From: Joel Geier
To: Benton Public Comment
Subject: LU-24-027: Information to add to the record regarding arsenic
Date: Saturday, May 3, 2025 6:11:57 AM
Attachments: WV Arsenic in Groundwater pb2002100743.pdf

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Dear Community Development Department staff and members of the Planning Commission:

Please enter the attached document:

> Arsenic in Ground Water of the Willamette Basin, Oregon
> by Stephen R. Hinkle and Danial J. Polette
> Water-Resources Investigations Report 98–4205

into the record for LU-24-027.

This is a publicly available document produced by the United States Geological Survey.

It contains scientific information that is highly relevant to the Applicant's claims that the high levels of arsenic discovered in groundwater at the landfill site are "typical" for this part of the Willamette Valley.

I've cited this document in my own previously submitted testimony on the arsenic issue (in which, I notice now, I misspelled the name of the second author as "Pollette").

This document is technical in nature, so some details may require a scientific background to understand. But here's one simple thing to note:

If you run text search on this document, you'll find that the names of the two counties closest to Coffin Butte Landfill, Benton and Polk, occur just four times in total. Two of those instances are as labels on maps. The other two are for a single document in the list of references. Nowhere in this document is there any mention of arsenic as being a naturally occurring problem in either Benton or Polk County.

In contrast, Linn and Lane counties are mentioned frequently (a total of about 75 time). Linn and Lane are the counties in the southern Willamette Valley where arsenic in groundwater is recognized to be a common problem, not Benton or Polk.

In other words, the "character of the area," in areas of Benton and Polk counties all around the landfill, has been an area where rural residents haven't had to worry about arsenic in their wells, at least up until now.

Thank you for taking this information into consideration for your deliberations.

Yours sincerely,
Joel Geier, Ph.D.
38566 Hwy 99W
Corvallis, OR 97330-9320

U.S. Department of the Interior
U.S. Geological Survey

Arsenic in Ground Water of the Willamette Basin, Oregon

Water-Resources Investigations Report 98–4205

Prepared in cooperation with
OREGON WATER RESOURCES DEPARTMENT

U.S. Department of the Interior
U.S. Geological Survey

Arsenic in Ground Water of the Willamette Basin, Oregon

BY STEPHEN R. HINKLE AND DANIAL J. POLETTE

Water-Resources Investigations Report 98–4205

Prepared in cooperation with
OREGON WATER RESOURCES DEPARTMENT

Portland, Oregon: 1999

U.S. DEPARTMENT OF THE INTERIOR
BRUCE BABBITT, Secretary

U.S. GEOLOGICAL SURVEY
Charles G. Groat, Director

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<http://wwworegon.wr.usgs.gov/projs_dir/willgw/willpage.html>

Information regarding Oregon District activities is available at:
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Acknowledgments

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Arsenic in Ground Water of the Willamette Basin, Oregon

By Stephen R. Hinkle *and* Danial J. Polette

SUMMARY AND CONCLUSIONS

Arsenic concentrations exceeding the U.S. Environmental Protection Agency (USEPA) current Maximum Contaminant Level (MCL) of 50 μg/L (micrograms per liter) are widespread in ground water in the Willamette Basin. The Oregon Water Resources Department and the U.S. Geological Survey began a cooperative study in the Willamette Basin in 1996. One goal of this study is to characterize the regional distribution of naturally occurring poor-quality ground water, such as ground water with high concentrations of arsenic. Characterization of the regional distribution of arsenic concentrations in the Willamette Basin will be useful to public health officials, water-resource managers, the medical community, and those using ground water for drinking and cooking.

The spatial distribution of arsenic concentrations in ground water of the Willamette Basin was assessed by combining historical data from 597 sites with data from 131 sites collected for this study. A total of 728 spatially distinct samples thus were available. Additional data also were collected to evaluate temporal variability of arsenic concentrations on a seasonal timescale. Samples were collected quarterly from 17 sites for 1 year for this purpose. Temporal variability was addressed for two reasons: First, characterization of temporal variability allowed evaluation of the acceptability of combining arsenic-concentration data collected during different seasons for determining the spatial distribution of arsenic concentrations. Second, knowledge of temporal variability will benefit well owners and water managers who require guidance on timing for sampling.

During the course of quarterly sampling, arsenic concentrations in water from many wells remained essentially constant, but variations of up to almost a factor of three were observed in other wells. No obvious correlation with season was apparent. Analytical accuracy, as determined from 11 standard reference samples submitted during the course of project work, generally was within ±10 percent, and always ±20 percent. Thus, analytical variability can only explain some of the observed temporal variability. One possible explanation for observed temporal variability in arsenic concentrations is that differences in the amount of pumpage prior to sampling may lead to variations in the amounts of water pumped from different sources (different aquifers or parts of aquifers), and thus, differences in water chemistry.

For a regional assessment of arsenic concentrations in ground water, where arsenic concentrations may vary in space by several orders of magnitude, the relatively smaller temporal variations such as those observed in the quarterly samples are not a significant limitation, and the aggregation of data collected at different times is justified. However, this conclusion may not necessarily apply to all investigations of arsenic concentrations in ground water. For some purposes, site-specific characterization may require characterization of temporal variability. Such characterization may require evaluation over a range of well uses and seasons.

Concentrations of arsenic in the 728 spatially distributed samples ranged from less than 1 to 2,000 μg/L. Concentrations in 58 (8.0 percent) of the samples exceeded the USEPA current MCL.

Regionally, the distribution of arsenic concentrations in ground water of the Willamette Basin appears to be primarily related to aquifer geology. High arsenic concentrations (concentrations exceeding the USEPA current MCL) are widespread in bedrock areas in south-central and eastern Lane County, and Linn County. High concentrations of arsenic also are present in some ground water in the Tualatin Basin (a subbasin in the northwestern part of the Willamette Basin). High arsenic concentrations in Lane and Linn Counties appear to be associated with two regionally extensive associations of rocks, (1) the Fisher and Eugene Formations and correlative rocks, and (2) the undifferentiated tuffaceous sedimentary rocks, tuffs, and basalt. (These rock associations are defined by Walker and MacLeod, 1991. The undifferentiated tuffaceous sedimentary rocks, tuffs, and basalt are approximately equivalent to the Little Butte Volcanic Series of Peck and others, 1964.) At land surface, these two rock associations cover 24 percent of the Willamette Basin. These associations of rocks include extensive volumes of silicic (rhyolitic) volcanic rocks, which are commonly associated with high concentrations of arsenic. High concentrations in the Tualatin Basin are associated with alluvial deposits. At a regional scale, well depth does not appear to be a useful predictor of arsenic concentration in the Willamette Basin. However, depth may be an important parameter on a local scale, particularly where wells of different depth tap aquifers in different geologic units.

Ground waters in bedrock areas in south-central and eastern Lane County, bedrock areas in Linn County, and alluvial areas in the Tualatin Basin may be more likely to yield water high in arsenic than ground water elsewhere in the basin. However, it cannot be assumed that these areas are the only areas in the basin that contain ground water with high concentrations of arsenic. Little or no data exist for many parts of the basin. Even in areas that have been sampled, geohydrologic heterogeneity makes it difficult to formulate meaningful generalizations regarding the likelihood of finding high-arsenic ground water. There is no substitute for actual sampling.

Available information, in combination with an understanding of processes known to promote arsenic mobilization, is sufficient to formulate hypotheses that explain arsenic sources and mobilization in the Willamette Basin. However, available geochemical data and interpretations are sparse. Thus, these hypotheses are preliminary, serving mainly to help direct future geochemical investigation in the Willamette Basin.

Anthropogenic sources of arsenic can be significant in some settings. Arsenical pesticides such as lead arsenate have been used in the basin, and arsenic can be released into the environment from industrial sources. However, regional patterns of arsenic occurrence in Willamette Basin ground water are not consistent with either industrial or agricultural sources of arsenic.

Naturally occurring arsenic commonly is found in a variety of solid phases. Arsenic can be a component of volcanic glass in volcanic rocks of rhyolitic to intermediate composition, adsorbed to and coprecipitated with metal oxides (especially iron oxides), adsorbed to clay-mineral surfaces, and associated with sulfide minerals and organic carbon. Examination of these potential arsenic sources for arsenic availability in the Willamette Basin apparently has never been done.

Two categories of processes largely control arsenic mobility in aquifers: (1) adsorption and desorption reactions and (2) solid-phase precipitation and dissolution reactions. Arsenic adsorption and desorption reactions are influenced by changes in pH, occurrence of redox (reduction/oxidation) reactions, presence of competing anions, and solid-phase structural changes at the atomic level. Solid-phase precipitation and dissolution reactions are controlled by solution chemistry, including pH, redox state, and chemical composition.

Several species of arsenic occur in nature, but arsenate (arsenic V) and arsenite (arsenic III) are the two forms commonly found in ground water. For this study, samples from five domestic wells were analyzed for arsenic species. Two additional analyses for arsenic species in ground water from the Willamette Basin were available in the literature. Arsenite was the predominant species of arsenic in six of these seven samples. The predominance of arsenite has both geochemical and toxicological implications. From a geochemical standpoint, mobility of arsenite differs from that of arsenate. From a public-health perspective, arsenite is more toxic than arsenate, and arsenite also is more difficult to remove from drinking-water supplies than is arsenate. Seven samples do not characterize regional arsenic speciation patterns. However, if the predominance of arsenite in Willamette Basin samples is substantiated by additional speciation work, public health officials and water managers may need to evaluate the scope of the arsenic problem with regard not only to arsenic concentrations, but also to arsenic speciation.

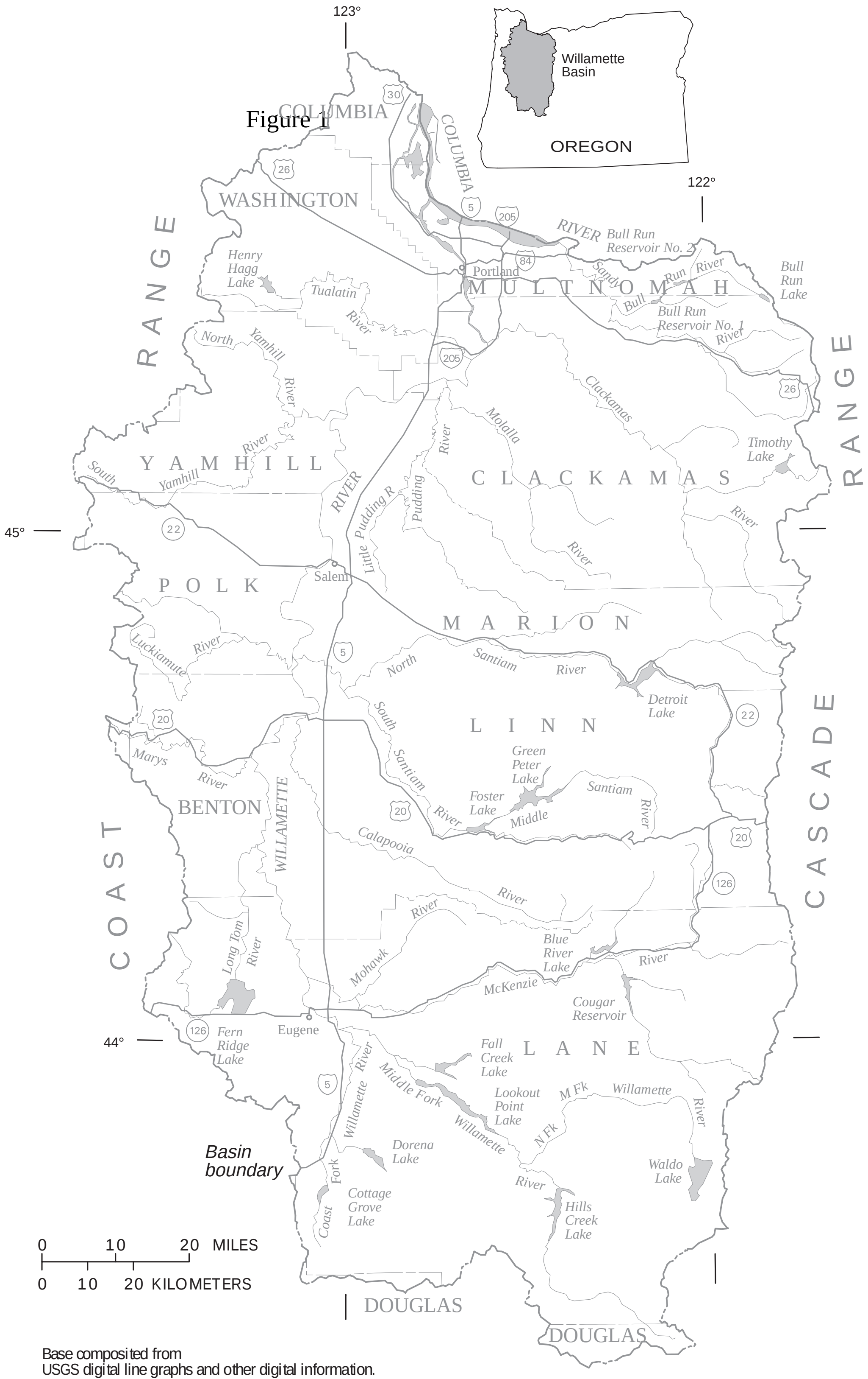
Existing data, including the speciation data, and published interpretations were used to establish preliminary hypotheses for the evolution of high-arsenic ground water in the Willamette Basin. For ground water in bedrock areas of Lane and Linn Counties, existing information suggests that at least some of the following controlling factors likely are important in adsorption and desorption reactions that often control arsenic mobility: (1) high pH, (2) presence of competing anions, and (3) occurrence of reducing conditions. Existing information did not allow for evaluation of the potential importance of adsorption and desorption reactions related to solid-phase structural changes at the atomic level, or solid-phase precipitation and dissolution reactions.

For alluvial ground water of the Tualatin Basin, presence of competing anions and occurrence of reducing conditions may be important controlling factors in arsenic adsorption and desorption reactions. These two factors might be more important than pH controls over arsenic adsorption and desorption. Reducing conditions and high concentrations of dissolved iron also suggest that dissolution of iron oxides, with subsequent release of adsorbed and (or) coprecipitated arsenic, may play a role in arsenic mobility in the Tualatin Basin.

Although the regional distribution of arsenic concentrations in ground water of the Willamette Basin has been evaluated by this study, an understanding of how ground water in parts of the basin evolved to contain high concentrations of arsenic has not yet been developed. Limited geochemical data have allowed establishment of preliminary hypotheses to explain the evolution of high-arsenic ground water. Developing an understanding of arsenic sources and processes responsible for evolution of high concentrations of arsenic, though, will require additional geochemical investigation. In particular, thermodynamic evaluation of ground water chemistry and study of solid phases present in aquifers would facilitate development of an understanding of adsorption and desorption and precipitation and dissolution reactions controlling arsenic mobility in the Willamette Basin. A key benefit of detailed geochemical study of arsenic in ground water of the Willamette Basin would be increased predictability of areas likely to yield ground water with high arsenic concentrations. Such increased predictability would be likely to have transfer value beyond the Willamette Basin.

INTRODUCTION

In response to increasing demands on ground-water resources in the Willamette Basin, Oregon (fig. 1), the Oregon Water Resources Department (OWRD) and the U.S. Geological Survey (USGS) began a cooperative study of the basin's ground-water resources in 1996. This study was designed to increase the current understanding of the ground-water resource, and to better characterize the distribution of naturally occurring poor-



Base composited from
USGS digital line graphs and other digital information.
Universal Transverse Mercator projection, Zone 10
1927 North American Datum

Figure 1. Location of the Willamette Basin, Oregon.

quality ground water in the basin. Essential components of the study of the physical ground-water resource are the development of a quantitative understanding of regional ground-water availability and flow, and of ground-water/surface-water interactions. Of paramount interest in the characterization of naturally occurring poor-quality ground water in the Willamette Basin is the distribution of arsenic in ground water, the subject of this report.

Arsenic contaminates many regional aquifer systems worldwide (Cantor, 1996; Thornton, 1996), and arsenic commonly is detected in ground water of the Willamette Basin at concentrations exceeding the U.S. Environmental Protection Agency (USEPA) current drinking water Maximum Contaminant Level (MCL) of 50 µg/L (micrograms per liter) (U.S. Environmental Protection Agency, 1996). Arsenic is associated with a number of adverse effects on human health. The USEPA considers arsenic to be a human carcinogen (U.S. Environmental Protection Agency, 1996). Examples of other adverse health effects attributed to consumption of arsenic range from weakness and abdominal pain to neurological and cardiovascular problems. A review of health effects associated with consumption of arsenic is given in a report by World Health Organization (1996).

Purpose and Scope

The primary purpose of this report is to describe the spatial distribution of arsenic concentrations in ground water of the Willamette Basin. Both historical data and data collected for this study (henceforth, “project data”) were used for this purpose. Project data also were used to evaluate temporal variability of arsenic concentrations. It is useful to have an understanding of temporal variability before arsenic-concentration data, collected at different times, is used to evaluate spatial distributions. Evaluation of temporal variability may also benefit well owners and water managers, who may require guidance on timing for sampling.

A secondary purpose of this report is to briefly summarize current knowledge of the geochemistry of arsenic in the Willamette Basin. Relevant geochemical data are few, so this discussion is inherently general, serving mainly to outline future research needs. Possible sources of arsenic are evaluated, and geochemical processes that may control arsenic mobilization are briefly discussed.

Location and Description of the Willamette Basin

The Willamette Basin is an approximately 12,000-square-mile basin in northwestern Oregon. Primary drainage is by the Willamette River, but for the purposes of the study, the basin is defined to also include the region drained by the Sandy River; both rivers are tributary to the Columbia River. The Willamette Basin was home to 69 percent of the State’s population in 1990 (Broad and Collins, 1996).

The crests of two north-south trending mountain ranges, the Coast and the Cascade Ranges, respectively define the western and eastern edges of the Willamette Basin. The Willamette Valley, an elongated, structural and erosional lowland, lies between these mountain ranges. The Coast Range is composed of marine sedimentary rocks and associated volcanic rocks. The Cascade Range is composed of lava flows and pyroclastic and epiclastic rocks. The Willamette Valley is filled with clastic basin-fill sediments of primarily alluvial origin; these alluvial sediments form the most important aquifers in the Willamette Basin. The geologic framework of the basin is described by Gannett and Caldwell (in press), and a regional representation of the surficial geology of the Willamette Basin is given on the geologic map of Oregon compiled by Walker and MacLeod (1991). Usage of geologic names in this report is consistent with that of Walker and MacLeod (1991).

