of groundwater sampling devices are bailers, positive displacement pumps, and suction lift pumps. In all cases, the equipment should be constructed of inert material which will not alter analyte concentrations. This could occur if constituents of the sampling device leach into groundwater or absorb compounds from the sample. The sampling method should also cause minimal sample agitation and minimize contact between the sample and the atmosphere. It should not allow volatilization or aeration of samples, particularly when volatile organic compounds are present.

Bailers are relatively inexpensive and easy to use, requiring no external power source. However, they are also time-consuming and labor intensive. Another disadvantage is that the transfer of water to a sample container may alter the chemistry of the sample due to volatilization or aeration. It is also difficult to determine the exact location in the water column from which a bailed sample was collected. Pumps can be used to overcome some of the problems associated with bailers.

Samples should be collected as soon as possible after a well is purged. Ideally, the rate of sample collection should be approximately the same as the actual groundwater flow rate. Low sampling rates, approximately 0.03 gallons per minute, are suggested by the EPA. Samples should be placed in clean, air-tight bottles. There

should be no head space in samples used for volatile compounds or dissolved gases. The type of analysis for which a sample is being collected determines the type of bottle, preservative, holding time, and filtering requirements. Preservatives can retard biological action, inhibit chemical reactions such as hydrolysis or oxidation, and prevent sorption. Generally, samples should be placed in a cooler maintained at 4 oC. Ideally, they should be shipped to an analytical laboratory within 24 hours of sample collection.

Sample labels should include the identification number, name of the collector, date and time collected, place of collection, and parameters to be analyzed by the laboratory. A chain of custody procedure should be designed to allow the owner/operator of a site to reconstruct how and under what circumstances a sample was collected. The purpose of this procedure is to prevent misidentification of samples, prevent tampering with the samples during shipping and storage, allow easy identification of any tampering, and permit easy tracking of possession.

Analytical results reported by the laboratory guide site characterization and remediation activities. Field technicians should develop a quality control program to ensure that the analytical results accurately express the actual concentrations of chemicals in groundwater. Laboratory accuracy can be checked by using spiked samples. A spiked sample contains a known concentration of a particular solute. It is submitted to the lab along with the other samples. The lab report is checked to verify that the proper concentration is reported for the spiked sample. Analytical precision is the ability of the laboratory to reproduce results. Precision is checked by using split samples. Split samples are obtained by mixing a sample and splitting it into two containers. Because the split samples are from the same source, they should yield identical concentrations when they are chemically analyzed.

Field collection techniques also influence the degree to which the laboratory results reflect what is actually in the groundwater. After a well is sampled, the sampling equipment should be cleaned thoroughly with deionized water before proceeding to the next well. Depending on the contaminants present, other cleaning solutions may be appropriate. Field blanks can be used to evaluate the quality of a field sampling program. Purified water is taken to the field in a sealed container. The water is run through the sampling equipment, placed in a container, and sent to the lab for analysis. It should not contain any contaminants. The presence of one or more contaminants is an indication that the sampling equipment was not properly cleaned between the collection of samples.

Several water quality parameters can be measured directly in the field. These include the electrical conductance, pH, and dissolved oxygen content of the groundwater.

Electrical Conductance

The conductivity of a solution is a measure of its ability to carry an electrical current. Electrical conductance is the conductivity of a mass of fluid of unit length and unit cross section at a specified temperature (Freeze and Cherry, 1979). Conductivity meters measure electrical conductance. Typical units for electrical conductance are millisiemens (mS) and microsiemens (μ S).

Pure liquid water has a very low electrical conductance, less than a tenth of a microsiemen at 25 oC (Hem, 1970). Charged ionic species make a solution conductive. Most substances dissolved in water dissociate into ions that can conduct an electrical current. Conductance is also affected by the type and number of ions in the solution. In general the larger the conductance, the more mineralized the water. Because a small change in temperature causes a large change in conductivity, the conductivity meter must compensate for this change at high and low temperatures. The readings are usually normalized to 25 oC. In the meter you will be using, temperature compensation is automatic from 10 to 40 oC. The temperature of shallow groundwater usually falls within that range.