STUDY DESIGN AND METHODS

The overall approach used to collect, assemble, and analyze data for this report is described in this section. First, a description of the sources of historical data is given, followed by a description of the sampling design for project data. Approaches used to define the quality of both historical and project data are discussed, as are collection and analytical methods used for project data. Finally, benchmarks for comparison of arsenic-concentration data, and methods for identification of well locations, are described.

Historical Data

Analysis of historical data (arsenic concentrations and site locations, and in most cases, well depths) from regional ground-water investigations was the starting point for evaluation of distribution of arsenic in the Willamette Basin. Some wells were sampled more than once; in these cases, the first-in-time sample was selected. Four sets of historical data used in this report are described below.

Historical data from the USGS National Water Information System (NWIS) database (Maddy and others, 1990) (271 wells). These data were collected between 1971 and 1997 as parts of various USGS projects. Many of these projects were regional in scope, and thus these data cover large areas in the Willamette Basin. In addition to arsenic concentrations, depth data also were retrieved. Data from both unfiltered and filtered samples were found in NWIS. Some of these NWIS data are discussed in the following reports: Frank and Collins, 1978; Gonthier, 1983; Leonard and Collins, 1983; Hinkle, 1997. Project data, although stored in NWIS, are discussed separately (see section “Sampling Design for Project Data”).

Data from four USGS studies (Frank, 1973, 1974, 1976; Helm and Leonard, 1977), not entered into NWIS (89 wells). These data, also from regional-scale projects, encompass large areas in the southern part of the basin. The data were collected between 1964 and 1973 by USGS personnel. Well depths were obtained from tables in the reports. Techniques used to process these samples (in particular, filtering or a lack thereof) are not known.

Data from a USGS study in Lane County, not entered into NWIS (171 wells; 1 nonthermal spring). These data were collected during 1962–63, and summarized by Goldblatt and others (1963). Many of these wells withdraw water from the arsenic-rich Fisher Formation (Goldblatt and others, 1963). Arsenic concentrations and well depths were compiled from original project notes from USGS files. Samples were collected and analyzed as unfiltered samples (A.S. Van Denburgh, U.S. Geological Survey, oral commun., 1997).

Data from Linn County Department of Health Services, Environmental Health Program (65 wells). More than 100 wells were sampled for arsenic by the Environmental Health Program in 1987. Most of these wells were located near Sweet Home, an area that received little coverage in the three regional data sets listed above. Drillers’ logs were on file with the Environmental Health Program for many of these wells. In 1996, USGS personnel were able to locate 65 homes corresponding to addresses from well drillers’ logs for sampled wells. Arsenic concentrations from Environmental Health Program files and well depths from well drillers’ logs were matched with the home locations to create a data coverage. Samples were probably collected and analyzed as unfiltered samples.

Sampling Design for Project Data

Project data were collected for several purposes. Ground-water samples were collected to fill gaps in the spatial distribution of the historical data and to illustrate the magnitude of temporal variability in arsenic concentrations. In addition, five ground-water samples were analyzed for arsenic species. Quality-control (QC) data were collected to evaluate the quality of project data, evaluate the quality of historical data (by resampling selected historical sites), and compare results obtained by different processing and analytical methods.

Samples from 125 wells and 6 nonthermal springs were collected during 1996 and 1997 to complement the spatial distribution of historical data. The wells and springs sampled were distributed throughout the lower elevation areas of the Willamette Basin, which are areas of greatest ground-water use. These sites had not previously been sampled for arsenic by the USGS. Some wells were sampled more than once during the course of this project; in these cases, the first-in-time samples were used to define the spatial distribution of arsenic.

To evaluate temporal variability of arsenic concentrations, samples were collected quarterly for 1 year from each of 17 sites. These samples also were collected during 1996 and 1997. A subset (5) of these 17 sites were sampled for arsenic species.

QC samples were used to evaluate the quality of techniques used to collect and analyze project samples. Twelve field equipment blanks, 6 sets of triplicate split samples, and 11 standard reference samples (SRSs) were analyzed over the course of the project. Field equipment blanks allow evaluation of the extent of any sample contamination resulting from sample collection, processing, and analysis. Replicate samples allow evaluation of the reproducibility (precision) of analyses. SRSs facilitate evaluation of analytical accuracy. The USGS SRS program, an interlaboratory testing program, is described in a report by Long and Farrar (1995). Preparation, description, and most probable values (MPVs) of constituents of individual SRSs used in this study are described by U.S. Geological Survey (1990) and Long and Farrar (1991, 1993, 1995). An MPV for an analyte is the median of the concentrations determined by the participating laboratories. Analytical results are reported as percentages of SRS-program MPVs.

Additional quality assurance included resampling 11 historical sites. Comparison between historical arsenic concentrations and concentrations determined upon resampling offers a measure of the reliability of historical data.

Comparisons between filtered and unfiltered samples, and between USGS and USEPA analytical methods, also were made. Characterization of differences in arsenic concentrations among unfiltered and filtered samples helps quantify the effect of sample filtering, and increases the transfer value of the data and interpretations presented in this report. Data from comparison of USGS and USEPA analytical methods facilitate comparison of arsenic concentrations determined by USGS methods with USEPA water-quality criteria.

Project Sample Collection Methods

Project samples from wells and springs used for evaluation of spatial distribution of arsenic were not filtered. Unfiltered samples, in addition to being more economical to collect than samples filtered through 0.45-μm (micrometer) filters, also have the advantage of being more representative of the water being consumed by most well owners. Another justification for collection of unfiltered samples is that many of the historical data, with which project data were combined, were from analyses of unfiltered samples. Furthermore, USEPA and World Health Organization (WHO) guidelines for water quality (see section “Comparisons with Water-Quality Criteria“) apply to “finished water.” For most project wells, which were primarily domestic wells, “finished water” generally is equivalent to unfiltered water. However, because unfiltered samples may contain more colloids and (or) more sediment entrained during pumping, unfiltered samples are less representative of water actually moving through an aquifer than are 0.45-μm-filtered samples. Thus, the use of unfiltered samples in characterization of ground-water quality represents an approximation. Additional, quantitative discussion on this topic is presented in the section, “Comparison of Processing and Analytical Methods.”

Of the 125 project wells used for evaluation of spatial distribution of arsenic, 116 were actively used domestic wells. The remaining 9 wells included 3 public-water-supply wells, 3 industrial wells, 2 irrigation wells, and 1 livestock well. Of these nine wells, those that were not actively used were purged a minimum of three casing volumes prior to sampling to remove standing water from the well. Samples from actively used wells were collected following a minimum purge time of 1 minute. Longer purge times, characteristic of most USGS ground-water-quality work, were deemed unnecessary for actively used wells in this project because these wells experienced a degree of regular purging from the frequent use of the wells. A resulting limitation, however, is that samples from these wells may lose arsenic by way of adsorption to iron casing or precipitation as ground water undergoes geochemical changes while residing in a well bore or casing. Thus, arsenic concentrations in samples from these wells may be biased toward low arsenic concentrations relative to water actually moving through the aquifer. The extent of this possible bias has not been quantified, but because these wells were actively used, this potential bias is likely to be small.

Project samples collected from springs were collected from flowing springs. Fine sediment was present along with the water in several of the springs, so spring samples were filtered through 0.45-μm nominal-pore-size filters.

Project wells sampled for evaluation of temporal variability in arsenic concentrations were actively used wells, sampled using the same methods as for project wells sampled for evaluation of spatial distribution of arsenic.

Project wells sampled for arsenic speciation also were actively used wells. Samples were collected as unfiltered samples following a minimum well purge time of 1 minute. Samples were collected without headspace in brown glass vials, wrapped in aluminum foil (to prevent photooxidation), and shipped on ice to the laboratory.

Eleven wells represented in the historical data set were resampled. All were actively used wells and were sampled using the same methods as for project wells sampled for evaluation of spatial distribution of arsenic.

Samples for comparison between filtered and unfiltered samples, and comparison of USGS and USEPA methods, were collected from a subset of the wells sampled for temporal variability. Each 10-liter sample was split in the field by mechanical agitation into four subsamples. One subsample was filtered through a 0.10-µm nominal-pore-size 47-mm-diameter filter. One subsample was filtered through a 0.45-µm nominal-pore-size 142-mm-diameter filter. Two subsamples were collected as unfiltered samples. For each set of the four subsamples, both of the filtered samples and one of the unfiltered samples were analyzed by USGS methods (see section “Project Analytical Methods”). The other unfiltered sample was analyzed by USEPA methods (see section “Project Analytical Methods”).

All arsenic samples, except samples collected for analysis of arsenic species, were field-acidified to below pH 2 with nitric acid. Samples for analysis of arsenic species were not acidified.

Project Analytical Methods

Arsenic analyses were done at the USGS National Water Quality Laboratory (NWQL) in Arvada, Colorado. Most arsenic determinations were done by hydride atomic absorption with a 3-minute sulfuric acid and potassium persulfate digestion (Fishman and Friedman, 1989). This is the standard USGS method, and it is referred to as the “USGS method” in this report. Ten analyses were done by graphite furnace atomic absorption with a 2-hour hydrochloric acid and nitric acid digestion (U.S. Environmental Protection Agency, 1994). This method is referred to as the “USEPA method” in this report. The analytical minimum reporting level (MRL) was 1 µg/L for both methods.

Samples for arsenic speciation were analyzed by the USGS Methods Research and Development Program at the USGS NWQL. Samples were analyzed for two inorganic species, arsenite (arsenic III) and arsenate (arsenic V), and two organic species, monosodium methylarsonate (CH_3AsO_3HNa) and sodium dimethylarsinate ($(CH_3)_2AsO_2Na$). Analyses were done by direct injection high-performance liquid chromatography/hydride generation/inductively coupled plasma-mass spectrometry. Method detection limits (MDLs) were 0.2 µg/L (expressed as mass of arsenic per liter). Samples were analyzed within 48 hours of collection.

Comparisons with Water-Quality Criteria

To provide benchmarks against which arsenic-concentration data can be compared, arsenic-concentration data are compared with USEPA and WHO drinking-water standards. Concentrations of arsenic are compared to the USEPA current drinking water MCL for arsenic of 50 µg/L (U.S. Environmental Protection Agency, 1996). The USEPA current MCL is the maximum concentration of a contaminant allowed in a public water system. This MCL is under review (U.S. Environmental Protection Agency, 1996). Bagla and Kaiser (1996) report that the USEPA is considering reducing the current MCL by 90 percent. However, until any such reduction in the MCL occurs, the current MCL remains a logical benchmark for comparison. As an alternative benchmark, concentrations of arsenic also are compared to the WHO provisional guideline of 10 µg/L (World Health Organization, 1996).

Water with an arsenic concentration below the USEPA current MCL or WHO provisional guideline is not necessarily free from health risks. For example, arsenic concentrations may be below the USEPA current MCL and WHO provisional guideline, but still be greater than the USEPA drinking-water Risk-Specific-Dose Health Advisory (RSDHA) of 2 µg/L (U.S. Environmental Protection Agency, 1996). (The RSDHA is defined as the concentration of a contaminant in drinking water that is expected to result in a specified increased risk of cancer. The USEPA RSDHA for arsenic is calculated at the 1-in-10,000 cancer risk level. Consumption of water containing a contaminant at the RSDHA 1-in-10,000 risk level is expected to be associated with the following risk: a 70-kg adult drinking 2 L of such water per day for 70 years faces an increased risk of cancer of approximately 1 in 10,000.) Furthermore, effects of arsenic consumption on human health are not uniform among different people, and no single threshold can be defined as the dividing line between “safe” and “unsafe.” Comparison of arsenic data to the USEPA current MCL and WHO provisional guideline are done solely for illustrative purposes; no implication of “safety” or lack thereof is implied.

Methods of Identifying Wells

All wells discussed in this report were assigned well location names corresponding to well locations. Well locations generally were determined when the wells were first visited. Well locations were identified using the Township, Range, and Section method of land subdivision. Two methods are shown on figure 2. Most wells were identified with a system that uses nested groups of the letters A, B, C, and D for section subdivision. Prior to about 1967, wells were identified with an alternative system, using letters A through R (excluding I and O) for section subdivision. To preserve linkage to historical data sources, all wells discussed in this report are referred to by the well location names originally assigned to them. It should be noted, however, that in some cases, the original well location names do not accurately describe the true locations of the wells. To provide accurate locational and identifying information for wells discussed in this report, corrected well locations, and additional identifying information (USGS site identification number and OWRD well log identification number), are listed along with original well locations in the

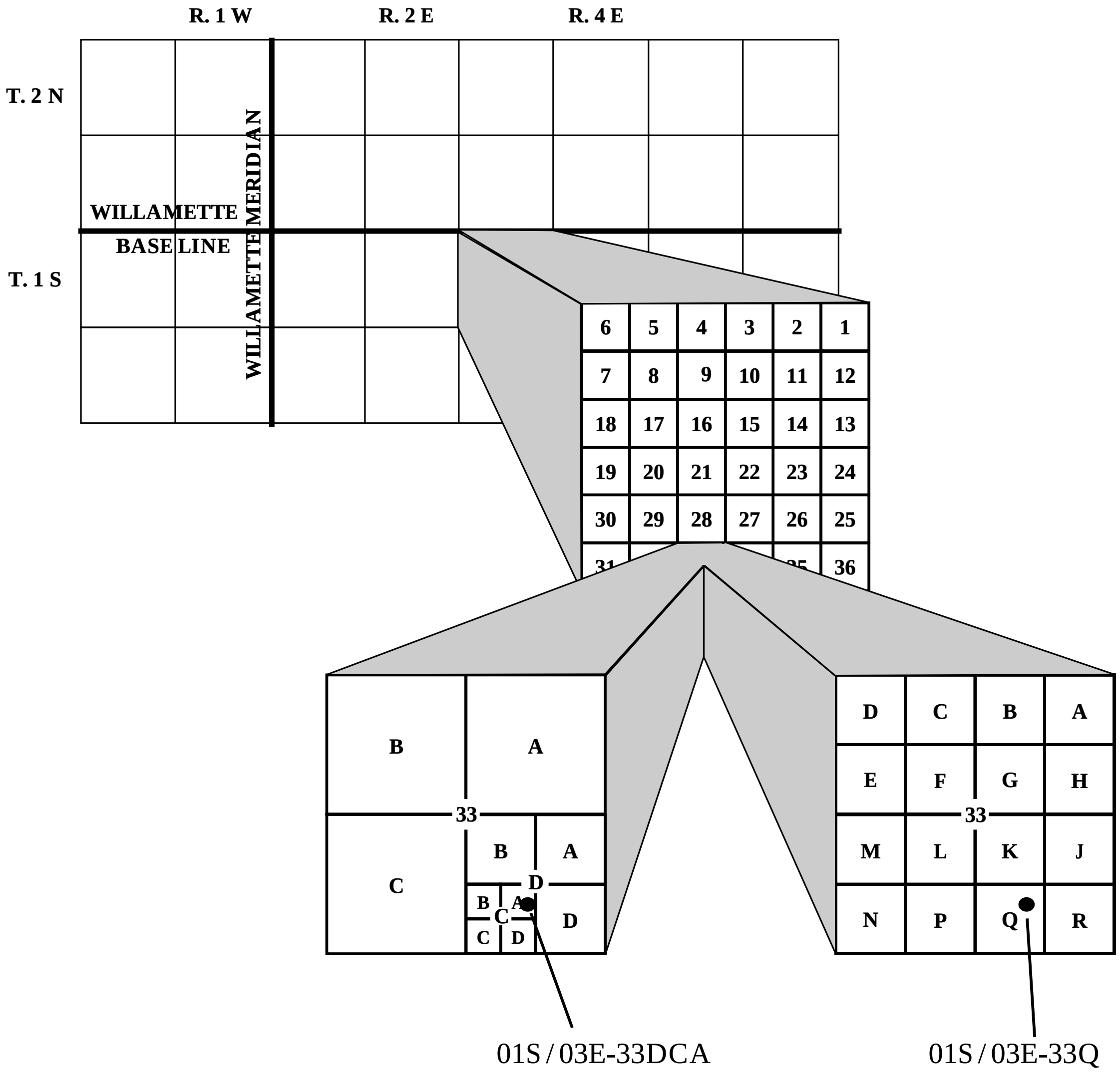


Figure 2. Well-location system.

Appendix. Note that in the project data report (Orzol and others, in press), wells are listed by corrected well location names.

DATA QUALITY: SAMPLING AND ANALYTICAL VARIABILITY VERSUS ENVIRONMENTAL VARIABILITY

In this section, project QC data are evaluated to characterize sampling and analytical variability. Sampling and analytical variability must be evaluated before environmental variability can be addressed. Three sets of QC data were collected as part of project data-collection activities. First, sampling and analytical variability of project data are evaluated. Second, the quality of historical data is discussed. Evaluation of historical data is difficult because few historical quality-control data are available. However, resampling of historically sampled sites provides insight into the quality of the historical data. Third, a comparison of sample processing and analytical methods is presented; these data illustrate the magnitude of the differences that can arise from use of various sample processing and analytical methods.

Quality of Project Data

All 12 project field equipment blanks yielded arsenic concentrations below the MRL of 1 µg/L. These results indicate that field and laboratory methods were noncontaminating.

The coefficient of variation (CV) (standard deviation divided by mean, expressed in percent) for each of the six sets of project triplicate split samples ranged from 0.0 to 14 percent. The median CV was 6.0 percent.

Analytical accuracy of project data was quantified with data from 11 SRSs. Analytical accuracy ranged from 92 to 110 percent for nine of the SRSs, but was 82 and 120 percent for the other two SRSs. In other words, reported concentrations were in error by up to about ±20 percent.

Contamination-free sampling and analysis, and reasonable analytical precision and accuracy, indicate that project data were adequate for definition of patterns of regional arsenic occurrence. However, because analytical accuracy was observed to range up to about ±20 percent, definitive characterization of temporal variability at individual sites is compromised where temporal variability also is on the order of ±20 percent or less.

Quality of Historical Data

The quality of project data is well characterized, so comparison of project and historical arsenic concentrations yields a measure of the quality of the historical data. Evaluation of the analytical accuracy of historical data is particularly desirable. However, arsenic concentrations determined in original studies and determined again during this study may differ for a variety of reasons unrelated to differences in data quality. Notably, differences between historical arsenic concentrations and arsenic concentrations determined from sampling during this project may reflect changes in the source of water being sampled at different times. Changes in the source of water being sampled can arise for a number of reasons. Ground-water flowpaths in aquifers can change over seasonal or longer time scales. Also, water often flows into wells from more than one permeable zone, and the relative contributions from different zones can change as pumping stresses change. Thus, changes in type of well use (for example, change from domestic use to lawn-watering use) or differences in the history of well use prior to sampling can result in changes in the source of water being withdrawn from wells. Finally, changes in well construction or well characteristics (for example, well cave-in over time) can result in changes in source water for wells. (Note, however, that none of the 11 wells were known to have been deepened between the time of historical sampling and the time of project resampling.) In addition to changes in source water to wells, variability in arsenic concentrations can arise from differences in sample processing prior to analysis, or, especially in the case of unfiltered samples, differences in the amount of colloid- or sediment-bound arsenic. Therefore, an absence of strong correlation between historical and project arsenic concentrations is not necessarily cause for rejection of the historical data. Because differences between historical and project data can arise from a number of factors in addition to differences in data quality, the central purpose for which the resampling data were collected was to determine if the magnitudes of the historical data are adequate for definition of patterns of regional arsenic occurrence.

Data from the 11 sites with historical data that were resampled are presented in table 1. Differences between historical and project data are variable. For example, a difference of less than 10 percent was observed for well 19S/03W-31E1, whereas an order-of-magnitude difference was observed for well 22S/03W-17N. However, historical and project arsenic concentrations were in agreement when interpreted relative to exceedances of the USEPA current MCL. Sites at which historical arsenic concentrations exceeded the USEPA current MCL also yielded water exceeding the USEPA current MCL upon sampling during this project, and sites at which historical arsenic concentrations were less than the USEPA current MCL also yielded water below the USEPA current MCL upon sampling during this project. The historical data therefore indicate a similar pattern of spatial variability of arsenic concentrations as the project data.

Closer examination of these data indicates that data from the early 1960s generally correlate poorly with project data, whereas later data demonstrate reasonably good correlation. This pattern may reflect improvements in analytical techniques since the early 1960s.

Because two historical samples dating from the mid- to late-1960s (wells 12S/01W-29N1 and 18S/04W-14ACB) had both arsenic and chloride data, these sites were sampled for chloride as well as arsenic during project sampling (table 1). For well 12S/01W-29N1, both the arsenic and chloride concentrations were slightly lower upon project sampling: the arsenic concentration upon project sampling was 86 percent of the historical concentration, and the chloride concentration, 88 percent of the historical concentration. For well 18S/04W-14ACB, both the arsenic and the chloride concentrations were considerably lower upon project sampling: the arsenic concentration upon project sampling was 60 percent of the historical concentration, and the chloride concentration, 33 percent of the historical concentration. Historical chloride concentrations would be expected to be reliable, and would have been negligibly affected by sample processing or the presence of colloids and sediment. Thus, the changes in chloride concentrations suggest that changes in the source of water being pumped by these two wells have occurred over time. If historical chloride concentrations had been similar to project chloride concentrations, then the historical arsenic analyses might be suspect. However, differences in chloride concentrations between historical and project sampling suggest that differences in arsenic concentrations between historical and project sampling were a result, at least in part, of changes in the source of water being pumped by these wells.

Comparison of historical arsenic concentrations and arsenic concentrations determined upon project sampling indicate that historical arsenic concentrations will not necessarily reflect current arsenic concentrations. Use of historical data in process-oriented geochemical studies could be problematic. However, the comparison does

Table 1. Comparison of historical arsenic concentrations with arsenic concentrations measured during this project
[Well location as recorded in original data source; arsenic concentrations in micrograms per liter; “--”, unknown]

		Historical data		Project resampling	
Source for historical data	Well location	Date	Arsenic concentration	Date	Arsenic concentration
USGS files; Goldblatt and others, 1963	18S/04W-22B	10/04/62	160	08/20/97	820
USGS files; Goldblatt and others, 1963	18S/04W-10D	10/17/62	120	08/20/97	520
USGS files; Goldblatt and others, 1963	22S/03W-17N	10/25/62	32	09/05/96	3
USGS files; Goldblatt and others, 1963	19S/03W-11E2	01/08/63	420	11/13/96	700
USGS files; Goldblatt and others, 1963	19S/03W-31E1	03/29/63	120	11/13/96	130
Frank, F.J., 1973	18S/04W-14ACB	06/12/69	500[a]	09/06/96	300[b]
Helm and Leonard, 1977	12S/01W-29N1	06/24/65	70[c]	08/29/96	60[d]
Linn County Dept. of Health Services	13S/01E-33	--/--/87	10	09/06/96	4
Linn County Dept. of Health Services	14S/01E-05	--/--/87	74	09/06/96	89
Linn County Dept. of Health Services	13S/01E-33AC	04/09/87	900	09/06/96	790
Linn County Dept. of Health Services	13S/01E-35	07/06/87	<5	09/06/96	3

[a] Chloride concentration 43 milligrams per liter.
[b] Chloride concentration 14 milligrams per liter.
[c] Chloride concentration 26 milligrams per liter.
[d] Chloride concentration 23 milligrams per liter.

suggest that the historical data are adequate for definition of patterns of regional arsenic occurrence.

Comparison of Processing and Analytical Methods

Hydrologists employ a variety of sample processing and analytical methods in geochemical and water-quality studies. Samples may be collected as unfiltered or as filtered samples. Filtering may be done using any of a variety of pore sizes, but 0.10-µm and 0.45-µm pore sizes are most commonly used. Analysis of arsenic usually is done using either hydride atomic absorption (commonly used by the USGS) or by graphite furnace atomic absorption (commonly used by the USEPA).

When combining data collected by a number of investigators using a variety of sample processing and analytical methods, questions about the comparability of data arise. Furthermore, comparison of analyses performed using non-USEPA analytical methods against USEPA water-quality criteria raises questions about comparability of analytical techniques. Information on comparability of different sample processing and analytical methods is given in this section. Data from split samples that were (1) filtered through 0.10-µm nominal-pore-size filters and analyzed by hydride atomic absorption, (2) filtered through 0.45-µm nominal-pore-size filters and analyzed by hydride atomic absorption, (3) analyzed as unfiltered samples using hydride atomic absorption, and (4) analyzed as unfiltered samples using graphite furnace atomic absorption are shown in table 2 and on figure 3.

Differences in reported arsenic concentrations between unfiltered and filtered samples generally were small. However, one set of samples (from well 21S/03E-08CBD2) demonstrated that concentrations of arsenic in unfiltered samples can be considerably greater (factor of three) than those in filtered samples. Differences between unfiltered and filtered samples may result from differences in the amount of colloid- or sediment-associated arsenic in the samples. Concentrations in both the unfiltered and the filtered samples from this site were greater than the USEPA current MCL, so interpretation was not affected significantly. However, interpretation of data from other sites could conceivably be affected by such differences between unfiltered and filtered samples and investigators will need to bear such potential differences in mind. Overall, however, combining filtered and unfiltered samples appears to be acceptable for definition of patterns of regional arsenic occurrence.

Differences in reported arsenic concentrations between the two analytical methods were small. Arsenic concentrations reported for samples analyzed by the USEPA method were slightly higher than those analyzed by the USGS method. These differences could be a result of differences in analytical methods. The longer digestion associated with the USEPA method could result in differences in reported arsenic concentrations. Different reagents used in sample digestion in the two methods also could result in differences in reported arsenic concentrations. However, the observed differences also could simply represent analytical variability.

Table 2. Comparison of arsenic concentrations for various processing and analytical methods

[USGS, U.S. Geological Survey; USEPA, U.S. Environmental Protection Agency; arsenic concentrations in micrograms per liter; processing and analytical methods described in text]

Well location	Filtered, 0.10-micrometer filter	Filtered, 0.45-micrometer filter	Unfiltered (USGS method)	Unfiltered (USEPA method)
01N/03W-04CCC	54	57	53	64
01N/03W-07CCD1	17	17	16	22
01N/03W-15ADB1	47	47	53	60
01S/03W-10BCA1	55	57	59	64
02S/02W-11CCD1	16	16	20	24
15S/01W-23CCA	18	18	18	21
17S/01W-24DCA	70	75	70	82
19S/01W-03ADB	41	43	40	46
21S/03E-08CBD2	62	64	180	180
18S/04W-14BBA	1,100	1,100	1,100	1,200

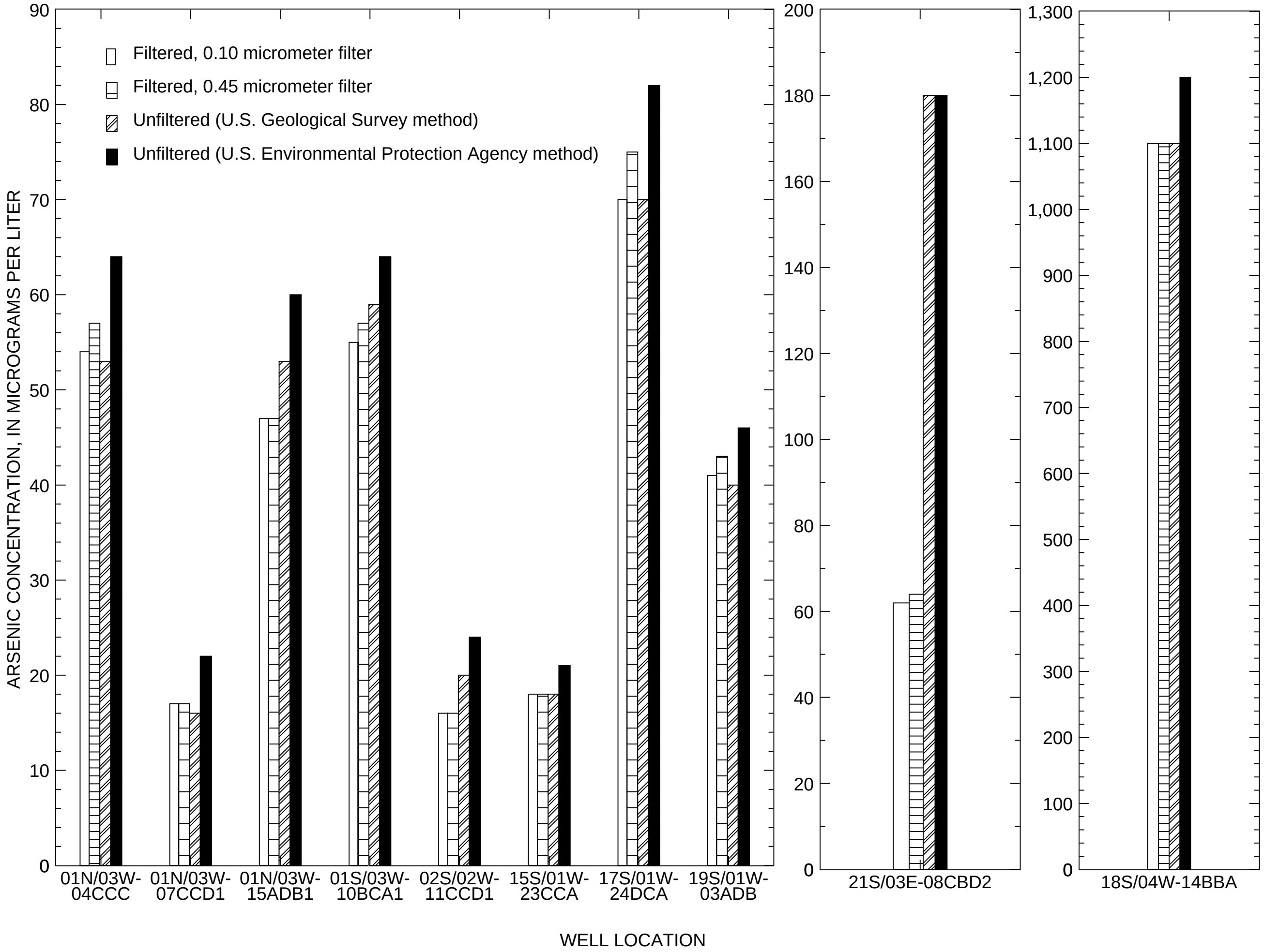


Figure 3. Comparison of arsenic concentrations determined by various processing and analytical methods.

DISTRIBUTION OF ARSENIC

The distribution of arsenic in ground water of the Willamette Basin is discussed in terms of temporal and spatial variability. First, project data are used to assess temporal variability. It is useful to assess temporal variability before arsenic-concentration data collected at different times is used to evaluate spatial distribution. Also, assessment of temporal variability should benefit well owners and water managers who desire guidance on when to sample for arsenic. Second, historical and project data are used to define spatial variability of arsenic concentrations in ground water in the Willamette Basin.

Temporal Variability

Project data were used to evaluate temporal variability of arsenic concentrations in ground water during a 1-year period. These data help characterize variability resulting from seasonal and other short- to medium-term factors. Characterization of temporal variability in arsenic concentrations over longer periods of time was not explicitly done, but long-term variability was discussed qualitatively in a previous section of this report, “Quality of Historical Data.”

Arsenic concentrations measured quarterly over a period of 1 year at 17 sites are given in table 3 and shown on figure 4. Field-measured specific conductance, a surrogate for dissolved solids, also is given in table 3. Arsenic concentrations did exhibit temporal variability. Although arsenic concentrations in water from many wells remained essentially constant over the course of sampling, concentrations at some sites varied by up to almost ±50 percent from mean concentrations, and arsenic concentrations in samples from well 19S/01W-

Table 3. Temporal variations in arsenic concentrations and specific conductance

[“As”, arsenic concentration in µg/L (micrograms per liter); “SC”, field-measured specific conductance in µS/cm (microsiemens per centimeter) at 25 degrees Celsius; “--”, not measured]

Well location	Date	As	SC	Date	As	SC	Date	As	SC	Date	As	SC	Date	As	SC
01N/03W-04CCC	11/21/96	53	587	02/20/97	52	585	05/15/97	82	496	08/19/97	97	522	12/05/97	72	616
01N/03W-07CCD1	11/19/96	16	345	02/20/97	17	357	05/15/97	19	345	08/19/97	26	314	12/05/97	18	345
01N/03W-15ADB1	11/21/96	53	1,220	02/20/97	47	1,300	05/15/97	64	1,390	08/19/97	63	1,500	12/05/97	52	1,450
01S/02W-29DBD	11/12/96	33	--	02/20/97	28	194	05/15/97	35	193	08/19/97	41	195	12/05/97	29	192
01S/02W-33BBA	11/12/96	12	--	02/20/97	10	274	05/15/97	12	272	08/19/97	13	275	12/05/97	12	274
01S/03W-10BCA1	11/19/96	59	341	02/20/97	52	340	05/15/97	62	339	08/19/97	56	339	12/05/97	56	341
02S/02W-11CCD1	11/19/96	20	335	02/20/97	16	335	05/15/97	18	334	08/19/97	19	336	12/05/97	18	336
15S/01W-23CCA	08/14/96	19	268	11/15/96	18	263	02/18/97	19	261	05/13/97	17	262	08/20/97	19	262
15S/01W-23CCC2	08/14/96	11	164	11/15/96	11	167	02/18/97	10	163	05/13/97	9	161	08/20/97	12	164
17S/01W-24DCA	09/06/96	85	194	11/15/96	70	198	02/19/97	74	195	05/13/97	84	191	08/21/97	69	193
18S/04W-14ACA	09/05/96	9	319	11/14/96	6	293	02/18/97	5	253	05/14/97	6	250	08/21/97	10	320
18S/04W-14BBA	09/04/96	830	1,040	11/15/96	1,100	1,070	02/20/97	640	797	[a]	[a]	[a]	08/20/97	1,100	1,060
19S/01W-03ADB	09/04/96	15	188	11/14/96	40	329	02/18/97	23	193	05/13/97	33	237	08/21/97	28	205
19S/03W-11E2	11/13/96	700	389	02/19/97	710	393	05/14/97	740	396	08/20/97	850	391	12/04/97	800	390
19S/03W-31E1	11/13/96	130	295	02/19/97	130	295	05/14/97	140	291	08/20/97	130	285	12/04/97	130	292
21S/03E-08CBD2	09/05/96	140	1,590	11/14/96	180	1,460	02/18/97	130	1,120	05/13/97	100	1090	09/04/97	69	1,450
22S/03W-17N	09/05/96	3	298	11/13/96	3	292	02/19/97	4	276	05/14/97	4	295	09/04/97	4	381

[a] Four samples within 37 hours:
05/13/97 at 7 p.m., As, 1100 µg/L; SC, 1090 µS/cm
05/14/97 at 10 a.m., As, 810 µg/L; SC, 952 µS/cm
05/14/97 at 8 p.m., As, 880 µg/L; SC, 809 µS/cm
05/15/97 at 8 a.m., As, 600 µg/L; SC, 877 µS/cm.

03ADB varied by a factor of 2.7 between the lowest and highest concentrations. The data as a whole demonstrate no obvious correlation with seasons. Analytical variability may be responsible for some of the observed variability, but can only explain up to about ±20-percent variation among samples. Differences in arsenic concentrations also could be due, in part, to temporally varying amounts of colloid- or sediment-bound arsenic. However, 10 of the 17 wells used for quarterly sampling also were used in the comparison of unfiltered and filtered samples. The resulting data showed little colloid- or sediment-bound arsenic present in samples from those wells at that time, except for well 21S/03E-08CBD2. Clearly, other factors are responsible for some of the observed temporal variability in arsenic concentrations.