Electrical conductance measurements can be used to estimate the concentration of total dissolved solids (TDS) in a water sample. However, it should be noted that natural water contains a variety of both ionic and uncharged chemical species. Therefore, conductance determinations cannot be used to obtain highly accurate estimates of TDS. To convert conductance to TDS, the following relation is used (Hem, 1970):

$$TDS = AC$$

where C is the conductance in microsiemens, TDS is expressed in milligrams per liter, and A is a conversion factor. For most groundwater, A is between 0.55 and 0.75, depending on the ionic composition of the solution. The concentration of dissolved minerals in groundwater is a general indication of its suitability for a particular use (Driscoll, 1986). Barring the presence of contaminants, water containing less than 500 milligrams per liter of dissolved solids is generally satisfactory for domestic use and many industrial purposes. If total dissolved solids exceed 1,000 milligrams per liter, the electrical conductivity of the water may cause serious electrolytic corrosion of metal well screens or pump components (Driscoll, 1986).

The conductivity meter measures in one of four ranges: 0.0 to 199.9 μ S; 0 to 1999 μ S; 0.00 to 19.99 mS; and 0.0 to 199.9 mS. The probe has four stainless steel electrodes. A PVC sleeve, with holes for air passage, protects the temperature sensor. The sleeve must not be removed during a measurement because it controls the flow of solution around the four electrodes.

The meter must be calibrated periodically. To calibrate the meter, pour part of the 12.88 mS solution into a beaker. Immerse the probe into the solution, making sure the solution covers the probe holes. Wait until thermal equilibrium has been attained. Press ON and select the 19.99 mS range. Tap the probe at the bottom end and rotate it to get rid of air bubbles trapped inside the sleeve. Turn the trimmer at the side of the instrument with a screwdriver until the display reads 12.88 mS. The calibration is now complete.

To take a measurement, press the ON/OFF key, and wait for the display to read zero. Immerse the probe into the solution to be measured, and press one of the four range keys. If the display shows an "I" on the left side with no readings, then you selected a range that is too low. Select the next higher range. The built-in temperature sensor requires about two minutes to attain thermal equilibrium. Tap and shake the probe

to remove air bubbles. After completing a measurement, switch off the meter and rinse the probe with clean tap water.

pH

The pH of water is a measure of its reactive characteristics. Formally, it is the negative logarithm of the hydrogen ion activity. It describes whether a solution is acidic ($\text{pH} < 7$), neutral ($\text{pH} = 7$) or basic ($\text{pH} > 7$). Low values of pH, particularly less than 4, indicate a corrosive water that will tend to dissolve metals and other substances. High pH values, above 8.5, indicate an alkaline water. Most groundwaters of the United States have pH values that range from about 6.0 to 8.5 (Hem, 1970).

The pH instrument, which measures from 0.00 to 14.00 pH units, can be used in temperatures between 0.0 and 70.0 °C. It has automatic temperature compensation from 0 to 50 °C.

To calibrate the instrument, remove the protective cap from the tester. Turn the tester ON. Immerse the probe in the pH 7 buffer solution. Do not submerge the probe above the maximum immersion level line. Press and hold the ON/OFF/CAL button for approximately three seconds, and then release the button. After approximately 15 seconds, the LCD will prompt you to immerse the probe into the second buffer solution. If you intend to test in the acidic range, immerse the tip of the tester into the pH 4 solution. To test in the alkaline range, immerse the tip of the tester into the pH 10 solution. The tester will automatically calibrate to the buffer value at the actual temperature of the buffer solution. Once the electrode stabilizes, the display will stop blinking, and the tester will go to its operational mode. The meter should be calibrated before each set of measurements.

To take a measurement, remove the protective cap and turn the tester ON. Place the tip of the electrode into the sample to be measured. The tester is measuring and compensating for temperature. Therefore, it may require several seconds for the display to stabilize. To display the measured temperature, press and hold the °C button.

The electrode should never be stored dry. When finished, place a few drops of pH 4 or 7 buffer solution into the protective cap of the electrode and replace the cap. Never use deionized or distilled water. If the electrode is left dry, it must be reactivated. Reactivation entails soaking the electrode up to the maximum immersion level in pH 4 or 7 buffer solution overnight.

To avoid changes caused by escape of CO₂ from the water, measurements of the pH of groundwater are normally made in the field immediately after a sample is collected (Freeze and Cherry, 1979). Carbon dioxide in groundwater normally occurs at a higher partial pressure than in the earth's atmosphere. When groundwater is exposed to the atmosphere, CO₂ will escape and the pH will rise.

Dissolved Oxygen

The oxygen meter measures the dissolved oxygen level and temperature of water and aqueous solutions. It includes an oxygen probe and thermistor, which is used to

measure temperature and compensate for changes in the oxygen sensor's response to temperature.