It is likely that temporal variability in project data reflects variation in contributing sources of water to wells, and in the absence of seasonal patterns in temporal data, variation in contributing sources to wells is probably largely due to short-term (hour-to-hour or day-to-day) variations in well use prior to sampling. Relatively heavy well use can temporarily deplete water from parts of an aquifer adjacent to the well, and thus the well can yield water of different chemical quality than when sampled after a period of relatively light use. Two pieces of evidence suggest that some of the 17 wells sampled for temporal variability yield water from different sources at different times, although a relation between differences in contributing sources to wells and differences in well use remains only a hypothesis. One piece of evidence for changing water sources to wells lies in the specific conductance data. Specific conductance generally was less variable in samples where arsenic concentrations were less variable (table 3). Furthermore, specific conductance had the greatest relative temporal variability for water from well 19S/01W-03ADB; the same site also had the greatest relative temporal variability in arsenic concentrations (table 3). Because large changes in the chemistry of individual bodies of ground water generally take place over a period of years, it is difficult to explain large seasonal changes in specific conductance of well water by processes other than changing water sources to wells. A second piece of evidence suggesting changing water sources to wells is derived from examination of data from well 18S/04W-14BBA. Maximum and minimum arsenic concentrations in samples from this site varied by nearly a factor of two over the course of a year. At this site, some additional temporal sampling was conducted. Four samples were collected over one 37-hour period. The observed variability during a

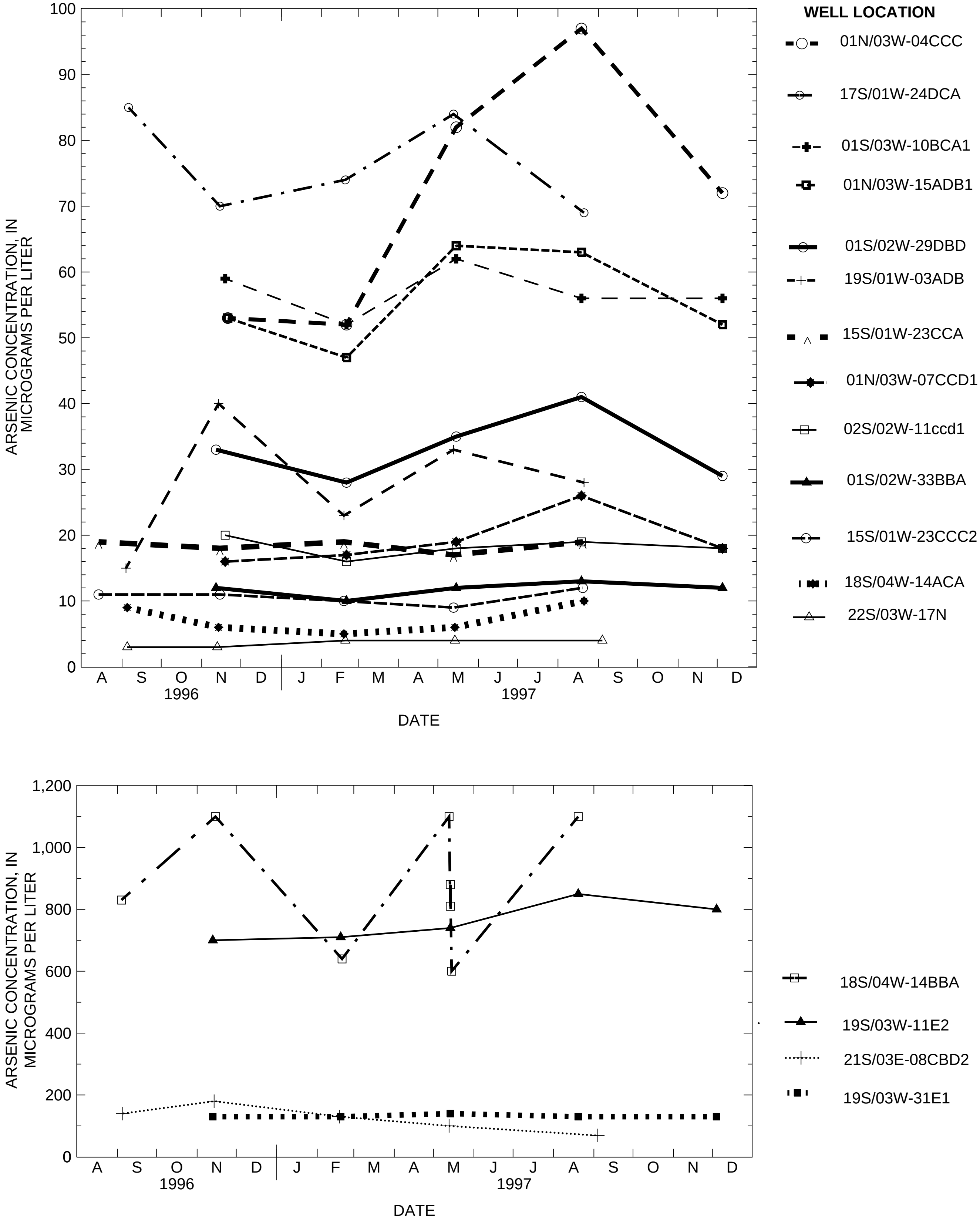


Figure 4. Temporal variations in arsenic concentrations.

37-hour period was as great as the variability observed during the course of a year (table 3, fig. 4). Such variability over the course of 37 hours cannot be ascribed to seasonal factors.

Certainly, a relationship between arsenic concentrations and well use prior to sampling remains only a hypothesis. But regardless of the processes resulting in the observed temporal variability, the data demonstrate that short-term variability in arsenic concentrations can be similar in magnitude to variability observed during the course of quarterly sampling.

The temporal variability of the project data (less than a factor of three) contrasts greatly with temporal variability in arsenic concentrations reported by Nadakavukaren and others (1984) for some wells in Lane County in the southern part of the Willamette Basin. Nadakavukaren and others (1984) reported temporal variability of up to about three orders of magnitude over the course of a year for some of the 14 wells sampled. Such variability is intriguing, because temporal variability of this magnitude in ground-water chemistry at individual sites is unusual.

Nadakavukaren and others (1984) noted that arsenic concentrations often were low (relative to mean concentrations) during the winter (rainy) season. However, they also reported that equally low concentrations were observed during other seasons, including the summer (dry) season, at several sites. Thus, although they observed temporal variability in arsenic concentrations, Nadakavukaren and others (1984) reported that they were unable to relate temporal variability to environmental factors. Unfortunately, sample-collection procedures were not defined in the paper. One aspect of the data not discussed in the original paper, but that may have significant bearing on interpretation of temporal variability, is that most of the wells sampled were irrigation wells. Irrigation wells tend to be unused during the rainy season, and frequently remain idle for long periods during other parts of the year. Recall that it was proposed that temporal variability of project data was related, at least in part, to well use prior to sampling, and recall, also, that project wells either were actively used domestic wells or were sampled after purging three well-bore volumes. It is possible that the extreme variability in arsenic concentrations reported by Nadakavukaren and others (1984) could be related to previous well use (or lack of well use), especially if the wells were not purged prior to sampling. Certainly, the use of irrigation wells for most of the work presented by Nadakavukaren and others (1984), and the absence of documentation of well-purging criteria, make interpretation of their temporal data difficult. Data of Nadakavukaren and others (1984) suggest that caution be applied when using historical data for which both well-use and well-purging information are unavailable.

The absence of seasonal trends in project data suggests that data collected at different times in the Willamette Basin can be combined for use in definition of spatial variability in arsenic concentrations. However, data of Nadakavukaren and others (1984), although difficult to interpret, suggest that historical data for which the history of well use and well purging are unknown may not always be sufficient for site-specific characterization. Thus, although temporal variability is not likely to be a significant problem for a regional evaluation of ground-water arsenic concentrations, the quality of historical data should be evaluated. Most of the historical data compiled for use in this report were from samples collected by USGS personnel. The remainder of the data (from Linn County Department of Health Services, Environmental Health Program) were collected from domestic wells, which presumably were actively used wells. USGS protocols have long required (at least as far back as 1960; Rainwater and Thatcher, 1960) that ground-water samples be collected from purged or actively used wells. Thus, from a standpoint of well use/well purging, the historical data used in this report are believed to be of adequate quality for a regional assessment of arsenic concentrations in ground water. Resampling of selected wells represented in the historical data supports this assumption, as was shown in the section “Quality of Historical Data.”

Spatial Distribution

Historical and project data were combined and used to evaluate the spatial distribution of arsenic concentrations in ground water of the Willamette Basin. A total of 728 spatially distinct samples thus were available—597 historical and 131 project samples. Of these 728 samples, 721 were from wells, and 7 were from nonthermal springs. These data are available in digital format

(CD-ROM) in a separate data report (Orzol and others, in press).

Concentrations of arsenic in the 728 samples ranged from < 1 to 2,000 µg/L. A histogram of these data is shown on figure 5. Concentrations in 58 samples (8.0 percent) exceeded the USEPA current MCL (50 µg/L), and 158 (21.7 percent) exceeded the WHO provisional guideline (10 µg/L). The 728 samples were not randomly distributed throughout the basin, so it does not follow that 8 percent of all wells in the basin will exceed the USEPA current MCL. Furthermore, because some of the data (in particular, data of Goldblatt and others, 1963, and data from Linn County Department of Health Services) were collected to address suspected arsenic problems, the cumulative data set contains a bias towards high arsenic concentrations (exceeding the USEPA current MCL). The data do, however, indicate the existence of extensive bodies of high-arsenic ground water in the basin.

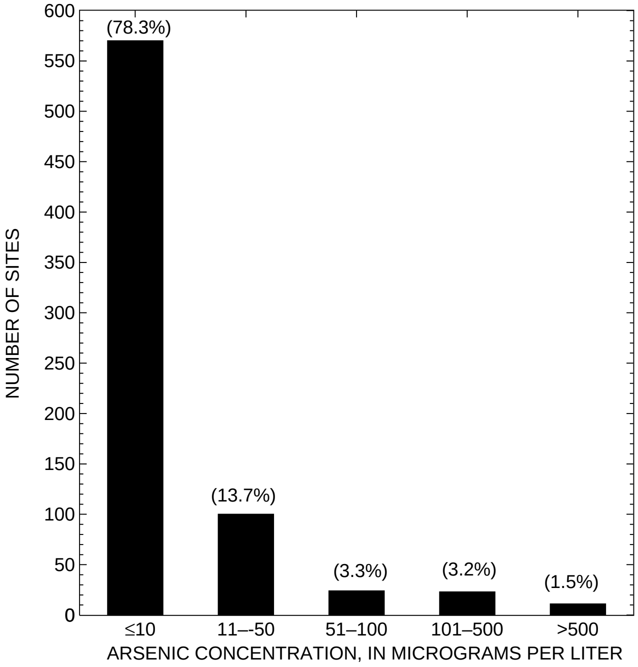


Figure 5. Arsenic concentrations for sites sampled in the Willamette Basin, Oregon. (Number in parentheses is percentage of the total number of sites.)

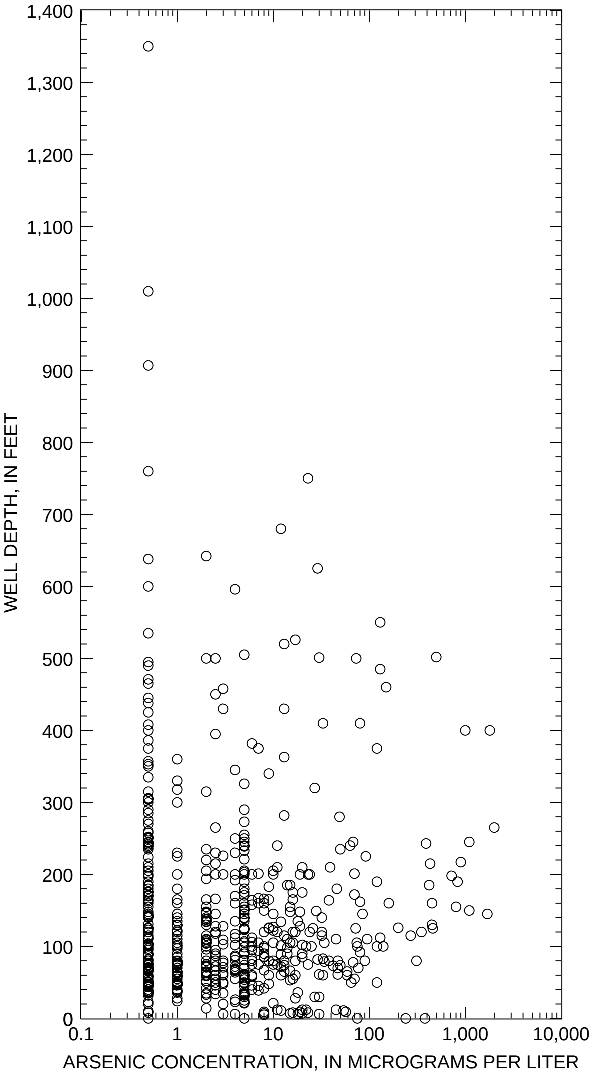


Figure 6. Relation of arsenic concentration to well depth. (Data from six springs also are included; “well depth” is set to zero for springs.)

Depth data were available for 651 of the 728 sites. The relation of arsenic concentration to depth is shown on figure 6. Data from springs were included on this figure; springs were assigned a “well depth” of zero. (For plotting purposes, censored data [concentrations below reporting levels] were arbitrarily plotted at one-half of the reporting levels. Censoring occurred at three concentration levels: 1, 5, and 10 µg/L.) No obvious relation of arsenic concentration to well depth was observed. On a regional scale, depth does not appear to be a useful parameter for predicting arsenic concentrations. However, depth may be an important parameter on a local scale.

The spatial distribution of arsenic concentrations is shown on plate 1. Patterns of arsenic occurrence are apparent. Most (53) of the 58 samples with high concentrations of arsenic came from wells and springs in bedrock areas (areas where bedrock is exposed at land surface or is covered by thin layers of alluvium) in south-central and eastern Lane County, and Linn County (pl. 1). The remaining 5 (of the 58) samples came from wells near the center of the Tualatin Basin in Washington County (northwestern part of the Willamette Basin) (pl. 1). These five wells produce water from alluvial deposits. Not only were most of the 58 occurrences of high arsenic concentrations in Lane and Linn Counties, but the highest concentrations also were found there. Arsenic concentrations ranged up to 2,000 µg/L in Lane and Linn Counties, and six samples contained ≥1,000 µg/L. In contrast, the maximum concentration of arsenic in the Tualatin Basin, 77 µg/L, was substantially smaller than many of the concentrations found in Lane and Linn Counties, although still a concentration of considerable concern.

All five exceedances of the USEPA current MCL in the Tualatin Basin were from filtered samples. Many of the exceedances of the USEPA current MCL in Lane and Linn Counties were from unfiltered samples, but concentrations in filtered samples from that part of the Willamette Basin have been observed to exceed 1,000 µg/L. Because filtered samples generally contain primarily dissolved constituents, the presence of high concentrations of arsenic in filtered samples suggests that geochemical conditions can be favorable for development of high dissolved-arsenic concentrations both in the Tualatin Basin and in bedrock areas of Lane and Linn Counties.

Intermediate arsenic concentrations (>10 µg/L and ≤ 50 µg/L) were widespread in the Willamette Basin (pl. 1). As might be expected, many of the occurrences of intermediate arsenic concentrations were located in the same regions where high arsenic concentrations were found. However, intermediate arsenic concentrations were found in many other areas as well, and in a variety of geologic materials.

Occurrence of high concentrations of arsenic in bedrock areas of Lane and Linn Counties appears to be related to the areal extent of two associations of older volcanic rocks: (1) the Fisher and Eugene Formations and correlative rocks (Oligocene and upper Eocene epochs), and (2) undifferentiated tuffaceous sedimentary rocks, tuffs, and basalt (Miocene and Oligocene epochs). (The undifferentiated tuffaceous sedimentary rocks, tuffs, and basalt are approximately equivalent to the Little Butte Volcanic Series of Peck and others [1964].) The surficial extent of these two rock associations is shown on plate 1. At land surface, the two rock associations cover 24 percent of the Willamette Basin. All detections of high concentrations of arsenic in Lane and Linn Counties occur in or very close to places where these volcanic rocks crop out, or in areas where thin layers of alluvial materials cover the rocks. These rocks include extensive volumes of silicic (rhyolitic) volcanic rocks. Ground water high in naturally occurring arsenic commonly is associated with volcanic rocks silicic to intermediate in composition (Welch and others, 1988). Thus, the apparent relationship between high concentrations of arsenic and geologic unit is not unexpected.

Interpretation of relationships between high concentrations of arsenic in ground water and geologic units could be improved upon at a local scale by use of more detailed (local) geologic maps. For example, although high concentrations of arsenic often occur in water within the Fisher and Eugene Formations and correlative rocks, Goldblatt and others (1963) suggest that the Fisher Formation, and not the Eugene Formation, is the source of most of the arsenic in that area. Similarly, water within basalt flows in the undifferentiated tuffaceous sedimentary rocks, tuffs, and basalt is not a likely candidate for high concentrations of arsenic because basalt typically yields water low in arsenic (Welch and others, 1988). The regional nature of the work presented in this report, with the requisite use of regional-scale rock associations, did not allow for finer-scale interpretation of the occurrence of high concentrations of arsenic relative to geologic characteristics. However, investigators involved in local-scale ground-water assessments should be able to make use of more detailed geologic mapping to help guide sampling.

Large portions of the area covered by the Fisher and Eugene Formations and correlative rocks, and the undifferentiated tuffaceous sedimentary rocks, tuffs, and basalt, are not represented by data collected and compiled for this report. Although most of the unsampled areas underlain by these rocks are not densely populated, they are not uninhabited, and the potential for impacts to human health are not insignificant. The surface exposure of these rocks alone represents 24 percent of the area of the Willamette Basin, and their full extent is greater. Additional sampling of wells completed in these arsenic-containing rocks would better define the spatial distribution of high-arsenic water in areas not sampled during this study. Further, the presence of high arsenic concentrations in other aquifers in the Willamette Basin (pl. 1) suggests that additional sampling might reveal still more problem areas.

GEOCHEMISTRY OF ARSENIC

An understanding of factors controlling the distribution of arsenic in ground water requires a knowledge of arsenic sources and of processes controlling arsenic mobility. To that end, possible sources of arsenic in Willamette Basin ground water are discussed in this section. Processes that have been shown to control arsenic mobility in other natural systems are discussed next. Then, arsenic speciation data collected as part of this project, along with some historical speciation data, are presented. Finally, geochemical data (including the speciation data) and information from existing interpretive reports are used to construct preliminary hypotheses regarding possible geochemical controls over mobilization of arsenic in the Willamette Basin. An understanding of arsenic sources and geochemistry in the basin could help guide future monitoring efforts both in the basin and elsewhere. However, rigorous geochemical investigation of reasonable hypotheses will be required before an adequate understanding of arsenic geochemistry in the Willamette Basin can be said to exist. Thus, this discussion may serve future research.

Sources of Arsenic

Arsenic can be introduced into ground water from anthropogenic and natural sources. Anthropogenic sources may be important in some settings. Because industrial activity tends to be localized, it would be difficult to explain regional patterns of arsenic occurrence in the Willamette Basin by introduction from industrial sources. However, arsenical pesticides such as lead arsenate were historically used in large quantities in agricultural areas of the Willamette Basin (Rinehold and Jenkins, 1993). High-arsenic ground water in bedrock areas of Lane and Linn County tends to occur in nonagricultural areas, so it is unlikely that the observed high concentrations of arsenic in ground water in those areas can be attributed to historical use of arsenical pesticides. However, in contrast to land-use patterns in the bedrock areas of Lane and Linn Counties, land use in alluvial portions of the Tualatin Basin includes a variety of agricultural land uses, and high-arsenic ground water in alluvium in the Tualatin Basin does generally coincide with occurrence of agricultural areas. Closer inspection of the data, however, shows that detections of high concentrations of arsenic in Tualatin Basin ground water generally are near rivers and streams (pl. 1). Ground water near these rivers and streams likely represents ground water near the end of ground-water flowpaths. Occurrence of high concentrations of arsenic in downgradient parts of ground-water flowpaths could result from transport of arsenic from upgradient areas where arsenical pesticides historically had been applied, or from mobilization of naturally occurring arsenic during geochemical evolution as ground water moves along flowpaths. Arsenic is nearly immobile in topsoils, and arsenic in arsenical-pesticide-contaminated topsoil leaches on timescales of decades or more (Aten and others, 1980). Thus, occurrence of high concentrations of arsenic primarily in downgradient areas, and not more uniformly distributed in the Tualatin Basin, is more consistent with a natural source than an anthropogenic source. However, no rigorous ground-water flowpath analysis has been done for arsenic transport in the Tualatin Basin, and instances of leaching of arsenic from sites of historical arsenical use into ground water of the Tualatin Basin cannot be ruled out. Nonetheless, regional patterns of arsenic concentrations in ground water of the Willamette Basin as a whole probably reflect primarily natural sources.

Naturally occurring arsenic commonly is found in volcanic glass in volcanic rocks of rhyolitic to intermediate composition; adsorbed to and coprecipitated with metal oxides, especially iron oxides; adsorbed to clay-mineral surfaces; and associated with sulfide minerals and organic carbon (Welch and others, 1988). Sulfide minerals can contain arsenic either as a dominant mineral-forming element or as an impurity; sulfide minerals are found locally in the Western Cascades (U.S. Geological Survey, 1969). Metal oxides and clay minerals are ubiquitous in the Willamette Basin. Organic carbon is widespread in many parts of the Willamette Basin, especially in alluvial deposits. Volcanic glass, commonly a major component of volcanic rocks, also is widely found in Willamette Basin aquifers, although much of the original glass in older volcanic rocks has been devitrified (Peck and others, 1964). Thus, arsenic originally associated with such volcanic glass either will have become associated with devitrification alteration products such as clays and metal oxides, or will have been released into solution and subsequently adsorbed or precipitated elsewhere or flushed from the aquifer. However, volcanic glass is still abundant in the Willamette Basin, and thus may constitute a current source of arsenic. At a minimum, the apparent relationship between rock associations containing silicic volcanic rocks and the occurrence of high concentrations of arsenic in ground water in Lane and Linn Counties described earlier in this report suggests that considerable amounts of arsenic might ultimately have come from volcanic glass. Thus, several sources of naturally occurring arsenic dispersed in aquifer materials can reasonably be postulated. However, examination of these various potential arsenic sources for arsenic availability in the Willamette Basin apparently has never been done.

Review of Geochemical Processes Controlling Arsenic Mobility

Two categories of processes largely control arsenic mobility in aquifers: (1) adsorption and desorption reactions and (2) solid-phase precipitation and dissolution reactions. Attachment of arsenic to an iron oxide surface is an example of an adsorption reaction. The reverse of this reaction, arsenic becoming detached from such a surface, is an example of desorption. Solid-phase precipitation is the formation of a solid phase from components present in aqueous solution. Precipitation of the mineral calcite, from calcium and carbonate present in ground water, is an example of solid-phase precipitation. Dissolution of volcanic glass within an aquifer is an example of solid-phase dissolution.

Arsenic adsorption and desorption reactions are influenced by changes in pH, occurrence of redox (reduction/oxidation) reactions, presence of competing anions, and solid-phase structural changes at the atomic level. Solid-phase precipitation and dissolution reactions are controlled by solution chemistry, including pH, redox state, and chemical composition.

Adsorption and Desorption Processes

Arsenic is a redox-sensitive element. This means that arsenic may gain or lose electrons in redox reactions. As a result, arsenic may be present in a variety of redox states. Arsenate and arsenite are the two forms of arsenic commonly found in ground water (Masscheleyn and others, 1991). Arsenate generally predominates under oxidizing conditions. Arsenite predominates when conditions become sufficiently reducing. Under the pH conditions of most ground water, arsenate is present as the negatively charged oxyanions $H_2AsO_4^-$ or $HAsO_4^{2-}$, whereas arsenite is present as the uncharged species $H_3AsO_3^0$ (Hem, 1985). The strength of adsorption and desorption reactions between these different arsenic species and solid-phase surfaces in aquifers varies, in part, because of these differences in charge. Differences in species charge affect the character of electrostatic interactions between species and surfaces.

Arsenate and arsenite adsorb to surfaces of a variety of aquifer materials, including iron oxides, aluminum oxides, and clay minerals. Adsorption and desorption reactions between arsenate and iron-oxide surfaces are particularly important controlling reactions because iron oxides are widespread in the hydrogeologic environment as coatings on other solids, and because arsenate adsorbs strongly to iron-oxide surfaces in acidic and near-neutral-pH water (Dzombak and Morel, 1990; Waychunas and others, 1993). However, desorption of arsenate from iron-oxide surfaces becomes favored as pH values become alkaline (Fuller and Davis, 1989; Dzombak and Morel, 1990). The pH-dependence of arsenate adsorption to iron-oxide surfaces appears to be related to the change in iron-oxide net surface charge from positive to negative as pH increases above the zero-point-of-charge (pH at which the net surface charge is equal to zero) of about 7.7 for goethite (crystalline iron oxide) (Stumm and Morgan, 1996) or 8.0 for ferrihydrite (amorphous iron oxide) (Dzombak and Morel, 1990). Where pH values are above about 8, the negative net surface charge of iron oxide can repel negatively charged ions such as arsenate.

Iron-oxide surfaces also adsorb arsenite, and both arsenate and arsenite adsorb to aluminum oxides and clay-mineral surfaces. However, these adsorption reactions appear generally to be weaker than is the case for arsenate adsorption to iron-oxide surfaces under typical environmental pH conditions (Manning and Goldberg, 1997). Nevertheless, pH-dependent adsorption and desorption reactions other than those between arsenate and iron-oxide surfaces may be important controls over arsenic mobility in some settings. As is the case for adsorption of arsenate to iron-oxide surfaces, adsorption of arsenite to iron-oxide surfaces tends to decrease as pH increases, at least between the range from pH 6 to pH 9 (Dzombak and Morel, 1990). Unfortunately, arsenate and arsenite adsorption and desorption reactions with other common surfaces are less well characterized, and apparently more complex than is the case for adsorption and desorption reactions with iron-oxide surfaces (Manning and Goldberg, 1997).

As a result of the pH dependence of arsenic adsorption, changes in ground-water pH can promote adsorption or desorption of arsenic. Because solid-phase diagenesis (water-rock interaction) typically consumes H^+ (Stumm and Morgan, 1996), the pH of ground water tends to increase with residence time, which, in turn, increases along ground-water flowpaths. Because iron-oxide surfaces can hold large amounts of adsorbed arsenate, geochemical evolution of ground water to high (alkaline) pH can induce desorption of arsenic sufficient to result in exceedances of the USEPA current MCL in some environments (see, for example, Robertson, 1989).

Similarly, redox reactions can control aqueous arsenic concentrations by their effects on arsenic speciation, and hence, arsenic adsorption and desorption. For example, reduction of arsenate to arsenite can promote arsenic mobility because arsenite is generally less strongly adsorbed than is arsenate. Redox reactions involving either aqueous or adsorbed arsenic can affect arsenic mobility (Manning and Goldberg, 1997).

Arsenic adsorption also can be affected by the presence of competing ions. In particular, phosphate and arsenate have similar geochemical behavior, and as such, both compete for sorption sites (Hingston and others, 1971; Livesey and Huang, 1981; Manning and Goldberg, 1996). Oxyanions in addition to phosphate also may compete for sorption sites. For example, Robertson (1989) suggested that correlation of arsenate with oxyanions of molybdenum, selenium, and vanadium in ground water of the Southwestern United States may be evidence for competitive adsorption among those oxyanions.

Finally, structural changes in solid phases at the atomic level also affect arsenic adsorption and desorption. For example, conversion of ferrihydrite to goethite or to other crystalline iron-oxide phases may occur gradually over time (Dzombak and Morel, 1990). Fuller and others (1993) demonstrated that as ferrihydrite crystallizes into goethite, the density of arsenic adsorption sites decreases. This decrease in density of adsorption sites can result in desorption of adsorbed arsenic. Structural changes in other solid phases may possibly affect arsenic mobility, too. The role of such solid-phase structural changes on ground-water arsenic concentrations has, however, received little attention to date.

Precipitation and Dissolution Processes

The various solid phases (minerals, amorphous oxides, volcanic glass, organic carbon) of which aquifers are composed exist in a variety of thermodynamic states. At any given time, some aquifer solid phases will be undergoing dissolution, whereas others will be precipitating from solution. Arsenic contained within solid phases, either as a primary structural component of or an impurity in any of a variety of solid phases, is released to ground water when those solid phases dissolve. Similarly, arsenic is removed from ground water when solid phases containing arsenic precipitate from aqueous solution. As an example, because arsenic often coprecipitates with iron oxide (Waychunas and others, 1993), iron oxide may act as an arsenic source (case of dissolution) or a sink (case of precipitation) for ground water. Furthermore, solid-phase dissolution will contribute not only arsenic contained within that phase, but also any arsenic adsorbed to the solid-phase surface. The process of release of adsorbed arsenic as a result of solid-phase dissolution is distinct from the process of desorption from stable solid phases.

The interplay of redox reactions and solid-phase precipitation and dissolution may be particularly important with regard to aqueous arsenic and solid-phase iron oxides and sulfide minerals. High concentrations of arsenic often are associated with iron oxides and sulfide minerals (Thornton, 1996). Iron oxides frequently dissolve under reducing conditions, but often precipitate under oxidizing conditions. Sulfide minerals generally are unstable under oxidizing conditions, but may precipitate under reducing conditions. Thus, as a result of the redox-sensitive nature of iron oxides and sulfide minerals, transfer of large amounts of arsenic between these solid phases and neighboring water may result from redox-facilitated precipitation and dissolution reactions.

Arsenic Speciation in the Willamette Basin

Three samples from alluvial wells in the Tualatin Basin and two from bedrock wells in Lane County were analyzed for four common species of arsenic. Concentrations of the two organic species of arsenic analyzed (monosodium methylarsonate, or CH_3AsO_3HNa, and sodium dimethylarsinate, or $(CH_3)_2AsO_2Na$) were below MDLs, so only the data for the two inorganic species are tabulated. These speciation data are given in table 4.

Two additional analyses for arsenic species in ground water from the Willamette Basin were available in the literature. Welch and others (1988) reported arsenite and arsenate concentrations for water from two wells in Lane County. Arsenite represented 7 percent and 62 percent of the total arsenic in these samples (total arsenic concentrations being 25 and 45 μg/L, respectively).

The most striking feature of the data from the two studies is the predominance of arsenite. The predominance of arsenite has both geochemical and toxicological implications. From a geochemical standpoint, the speciation data are of interest because mobility of arsenite differs from that of arsenate (see section “Review of Geochemical Processes Controlling Arsenic Mobility”). From a public-health perspective, the speciation data are interesting because arsenite is more toxic than arsenate in at least some of its effects. In human acute toxicity studies, arsenite has been shown to be more potent than arsenate (U.S. Environmental Protection Agency, 1988). With regard to human chromosome breakage, arsenite is about an order of magnitude more potent than arsenate (U.S. Environmental Protection Agency, 1988). Morrison and others (1989) report that arsenite is 50 times as toxic as arsenate, but do not report the organisms studied. Also, arsenite is more difficult to remove from drinking-water supplies than is arsenate (Gupta and Chen, 1978; Schneiter and Middlebrooks, 1983). However, it would be premature to make generalizations regarding arsenic toxicity in the Willamette Basin based upon such limited speciation data (seven samples). Complicating the matter, in the benchmarks against which drinking water arsenic concentration data commonly are compared—the USEPA current MCL and the WHO provisional guideline—no differen- tiation is made between arsenite and arsenate. But if the apparent predominance of arsenite in Willamette Basin ground water is confirmed by additional speciation work, public health officials and water managers may need to evaluate the scope of the arsenic problem with regard not only to arsenic concentrations, but also to arsenic speciation.

Table 4. Speciation of arsenic

[Total arsenic concentration is from a separate analysis of a separate sample, and differs from the total of arsenite-plus-arsenate because of sampling and (or) analytical variability. Recovery, total of arsenite-plus-arsenate divided by total arsenic; μg/L, micrograms per liter]

Well location	Date	Arsenite (percent of total of arsenite-plus-arsenate)	Arsenate (percent of total of arsenite-plus-arsenate)	Arsenite (μg/L)	Arsenate (μg/L)	Total of arsenite-plus-arsenate (μg/L)	Total arsenic concentration (μg/L)	Recovery (percent)
01N/03W-04CCC	08/19/97	76	24	61.1	18.9	80.0	97	82
01S/03W-10BCA1	08/19/97	96	4	58.8	2.3	61.1	56	110
02S/02W-11CCD1	08/19/97	94	6	15.3	.9	16.2	19	85
18S/04W-14BBA	08/20/97	>99	<1	1,200	6.1	1,210	1,100	110
19S/03W-31E1	08/20/97	68	32	61.5	29.4	90.9	130	70

Geochemistry of Arsenic in the Willamette Basin

Few routine chemical analyses (of major ions and field parameters) are available for high-arsenic ground water from bedrock areas of Lane and Linn Counties. Goldblatt and others (1963) noted that high-arsenic ground water tended to have high pH (>8.0) and high orthophosphate concentrations, although only two routine chemical analyses for high-arsenic ground water were published. Reliable measures of redox conditions were not collected. However, the observation was made during site visits that water from many of the wells in bedrock areas of Lane and Linn Counties that yielded high-arsenic ground water during project sampling also had sulfide odors. The presence of sulfide in water indicates chemically reducing conditions. The observation of sulfide, along with the detection of arsenite (the more reduced of the two major arsenic species) in some ground-water samples, indicates the presence of reducing conditions in some ground water in these areas.

Together, these data suggest that for ground water in bedrock areas of Lane and Linn Counties, one or more of the following controlling factors likely are important in adsorption and desorption reactions that in turn often control arsenic mobility: (1) high pH, (2) presence of competing anions, and (3) occurrence of reducing conditions. The sparse available data do not allow even for speculation about adsorption and desorption reactions related to solid-phase structural changes at the atomic level in ground water of bedrock areas of Lane and Linn Counties. Similarly, evidence is lacking to even begin to develop hypotheses about solid-phase precipitation and dissolution reactions.

Previous investigations of the quality of Tualatin Basin ground water provide some preliminary insight into arsenic geochemistry there. Rounds and others (1994) reported that high phosphorus concentrations (up to 2.9 mg/L [milligrams per liter]) are common in Tualatin Basin ground water. In an analysis of 47 filtered ground-water samples from the Tualatin Basin, Hinkle (1997) reported that the median arsenic concentration in low-dissolved-oxygen samples (dissolved oxygen concentrations < 1.0 mg/L) was greater than the median arsenic concentration in well-oxygenated samples. The difference was statistically significant. Of the 47 samples, the 4 that exceeded the USEPA current MCL not only had low dissolved-oxygen concentrations, but also had high concentrations of orthophosphate (0.36 to 2.0 mg/L) and iron (160 to 1,900 µg/L). However, pH was not unusually high; pH of three of the four high-arsenic samples ranged from 7.5 to 7.6, and was 8.1 for the fourth sample.

These data suggest that for alluvial ground water in the Tualatin Basin, presence of competing anions and occurrence of reducing conditions may be important controlling factors in arsenic adsorption and desorption reactions. These two factors might be more important than pH controls over arsenic adsorption and desorption. Reducing conditions and high concentrations of dissolved iron also suggest that dissolution of iron oxides, with subsequent release of adsorbed or coprecipitated arsenic, may play a role in arsenic mobility in the Tualatin Basin.

Hypotheses about factors affecting arsenic adsorption and desorption reactions should account for arsenic speciation. Limited geochemical data suggest that desorption of arsenic from solid phases may be an important process in ground water both in bedrock areas of Lane and Linn Counties and in alluvium in the Tualatin Basin. Desorption of arsenate from iron oxides commonly results from high pH or the presence of competing ions. Such processes, of course, require the presence of arsenate on solid-phase surfaces. Because redox reactions often are slow and frequently far from equilibrium, it would not be unexpected to find arsenate adsorbed to solid-phase surfaces in chemically reducing environments. It might appear, though, that the predominance of arsenite relative to arsenate in aqueous speciation samples would be inconsistent with a hypothesis of desorption of arsenate from iron-oxide surfaces. However, it may be that arsenate is desorbed from aquifer surfaces and subsequently reduced to arsenite.

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APPENDIX

APPENDIX

Table 1. Wells discussed in report text, tables, and figures, listed by well location name from original source, and cross-referenced by recalculated well location and by U.S. Geological Survey site identification number and Oregon Water Resources Department well log identification number

[“--”, identical to “Well location (original source)”; N/A, none or not determined].