In a dissolved oxygen probe, oxygen diffuses through a membrane into a measuring cell at a rate proportional to the partial pressure of oxygen in the water. Inside the sensor, the oxygen reacts with an electrolyte and is reduced by an applied voltage. The current that is generated is directly proportional to the partial pressure of oxygen in the water outside the sensor (Freeze and Cherry, 1979).

To measure the dissolved oxygen concentration of a sample, connect the probe to the DIN jack on the instrument. Turn the instrument dial to the O₂ position. Allow the meter to warm-up for 20 minutes. Calibrate the instrument.

Air Calibration

The amount of oxygen dissolved in water is dependent upon several parameters. Temperature and atmospheric pressure each have an effect on the measurement of dissolved oxygen. You must correctly compensate for these parameters before taking a measurement.

Without touching the membrane, wrap the probe in a cloth soaked with deionized water. The membrane should remain covered with a thin layer of water while wrapped in the cloth. Allow approximately two minutes for the probe to stabilize. To convert a reported barometric pressure to an actual atmospheric pressure, find the appropriate multiplier in Table 6-1. Obtain the temperature of the sample with the meter in the TEMP mode. Use this temperature and the actual atmospheric pressure with Table 6-2 to determine the calibration value for dissolved oxygen in air-saturated water. Adjust the CAL control to the value determined from Table 6-2.

To measure the concentration of dissolved oxygen in a sample, place the probe in the sample and stir. Stirring ensures a continuous supply of fresh sample to prevent depletion of oxygen near the probe. Allow at least three minutes for the probe to stabilize. Read the dissolved oxygen concentration in milligrams per liter. Turn the instrument to the TEMP position and read the temperature of the sample. Rinse the probe with deionized water between sample readings and upon completion of meter use.

Exercises

1. Measure the electrical conductance, dissolved oxygen, and pH of wells 1 through 4.
2. A sample from a drinking fountain in the General Academic Building had an electrical conductance of 420 μ S. Is this value higher or lower than your measurements in problem 1? Speculate why.
3. Estimate the concentration of total dissolved solids in each well.

Table 6-1. Altitude Multiplier¹

Altitude (feet)	Multiplier
- 500	1.018
0	1.000
500	0.982
1000	0.964
1500	0.947
2000	0.930
2500	0.913
3000	0.896
3500	0.880
4000	0.864
5000	0.832
6000	0.801
7000	0.772
8000	0.743
9000	0.715
10000	0.688

¹ CRC (1989).

Table 6-2. Dissolved Oxygen Calibration Values¹

Temperature		Pressure (millimeters Hg)					
°C	°F	775	760	750	725	700	675
0	32	14.9	14.6	14.4	13.9	13.4	12.9
2	36	14.1	13.8	13.6	13.2	12.7	12.2
4	39	13.3	13.1	12.9	12.5	12.0	11.6
6	43	12.7	12.4	12.3	11.8	11.4	11.0
8	46	12.0	11.8	11.7	11.3	10.9	10.5
10	50	11.5	11.3	11.1	10.7	10.4	10.0
12	54	11.0	10.7	10.6	10.2	9.9	9.5
14	57	10.5	10.3	10.1	9.8	9.5	9.1
16	61	10.0	9.8	9.7	9.4	9.0	8.7
18	64	9.6	9.4	9.3	9.0	8.7	8.4
20	68	9.2	9.1	8.9	8.6	8.3	8.0
22	72	8.9	8.7	8.6	8.3	8.0	7.7
24	75	8.6	8.4	8.3	8.0	7.7	7.4
26	79	8.3	8.1	8.0	7.7	7.4	7.2
28	82	8.0	7.8	7.7	7.4	7.2	6.9
30	86	7.7	7.6	7.4	7.2	6.9	6.7
32	90	7.4	7.3	7.2	7.0	6.7	6.4
34	93	7.2	7.1	7.0	6.7	6.5	6.2
36	97	7.0	6.8	6.7	6.5	6.3	6.0
38	100	6.8	6.6	6.5	6.3	6.1	5.8
40	104	6.6	6.4	6.2	6.1	5.9	5.7
42	108	6.4	6.2	6.1	5.9	5.7	5.5
44	111	6.2	6.0	5.9	5.7	5.5	5.3
46	115	6.0	5.8	5.8	5.5	5.3	5.1
48	118	5.8	5.7	5.6	5.4	5.2	5.0
50	122	5.6	5.5	5.4	5.2	5.0	4.8

¹ Hitchman (1978).

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