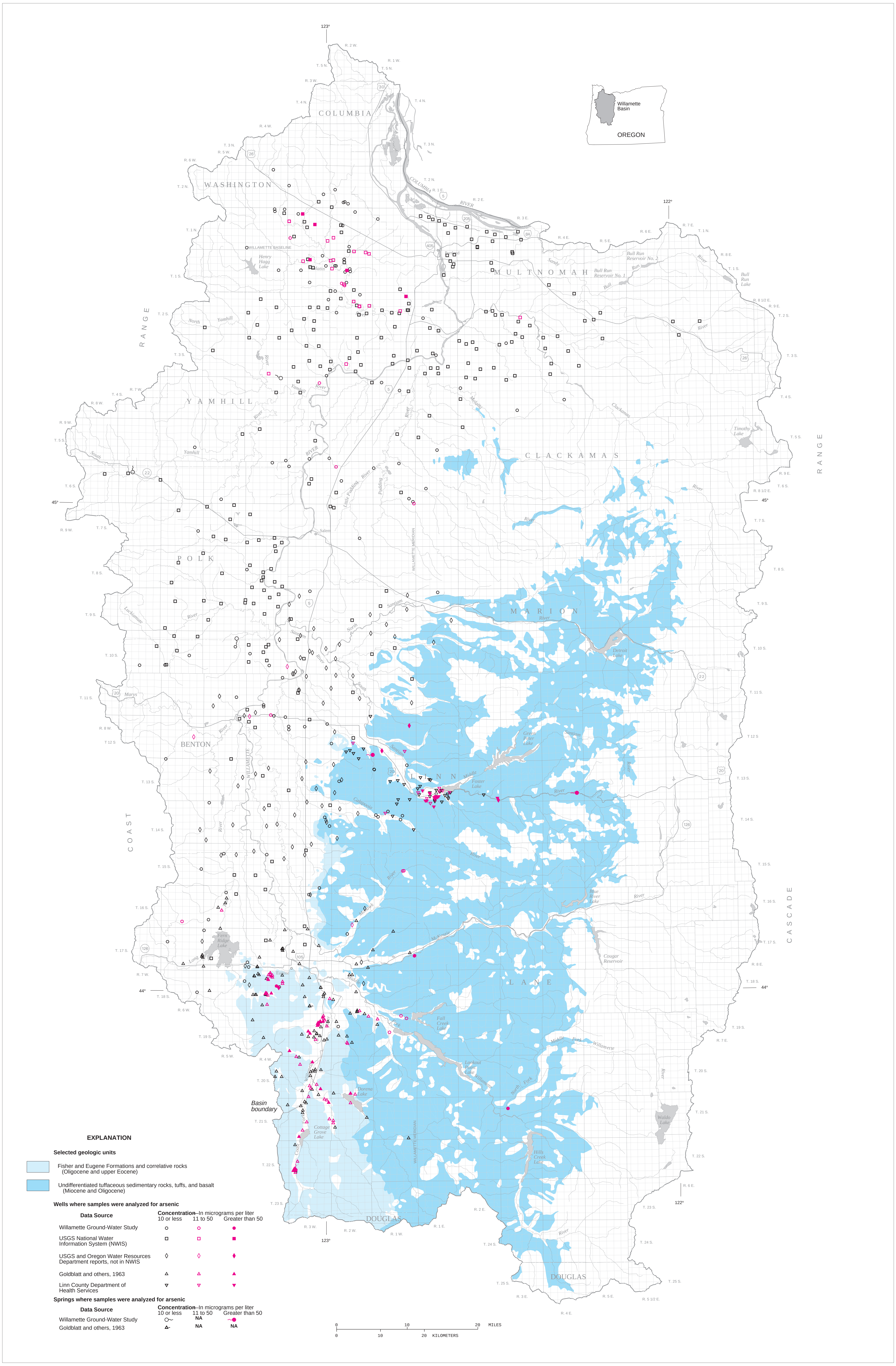
Well location (original source)	Well location (recalculated)	U.S. Geological Survey site identification number	Oregon Water Resources Department well log identification number
01N/03W-04CCC	--	453540123041101	WASH 5967
01N/03W-07CCD1	--	453445123063201	WASH 6037
01N/03W-15ADB1	--	453422123020201	N/A
01S/02W-29DBD	--	452707122572201	WASH 10406
01S/02W-33BBA	--	452651122565001	WASH 10475
01S/03W-10BCA1	--	453002123025301	WASH 143
02S/02W-11CCD1	--	452416122541601	WASH 12572
12S/01W-29N1	12S/01W-29CCA	442934122502801	LINN 9588
13S/01E-33	13S/01E-33DB1	442332122412701	LINN 12832
13S/01E-33AC	--	442348122412301	LINN 12776
13S/01E-35	13S/01E-35BD1	442347122391001	LINN 12914
14S/01E-05	13S/01E-32CD1	442323122424801	LINN 10997
15S/01W-23CCA	--	441447122464501	LANE 50736
15S/01W-23CCC2	--	441446122465701	LANE 5873
17S/01W-24DCA	--	440420122445701	LANE 2085
18S/04W-10D	18S/04W-10BB1	440125123095901	N/A
18S/04W-14ACA	--	440029123080301	LANE 17048
18S/04W-14ACB	--	440024123080901	LANE 17052
18S/04W-14BBA	--	440036123083201	LANE 16780
18S/04W-22B	18S/04W-22BA1	435942123092501	N/A
19S/01W-03ADB	--	435656122471801	LANE 19429
19S/03W-11E2	19S/03W-11BC3	435606123012501	N/A
19S/03W-31E1	19S/03W-31BB1	435237123061801	N/A
21S/03E-08CBD2	--	434528122290901	LANE 23527
22S/03W-17N	22S/03W-17CC1	433859123045601	N/A

U.S. DEPARTMENT OF THE INTERIOR
U.S. GEOLOGICAL SURVEY

Prepared in cooperation with
OREGON WATER RESOURCES DEPARTMENT

WATER-RESOURCES INVESTIGATIONS REPORT 98-4205
Arsenic concentrations in ground water, and selected geologic units—PLATE 1

Hinkle, S.R. and Polette, D.J.
Arsenic in Ground Water of the Willamette Basin, Oregon



EXPLANATION

Selected geologic units

▭	Fisher and Eugene Formations and correlative rocks (Oligocene and upper Eocene)
▬	Undifferentiated tuffaceous sedimentary rocks, tuffs, and basalt (Miocene and Oligocene)

Wells where samples were analyzed for arsenic

Data Source	Concentration—In micrograms per liter 10 or less	 11 to 50	 Greater than 50
Willamette Ground-Water Study	○	○	●
USGS National Water Information System (NWIS)	□	□	■
USGS and Oregon Water Resources Department reports, not in NWIS	◇	◇	◆
Goldblatt and others, 1963	△	△	▲
Linn County Department of Health Services	▽	▽	▼

Springs where samples were analyzed for arsenic

Data Source	Concentration—In micrograms per liter 10 or less	 11 to 50	 Greater than 50
Willamette Ground-Water Study	○~	NA	~●
Goldblatt and others, 1963	△~	NA	NA

Base modified from U.S. Geological Survey Digital base from
U.S. Geological Survey Digital Line Graphs published at 1:100,000, 1987;
U.S. Bureau of the Census, TIGER/Line(R), published at 1:100,000, 1990;
U.S. Geological Survey Digital Line Graphs published at 1:100,000, 1988
and modified by the State of Oregon. 1994;
U.S. Geological Survey quadrangle maps published at 1:24,000
as digitized by Oregon Department of Transportation;
Universal Transverse Mercator projection, Zone 10
1927 North American Datum

DISTRIBUTION OF ARSENIC CONCENTRATIONS IN GROUND WATER, AND SELECTED GEOLOGIC UNITS OF THE WILLAMETTE BASIN, OREGON

By
Danial J. Polette and Stephen R. Hinkle

Geology from Walker and MacLeod, U.S. Geological Survey Digital Line
published at 1:500,000, 1991 and modified by the State of Oregon, 199

From: Joel Geier
To: Benton Public Comment
Subject: LU-24-027: Information to add to the record regarding groundwater
Date: Saturday, May 3, 2025 6:48:59 PM
Attachments: Surface Water and Groundwater InteractionsinaWatershedSystemintheWillametteBasinin Western Oregon
Ochoa_etal_2022.pdf

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Dear members of the Planning Commission and Community Development Department staff:

Please enter the attached document:

> Ochoa, C.G.; Jarvis, W.T.; Hall, J.
> A Hydrogeologic Framework for Understanding Surface Water and Groundwater Interactions in a Watershed System in the Willamette Basin in Western Oregon, USA.
> *Geosciences* **2022**, *12*, 109. https://doi.org/10.3390/geosciences12030109

into the record for LU-24-027. This is a peer-reviewed, open-access article distributed under the terms and conditions of the Creative Commons Attribution license (https://creativecommons.org/licenses/by/4.0/).

The article contains scientific information that is highly relevant to understanding the flaws in the Applicant's claims that construction of the proposed new landfill will not impact groundwater access and groundwater quality on adjacent properties.

The article describes findings from a scientific investigation of groundwater and its relationship to surface water flow in the Oak Creek watershed, on the west side of Corvallis and also in Benton County.

The Oak Creek watershed is less than 10 miles south of the proposed new landfill, bordering on the Soap Creek watershed in its upper reaches. It is similar to the proposed development in terms of:

- bedrock geology,
- climate including seasonal precipitation,
- topographic relief,
- proximity to the Corvallis Fault.

I've cited this article in my own previously submitted testimony on groundwater issues. The second author, Dr. William (Todd) Jarvis, an Engineering Geologist registered in Oregon, kindly reviewed my testimony on that topic before I sent it to you, and confirmed that my representations of the technical findings in this article were accurate.

In particular, I cited this document as an example of how:

- Groundwater divides do not necessarily coincide with topographic divides (contrary to the applicant's claims).
- Groundwater directions in fractured bedrock (similar to that under Tampico Ridge) cannot be fully predicted from topography and shallow investigations.
- Bedrock structures such as the Corvallis Fault (which also angles down under the proposed development site) can influence groundwater flow directions.

This study also serves as an example of what the Applicant could have done, if they were genuinely interested to develop an understanding of groundwater in the area around the proposed new landfill. Such an investigation could provide a well-founded analysis of the risks for groundwater resources on adjacent properties.

But the Applicant has not undertaken such an investigation. Instead they're expecting you to accept arguments based almost entirely on the very limited data that they've collected on their own site. They know that this leaves major gaps, which they've tried to fill by conjecture and "arm-waving."

I hope that you will see that **arm-waving in the absence of relevant data is not an acceptable way to dismiss the legitimate concerns of nearby residents**, for whom loss of access of to safe and reliable well water would **seriously interfere with use of their property (Benton County Code 53.215)**.

Thank you for taking this information into consideration for your deliberations.

Yours sincerely,
Joel Geier, Ph.D.
38566 Hwy 99W
Corvallis, OR 97330-9320

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A Hydrogeologic Framework for Understanding Surface Water and Groundwater Interactions in a Watershed System in the Willamette Basin in Western Oregon, USA

Article *in* Geosciences · February 2022

DOI: 10.3390/geosciences12030109

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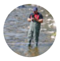

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Article

A Hydrogeologic Framework for Understanding Surface Water and Groundwater Interactions in a Watershed System in the Willamette Basin in Western Oregon, USA

Carlos G. Ochoa [1,*], William Todd Jarvis [2] and Jesse Hall [3]

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Abstract: A broad understanding of local geology and hydrologic processes is important for effective water resources management. The objectives of this project were to characterize the hydrogeologic framework of the Oak Creek Watershed (OCW) geographical area and examine the connections between surface water and groundwater at selected locations along the main stem of Oak Creek. The OCW area comprises the Siletz River Volcanic (SRV) Formation in the upper portion of the watershed and sedimentary rock formations in the valley. Past hydrologic and geologic studies and our field measurement data were synthesized to create a hydrogeologic framework of the watershed, including a geologic interpretation and a conceptual model of shallow, deep, and lateral groundwater flow throughout the OCW. The highly permeable geology of the SRV formation juxtaposed against the Willamette Basin's sedimentary geology creates areas of opposing groundwater flow characteristics (e.g., hydraulic conductivity) in the watershed. The Corvallis Fault is the primary interface between these two zones and generally acts as a hydraulic barrier, deflecting groundwater flow just upstream of the fault interface. The extreme angle of the Corvallis Fault and adjacent less permeable sedimentary geology might facilitate subsurface bulk water storage in selected locations. The stream-aquifer relationships investigated showed gaining conditions are prominent in the upper watershed's northern volcanic region and transition into neutral and losing conditions in the downstream southern sedimentary region in the valley. Agriculture irrigation seepage in the valley appeared to contribute to streamflow gaining conditions. Results from this case study contribute critical information toward enhancing understanding of local hydrogeologic features and potential for improved SW-GW resources management in areas near coastal ranges such as those found in the Pacific Northwest, USA.

Keywords: hydrology; geology; surface water-groundwater interactions; fault permeability; gaining stream; losing stream

Citation: Ochoa, C.G.; Jarvis, W.T.; Hall, J. A Hydrogeologic Framework for Understanding Surface Water and Groundwater Interactions in a Watershed System in the Willamette Basin in Western Oregon, USA. *Geosciences* **2022**, *12*, 109. https://doi.org/10.3390/geosciences12030109

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1. Introduction

Understanding the interactions between surface water and groundwater (SW-GW) is critical for improved water resources management [1,2]. Surface water and groundwater cannot be seen as isolated components [3] and the role of geology in their spatial and temporal variability throughout the landscape needs to be better understood. The replenishment of groundwater from surface water sources such as irrigation and precipitation can be influenced by geologic characteristics, such as the presence of fractured bedrock or basalt [4], and the timing and quantity of precipitation [5,6] or irrigation [7,8]. A common expression of SW-GW relationships occurs at the stream and aquifer intersection. Geology has long been considered a primary driver of watershed hydraulic characteristics [9–11] such as streambed permeability and the rate of surface water infiltration into the aquifer [12]. A variable SW-GW flow exchange can influence the streamflow volume between any two points along the stream and can create gaining conditions where groundwater inputs contribute to streamflow, or losing conditions where streamflow is lost through the permeable substrate [13–15]. Groundwater inputs can constitute a significant portion of the overall streamflow volume of a gaining reach or stream, especially during low-flow conditions [16]. The SW-GW connections are highly dependent on the geological features responsible for water transport and distribution throughout the landscape [4,17–19].

Geologic faults are another feature that significantly influences SW-GW relationships, particularly the dynamics of groundwater flow. Faulting impacts groundwater hydrology by altering flow vectors near the fault core zone [20]. Fault gouge, coupled with separation, creates a broad spectrum of hydraulic barriers, conduits, or a combination of these [4,17,20,21]. Impermeable areas within fault zones can sever the hydraulic continuity of aquifers, producing incongruent groundwater tables and isolated groundwater compartments. Active faults can have irregular fault core thickness and a degree of damage to surrounding rock along their profile length [21]. Varying degrees of damage and fault gouge create significant contrasts in permeability along with the fault profile, and as a result, a single fault can be both a barrier and a conduit for groundwater flow [21,22].

A common resource management integration of SW-GW relationships is by using groundwater replenishment tools such as managed aquifer recharge (MAR). In a 60-year retrospective of global MAR, Dillon et al. [23] indicated that "*MAR is a management tool to consider with and complement new efficiency measures in irrigation, switching to low water use crops, conjunctive use of surface water and groundwater resources including substituting use of recycled water for groundwater, and foregoing extraction*". The drivers for considering MAR in various forms include limited surface reservoir sites, increased costs associated with surface water treatment for harmful algal blooms, decreasing barriers to fish passage, and increased costs to purchase water from regional water suppliers. Over the past 50 years, the storage of managed aquifer recharge in the US has increased nearly 10-fold from approximately 300 $\text{Mm}^3\ \text{year}^{-1}$ in 1965 to 2570 $\text{Mm}^3\ \text{year}^{-1}$ in 2015 [23].

According to the Oregon Water Resources Department (OWRD), managed recharge and related aquifer storage and recovery (ASR) projects have been operating in Oregon since 1996, with more than 7.5 Mm^3 per year stored annually. Potential aquifer storage identified by Woody [24] approaches Oregon's annual public water supply in the year 2000, or 500 Mm^3. Oregon currently has approximately 15 state-authorized and planned ASR test sites servicing municipalities and agriculture. Most of these sites store water in volcanic rock aquifers, including the Siletz River Volcanics [24]. The use of small-scale managed recharge utilizing rainwater or spring water for domestic well storage is increasingly considered in mountainous areas underlain by low permeability volcanic rocks and siltstone [25,26]. Likewise, a market for aquifer storage space is gaining momentum as the concept of Aquifer Recharge Units (ARU) is being developed [27].

Characterizing hydrogeology's role in SW-GW interactions is critical for understanding the potential for integrated water resources management, including stream-aquifer interactions and groundwater recharge. This project examined hydrogeological features influencing SW-GW interactions in the southern portion of the Willamette Basin region in western Oregon, USA. The objectives of this case study were to: (1) characterize the hydrogeological framework of the watershed geographical area, and (2) examine stream-aquifer relationships along the main stem of the creek.

2. Materials and Methods

2.1. Study Site

The study was conducted in the Oak Creek Watershed (OCW) within the Willamette Basin region. The OCW (44.57° latitude; −123.30° longitude) encompasses 3360 hectares, and it is adjacent (west) to the City of Corvallis, Oregon, USA. The main stem of Oak Creek flows 11 km from its source and highest point (650 m above sea level, MASL) in the McDonald-Dunn Forest to its lowest point (64 MASL) at the confluence with Marys River. The climate in the region is a Mediterranean type, with a warm and dry season in the summer and a mild and wet winter season. Most precipitation occurs as rainfall between November and April. Mean annual precipitation within the basin ranges from 2500 mm at higher elevations to 1000 mm at lower elevations [28]. The monthly-averaged lowest temperature occurs in January (0.67 °C), while the highest occurs in August (27.4 °C). The lowest and highest total monthly precipitation occurring in July (9.1 mm) and December (181.4 mm), respectively [29]. Oak Creek is a fourth-order headwater stream with discharge volumes generally ranging between 0.01 $m^3 s^{-1}$ and 0.25 $m^3 s^{-1}$, and peak discharge reaching up to 6 $m^3 s^{-1}$ in the winter. Upland overstory vegetation at the OCW is dominated by Douglas-Fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), and western red cedar (*Thuja plicata*). At lower elevation sites, White oak (*Quercus alba*), red alder (*Alnus rubra*), Oregon ash (*Fraxinus latifolia*), and bigleaf maple (*Acer macrophyllum*) dominate the landscape.

Land cover in OCW encompasses forest (70%), agriculture (12%), urban (12%), and undeveloped (6%). Land use categories at OCW include forestry in the uplands, agriculture in mid to lower elevation sites, and urban including Oregon State University (OSU)'s main campus at the bottom of the watershed. Much of the area of the OCW is a mixture of publicly and privately-owned forest maintained for uses such as timber harvest, protected water drinking water-supply source areas, recreation, and forestry and agricultural research stations associated with the Colleges of Agricultural Sciences and College of Forestry at OSU. Approximately 20 hectares of OSU's pasture fields in the lower valley are irrigated with water diverted from Oak Creek during the summer. Water pumped from the creek is run through underground pipes that feed individual irrigation lines with multiple sprinkler pods at different fields. The rest of the agricultural activity in OCW depends mostly on the soil moisture accumulated during the wet precipitation season in the winter. No deep groundwater extraction for agriculture irrigation currently exists at the OCW. The Siletz River Volcanic (SRV) rock formation underlying much of OCW has become increasingly important for municipal and domestic water supplies in the last 30 years [30]. Over the past 20 years, significant residential growth near the cities of Corvallis and Philomath, Oregon, USA has been occurring [30,31].

The generalized hydrostratigraphy of the rocks underlying OCW has been summarized by [20,30,32,33] based on water well logs available from the OWRD, oil exploration wells available from the Oregon Department of Geology and Mineral Industries (ODGMI), field examination of rock quarries, outcrop examination, and hydrogeologic mapping (Table 1).

Table 1. Lithology, permeability, and hydrologic features of the various geologic formations at OCW.

Formation Name	Generalized Lithology	Permeability Architecture	Hydrologic Properties
Alluvium	Sand, gravel, cobbles thin, 10 m thick	Porous media.	Prolific yield for irrigation near Willamette River.
Spencer/Tyee/Yamhill, undifferentiated	Shale, siltstone, interbedded sandstone, up to 3000 m thick	Microfractures in siltstones and shale, porous media in sandstones.	Low yield for domestic wells. Some saline water and artesian flow in deep sandstones.
Siletz River Volcanics	Basalts with interbedded claystone and sandstone, up to 1000 m thick	Fractures, conduit flow in interflows. Stratigraphic and structural groundwater compartments common.	Moderate yield for domestic and municipal wells.

2.2. Hydrogeologic Framework

2.2.1. Site Geologic Conditions Map and Hydrogeologic Cross-Sections

Information from several sources including previous geophysical research conducted in the OCW [20], oil and gas exploration well logs maintained by the ODGMI, United Stated Geologic Survey (USGS) geological maps, OWRD water well logs, and our field-based data were used to create a Site Geologic Conditions Map along with cross-sections of site geologic and hydrogeologic conditions (Figure 1).

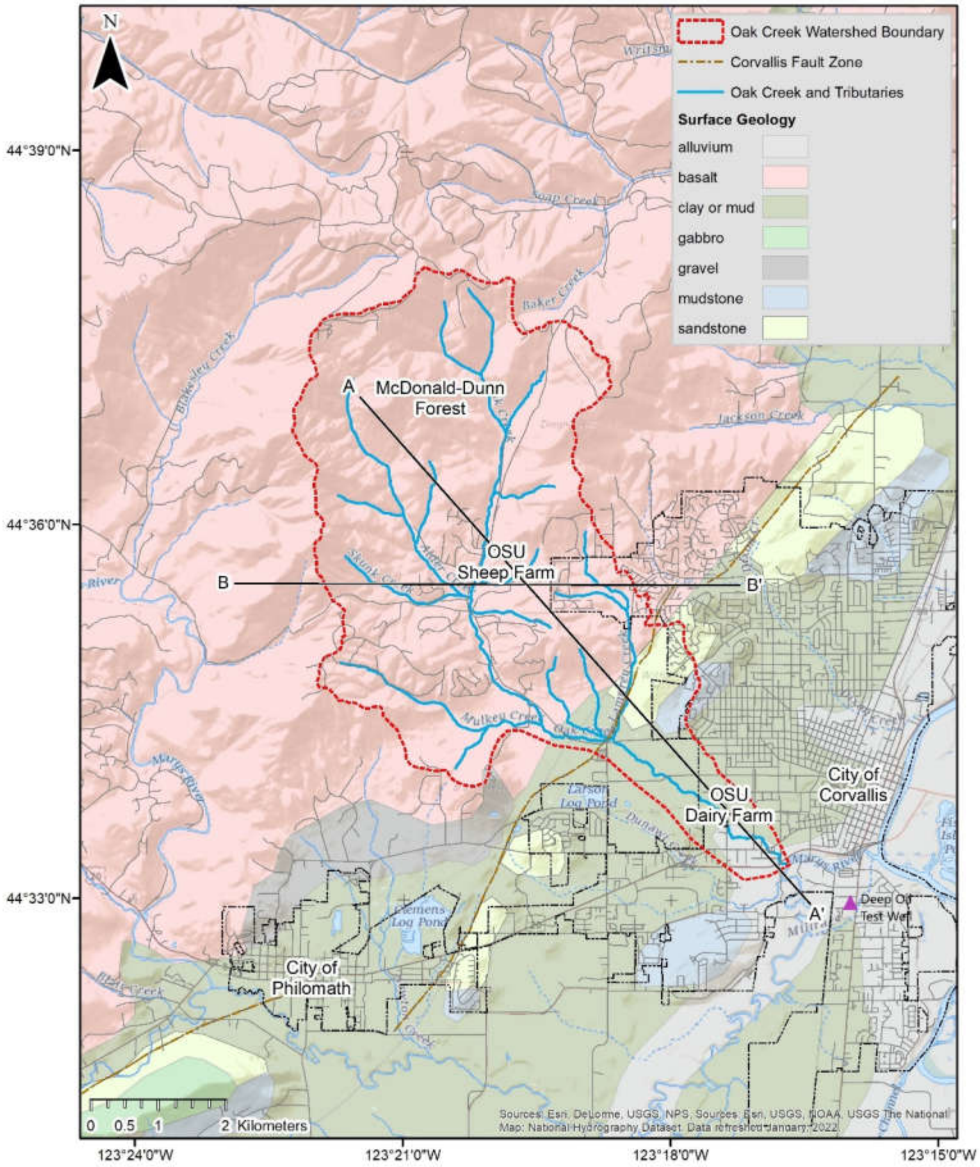


Figure 1. Surface geology map showing geologic cross-section locations and the main geologic features in the Oak Creek Watershed (OCW) and the surrounding region including Corvallis, Philomath, and Oregon State University (OSU).

Well logs of particular interest included those of a 655 m deep well completed in 1934 by the Willamette Petroleum Syndicate (WPS) located in south Corvallis near the outlet of the watershed (see Figure 1), and a 2581 m deep well completed in 1964 by the Gulf Oil Corporation of California (GOC) located approximately 18 km to the southeast near

Halsey, Oregon, USA (location not shown on Figure 1). The GOC well penetrates surficial alluvial deposits of soil, clay, sand, and gravel before encountering the Eugene Formation, the Spencer Formation, the Yamhill Formation, the Tyee Formation, and finally, the SRV Formation at 2160 m below ground surface. The WPS well records are relatively complete to a depth of 655 m, other than a missing section between 412 and 475 m depth, which was likely omitted due to a high-pressure water flow event occurring when the borehole was deepened with a cable tool. The well log annotation substantiated that the drillers required a water shut-off valve when they reached approximately 450 m. This interval is likely a confined aquifer and is stratigraphically located in a volcano-clastic conglomerate mentioned in the GOC well log. If the missing zone and the volcano-clastic zone are stratigraphically equivalent, this zone's apparent thickness likely decreases from nearly 450 to 60 m thick from east to west.

Seismic reflection data and deep exploratory well logs throughout the Willamette Valley reveal the bottom SRV unit extends from beneath the continental shelf in the west [34,35] to as far east as the Cascade Range [20]. Geophysical evidence shows the thickness of the SRV formation to be approximately 8 to 18 km [36–38], with about 4.5 km exposed west of the Corvallis Fault [20].

The Corvallis Fault, which bisects the watershed, is a low-angle reverse-thrust fault [20] uplifting the watershed's northern volcanic region over the southern sedimentary region and pushing it to the southwest. The position of the fault creates two distinct geologic areas with opposing hydrogeologic conditions. Portions of the study site north of the Corvallis Fault overlie the SRV Formation, which consists of a mélange of permeable Paleocene to Eocene submarine basaltic pillow lavas interbedded with less permeable tuffaceous marine sediments which dip approximately 20 degrees to the west. The fault-cutting alignment perpendicular across the watershed creates an abrupt interface between the northern high conductivity volcanic geology and the southern low conductivity sedimentary geology. Additionally, numerous dikes of gabbro, diorite, and aplite rock cut through the fault zone and act as hydraulic barriers, completely severing the hydraulic connection at the fault interface and redirecting groundwater in these areas [20]. The low angle of the Corvallis Fault creates a wedge of Siletz Formation volcanic rocks overlying Spencer and Tyee formation sedimentary rocks [20] which compartmentalizes vertical groundwater movement and promotes horizontal groundwater movement parallel to the fault line.

South of the Corvallis Fault, geology abruptly transitions into thick repetitious sequences of bedded units of graded sandstones, siltstones, and shales [39]. Fractured marine basalt found north of the Corvallis Fault interface is highly permeable due to large conduits (fractures) cutting through the formation; however, the rock itself is nearly nonporous. Conversely, the sedimentary formations found south of the fault are more porous but less permeable.

These two regions' opposing hydrological characteristics influence where and how groundwater flows and how late into the dry summer season water remains in these formations to supply surface water streams (baseflow). The Tyee and Spencer formations, sandstones and siltstones with high porosity (14% to 49%) but low permeability (1×10^{-11} to 6×10^{-6} m s^{-1}), can sustain baseflow into surface water streams later into the dry season when compared to the less porous (3% to 35%) and more permeable (4×10^{-7} to 2×10^{-2} m s^{-1}) basalts of the SRV Formation [40,41].

To better understand the geologic structure of the OCW, we constructed two hydrogeologic cross-sections (A-A′ and B-B′), the locations of which are shown on the surface geology map (see Figure 1). Cross-section A-A′ runs from the McDonald-Dunn Forest in the northern region of the watershed in a southeast direction through Oak Creek and the Corvallis Fault. Cross-section B-B′ runs from west to east, approximately perpendicular to Oak Creek at the OSU Sheep Center.

Subsurface geology and static water level information represented on the geologic cross-sections were obtained through the analysis of over 600 OWRD well logs, two deep exploration drill logs (WPS and GOC), geology maps, previous research reports [20,30,32,42], and our stream and shallow groundwater well level information. Geologic and hydrologic data of wells within 250 m of each side of the cross-section line were projected onto the cross-section transect to create a vertical hydrogeologic profile (n = 620). Additionally, well log stratigraphy from the nearby WPS and GOC deep exploration wells were projected to the cross-section profile (n = 2). Together, the cross-sections provide a three-dimensional representation of the hydrogeologic framework of the OCW site and an indication of how the Corvallis Fault affects OCW groundwater dynamics.

2.2.2. Stream-Aquifer Interactions

We used shallow groundwater and stream levels data to assess stream-aquifer interactions at one site (OSU Sheep Center; 118 MASL) in the northern volcanic portion of the watershed and one site (OSU Dairy Center; 80 MASL) in the southern basin-sediment infills. Data from previously installed monitoring wells (<6 m depth; 50 mm diameter) were used to characterize water table fluctuations at these two sites [33]. Two wells (sw-1 and sw-2) were in the riparian area at the OSU Sheep Center, and three wells (dw-1 to dw-3) were located in the riparian zone at the OSU Dairy Center. A laser level was used to measure onsite soil surface, stream, and water table elevation to develop cross-section profiles at each location. Data from two other wells in a 2-ha irrigated pasture grass field and one well in its adjacent riparian area, near the stream gauge, were used to characterize irrigation percolation contributions to the shallow aquifer and potential return flow to the stream. All wells were equipped with water level loggers (Model HOBO U20-001-01, Onset Computer, Corp., Bourne, MA, USA) and programmed to record data every hour. A water level meter (Model 101, Solinist Canada Ltd., Georgetown, ON, Canada) was used to collect depth to water table during selected dates. These data were used for verification or calibration of the water level loggers. A water level logger (Model HOBO MX2001, Onset Computer, Corp., Bourne, MA, USA) was installed in each location (OSU Sheep Center and OSU Dairy Center) to measure stream water level fluctuations.

In addition to the monitoring of stream and groundwater levels to assess stream-aquifer interactions, we also monitored hydrostatic pressure differential at selected locations in the streambed during summer groundwater baseflow conditions. We built a hydraulic potentiometric manometer (potentiomanometer) [33] to collect hydrostatic pressure data from 20 different locations along the creek, from its headwaters to its mid and lower reaches. Three measurements at each location were taken at 1 m depth in areas where streambed conditions permitted an adequate seal between the potentiomanometer and surrounding sediment. Locations with large rocks, highly compacted sediments, or sand and gravel bottoms did not provide adequate conditions for manometer readings. A ‘vertical gradient’ (positive or negative) was obtained for each location based on the potentiomanometer readings. Hydrostatic pressure values of zero or between −1 and 1 mm were considered to be neutral [33].

2.2.3. Potentiometric Surface Map

We developed a potentiometric surface map based on a previous study on springs and losing reaches of the upper watershed [30], static water level data obtained from OWRD well logs, surface water levels of surrounding rivers and streams, and data obtained from our observation wells. Potentiometric contour lines were created by first generating an Excel® spreadsheet detailing the position of wells, surface water observation locations, and the respective static water level of our observation wells at each location. Static water level (meters below ground surface) was then subtracted from the ground surface elevation to obtain hydrostatic head in meters at each location. The OWRD water level measurements span approximately 65 years and at different times of the year, as summarized in their well-log database. We acknowledge there could be much uncertainty; consequently, all the wells’ data are referenced as a generalized summary of water level measurements. Long-term data from observation wells at or near OCW are scarce.

Data from a well near the OCW showed seasonal static water level variations of about 4 to 25 m during the period or record 1994 to 2014 (Figure A1). This is within the contour interval of 25 m we used in generating the potentiometric surface map.

Given static water levels in individual wells represent a snapshot in time, synoptic contour lines were generated using the spatial analyst extension of ArcMap® (Version: 10.4.1, ESRI, Redland, CA, USA) to create a digital elevation model (DEM) of the water table surrounding the OCW. Spatial analyst does not consider the influence of the Corvallis Fault when using water elevation data to generate the DEM. Thus, specific contour lines were manually edited to show the elevation of groundwater more accurately as it crosses the geologic interface at the Corvallis Fault. These maps were compared to and shown to be consistent with past hydrologic research in the area (see for example [32]).

Potential lines were edited where there were conflicting or limited data. For example, the potential lines were not shown in areas with a limited potentiometric low associated with pumping from clusters of domestic wells. Likewise, the potential lines were adjusted where our field observations of the potentiomanometer readings suggested gaining or losing reaches that conflicted with direct field observations such as a spring occurrence or detailed water level measurements in dedicated observation wells.

3. Results

3.1. Hydrogeologic Framework

3.1.1. Hydrogeologic Cross-Sections

Figure 2a,b shows the cross-sectional representation of the three distinct geologic formations found at the OCW study site: the fractured and highly permeable SRV Formation to the north of the Corvallis Fault, the low permeability Tyee and Spencer Formations to the south of the fault, and the fault zone itself. The water table in the sedimentary basin region is laterally continuous compared to stratigraphically compartmentalized water-bearing zones found in the Siletz Formation [30]. Depths to water in the OCW site ranged from nearly ground-surface (<1 to 3 m) in the upper region of the watershed near the McDonald-Dunn Forest to nearly 100 m below ground surface in the central portion of the watershed near the OSU Sheep Center (see Figure 1).

Figure 2a (cross-section A-A′) shows the multilayered aquifer system of the SRV formation, as well as the shallow and deep aquifer zones of the basin region east of the fault. Data from the WPS and GOC deep exploration wells showed that the water table in the basin east of the fault was generally within 3 to 5 m below the ground surface, with slight fluctuation throughout the year. The GOC drill logs showed a 400+ m thick layer of a water-bearing zone of volcano-clastic conglomerate extending as far west as the Corvallis Fault. Dip estimations suggest the volcano-clastic layer shallows to the west from the GOC well and could be located 10 to 60 m below the OSU Horse Center (Figure 2a). Previous field observations by Goldfinger [20] near the City of Philomath, OR, noted this volcano-clastic layer outcropping in a gravel pit within Spencer Formation siltstones on the Corvallis Fault interface. The volcano-clastic layer at the Philomath Gravel Pit supports the notion that this layer is contiguous east to the GOC exploration well location, dipping between 1 and 3 degrees to the north striking to the southwest.

Figure 2b (cross-section B-B′) illustrates the Siletz River Volcanics region of the watershed. Data from nearby well logs showed unconfined, confined, perched, and compartmentalized aquifer systems within the SRV formation near the OSU Sheep Farm. Depth to water table data was more erratic as the distance from Oak Creek increased, with static water levels adjacent to the creek found near the ground surface. Static water levels decreased in elevation as the distance from the creek increased. Additionally, artesian wells near the west end of the profile (data not shown) indicated confined aquifers within the SRVs near the OSU Sheep Center and McDonald-Dunn Forest. During the dry season, the central water-bearing unit in this region appeared to be individual isolated pockets of sandstone within the SRVs.

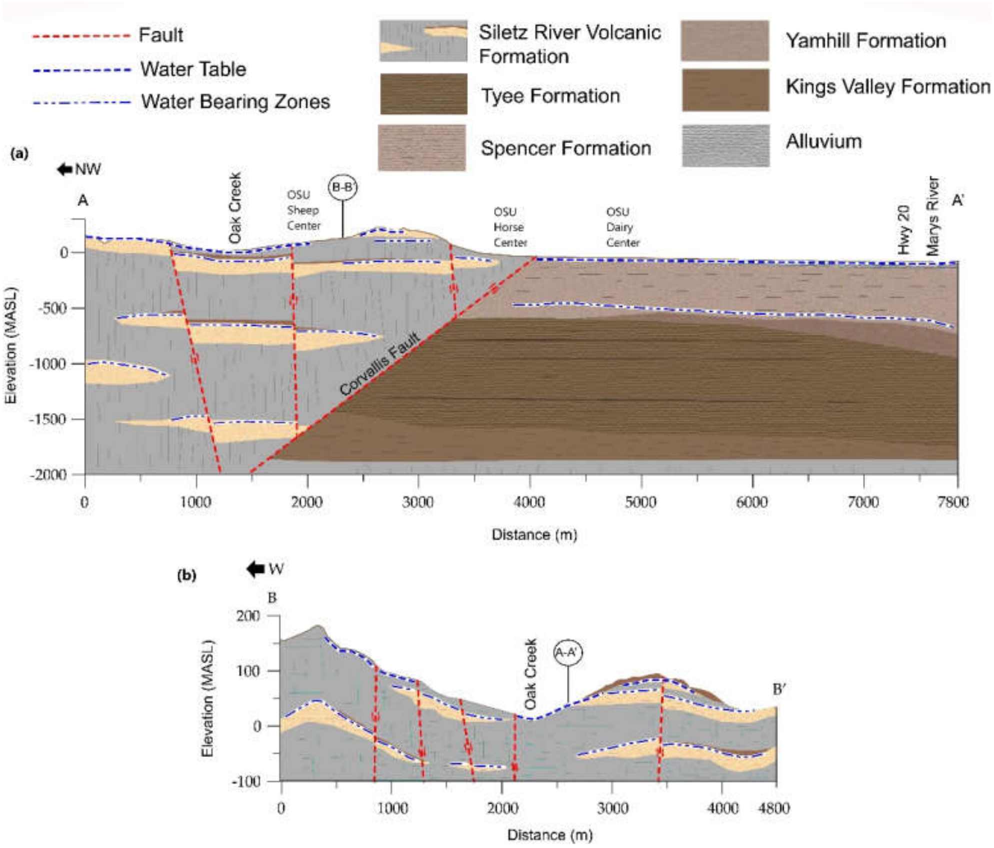


Figure 2. Geologic cross-sections of the Oak Creek Watershed. (**a**) Cross-section A-A′ (drawn to scale) runs from the headwaters of Oak Creek in McDonald-Dunn Forest, southeast to the confluence with Marys River. (**b**) Cross-section B-B′ (drawn 5× vertical exaggeration) runs from the headwaters of Oak Creek in the McDonald-Dunn Forest southeast to its confluence with Marys River.

3.1.2. Stream-Aquifer Relationships

Figure 3 shows three years (2019–2021) of seasonal water level fluctuations in the stream, the monitoring well in the riparian zone, and a well in a non-irrigated crop field at the OSU Dairy Center, in the sedimentary rock formations (see Figure 1). The well in the riparian area is 6 m north of the stream, the well in the irrigated field is 160 m north of the stream, and the well in the non-irrigated agricultural field is 90 m south of the stream. Relatively rapid rises and declines in stream level in response to rainfall events during winter and spring were observed. Groundwater levels in all wells began rising with the onset of winter precipitation, which reached its peak in January. Following the end of the rain season in the spring, groundwater levels started a steady decline, which in the non-irrigated field continued until baseflow conditions were reached in the fall. In the irrigated field, groundwater levels started rising soon after the onset of the irrigation season (July to September) and remained relatively high during the summer and fall. A slight groundwater level rise during the irrigation season was also observed in the riparian area well, located 6 m north of Oak Creek and 40 m south of the irrigated field's edge. Marginal to no summer precipitation was observed during the three years evaluated; therefore, the groundwater level rises observed in the wells in the irrigated field and riparian area were attributed to irrigation seepage and return flow to the stream, respectively. All irrigation applications were from water diverted from Oak Creek. Total water applied (irrigation depth) for the pasture field depicted in Figure 3 was 510 mm in 2020 and 670 mm in 2021. No records are available for 2019. Annual total precipitation was 871, 1004, and 1043 mm in 2019, 2020, and 2021, respectively.

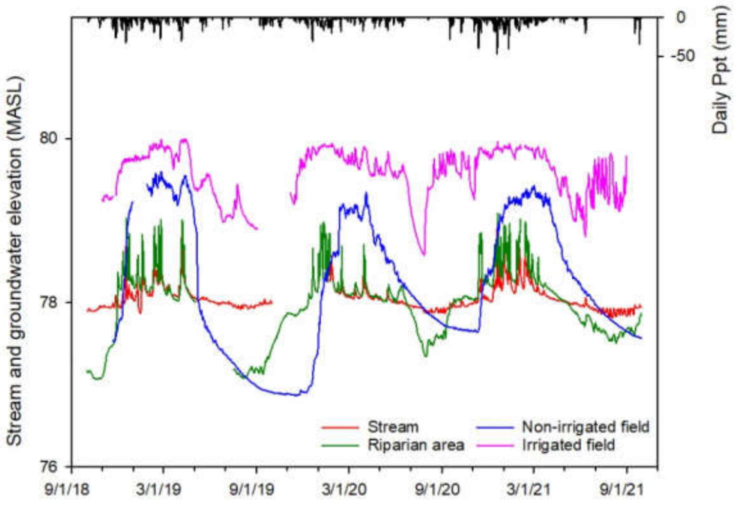


Figure 3. Seasonal surface water and shallow groundwater response to precipitation (Ppt) in three water years (2019 to 2021) at in-stream and various monitoring well locations at variable distance from Oak Creek's mainstem near the OSU Dairy Center: a well in the riparian area 6 m north of the stream, a well in an irrigated field 160 m north of the stream, and a well in a non-irrigated agricultural field 90 m south of the stream. Irrigation season is typically July to September every year. Daily Ppt records obtained from https://agsci.oregonstate.edu/hyslop-weather-station (accessed on 30 January 2022).

A closer look at stream and groundwater levels variability in the wells in irrigated, non-irrigated, and riparian areas during the 2020 irrigation season (27 July to 12 September) is shown in Figure 4. A downward groundwater level trend, typical of the transition to baseflow conditions in the summer, can be observed in all wells before the start of irrigation. While a steady decline in groundwater level occurred in the non-irrigated field during the rest of the season, the well in the irrigated field first, followed by the well in the riparian area, showed a rise in groundwater level of up to 1.18 m a few days after irrigation started. Several peak and slow decline events observed in the irrigated field well corresponded to irrigation applications. Groundwater levels for both the riparian and irrigated field wells continued higher than the non-irrigated field during the remainder of the irrigation season.

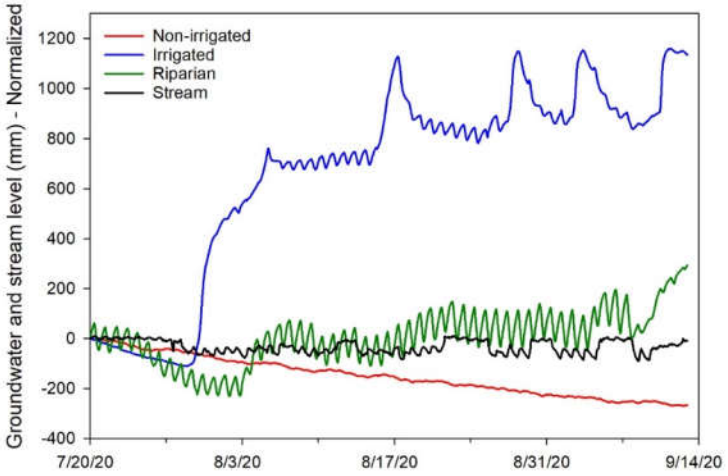


Figure 4. Stream and shallow groundwater level variability in wells at riparian, irrigated, and non-irrigated field locations during the 2020 irrigation season (27 July to 12 September).

Variable seasonal stream-aquifer interactions along non-irrigated fields in both the SRV and Sediments formations were observed. Figure 5a shows groundwater and Oak Creek levels at an upstream cross-section at the OSU Sheep Center, in the SRV rocks-dominated area of the OCW in 2021 (See Figure 1). The steep gradient between groundwater and stream levels for peak flow in the winter and baseflow in the summer indicated 'gaining' streamflow conditions. Figure 5b shows groundwater and Oak Creek levels at a downstream cross-section at the OSU Dairy Center, in the Sediments portion of the OCW. At this reach, groundwater and stream levels showed 'losing' streamflow conditions during the summer (baseflow) and 'gaining conditions' during the winter precipitation season.

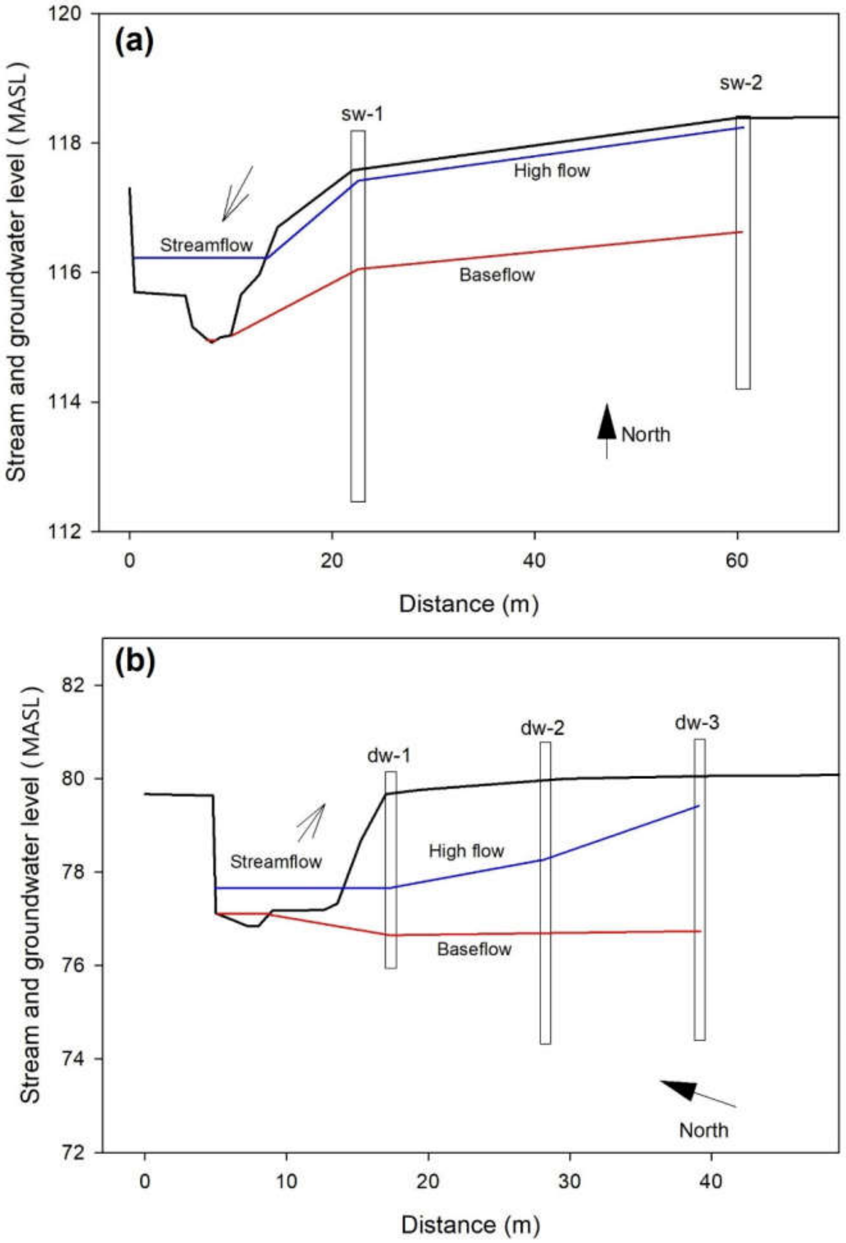


Figure 5. Surface water and groundwater levels (MASL) at baseflow (summer) and high streamflow (winter) conditions in (**a**) the Siletz River Volcanics formation at the OSU Sheep Center and (**b**) the alluvium in the Spencer formation at the OSU Dairy Center.

Data from the potentiomanometer test conducted under baseflow conditions in the summer showed that the Oak Creek streambed is losing water to the aquifer at locations upstream from the confluence with Alder Creek (see Figure 1), in the permeable fractured volcanic geology, as indicated by the negative pressure values obtained. The negative vertical hydraulic gradient gradually decreased until it reversed at approximately 3 km upstream of the Corvallis Fault, where multiple readings showed a positive slope immediately upstream of the fault. Once Oak Creek crossed the Corvallis Fault, hydraulic gradients became much less variable and maintained neutral or positive pressure conditions, as reported in [33].

3.1.3. Potentiometric Surface Map

The potentiometric surface map (Figure 6) generated from shallow static water level and surface water level data showed that the region's water table generally followed surface topography across the study area. Most of the groundwater within the OCW originated in higher elevations in the McDonald-Dunn Forest and flowed down-gradient to the south towards Oak Creek and Marys River's confluence. Water table elevation levels are 1 to 3 m higher on the northern side of the fault line and then drop significantly immediately past the fault. Once past the fault plane, water table elevations and gradients had much less variance than those of the volcanic region. In the southern region, surface topography affected water table elevations in places, but groundwater generally flows at a consistent gradient toward either the Marys River or the Willamette River.

The watershed's northern volcanic portion was characterized by downward vertical gradients that flowed from several aquicludes and aquitards, resulting in a multilayered aquifer system (see Figure 2a: Cross-Section A-A'). Water table levels in the southern sedimentary region of the watershed generally remained much more consistent, only fluctuating between 1 to 5 m throughout the year.

The Corvallis Fault lies at the boundary between the Siletz Formation and the Tyee and Spencer Formations and acts as the interface between these two opposing hydrologic zones. Potentiometric contour lines generated from OWRD well log data indicated that the fault zone acts as both a barrier and a conduit for groundwater flow through the fault interface. Impermeable zones within the fault deflect groundwater, while large fractures and truncations allow groundwater to pass. Additionally, the abrupt change in permeability between the two formations impacts groundwater paths and velocities, as fluid transfer through geologic material is much more efficient in the watershed's northern volcanic region.

Hydraulic gradients within the OCW generally mimic the topography, with groundwater circulating from higher elevations towards lower elevations in the creek channel or to the south in the basin. This is generally true down to the Corvallis Fault, where the fault's impermeable nature causes groundwater to flow parallel to the fault plane until it either finds a conduit or flows through the terminus of the fault damage zone.

Potentiometric contour lines adjacent to the fault have been altered by the change in permeability architectures and the less-permeable fault zone, which has caused them to become fragmented in places. Additionally, water table elevations in the Oak Creek basin within 1 to 2 km northwest of the fault-line have been elevated, as indicated by the area's diminutive groundwater flow gradient.

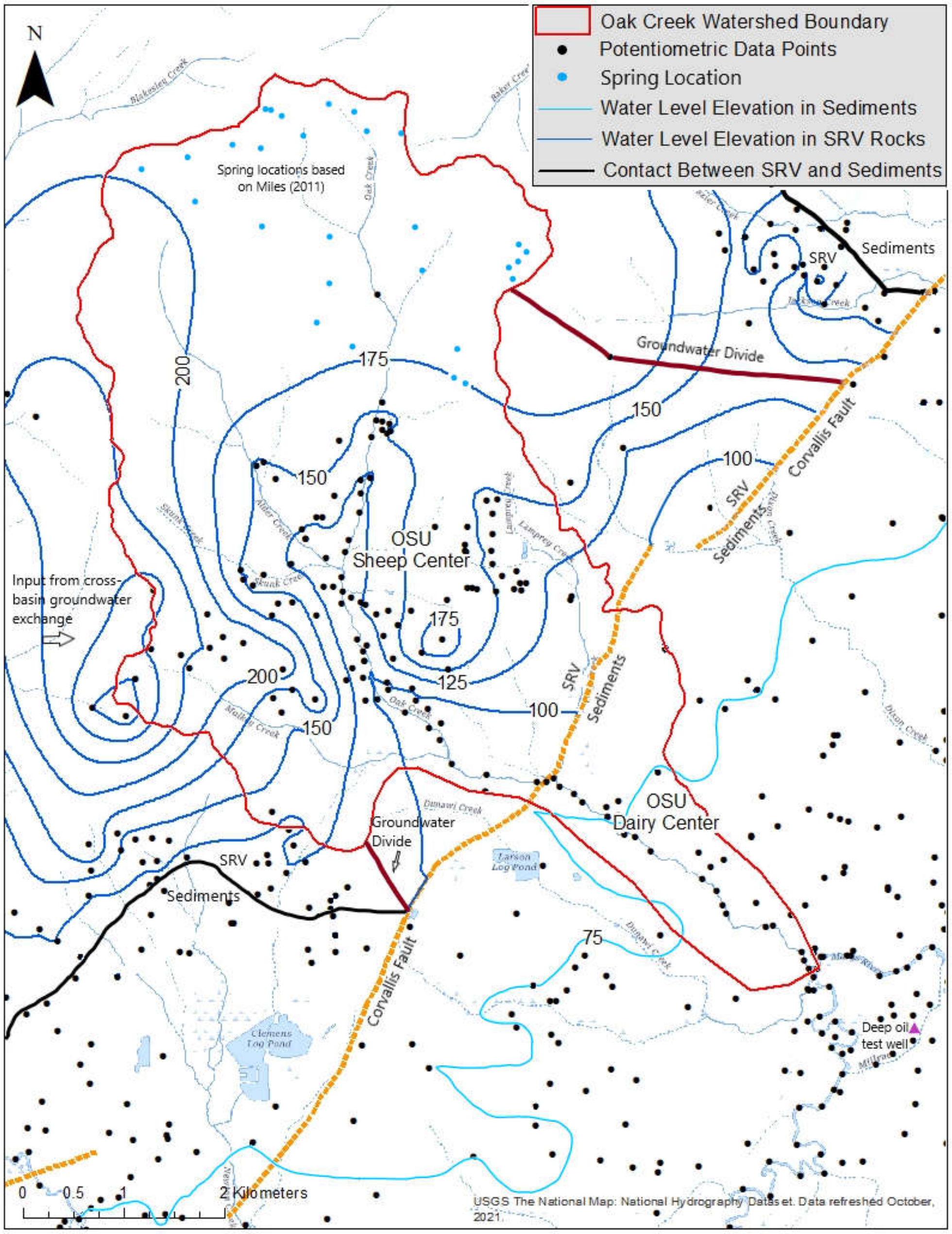
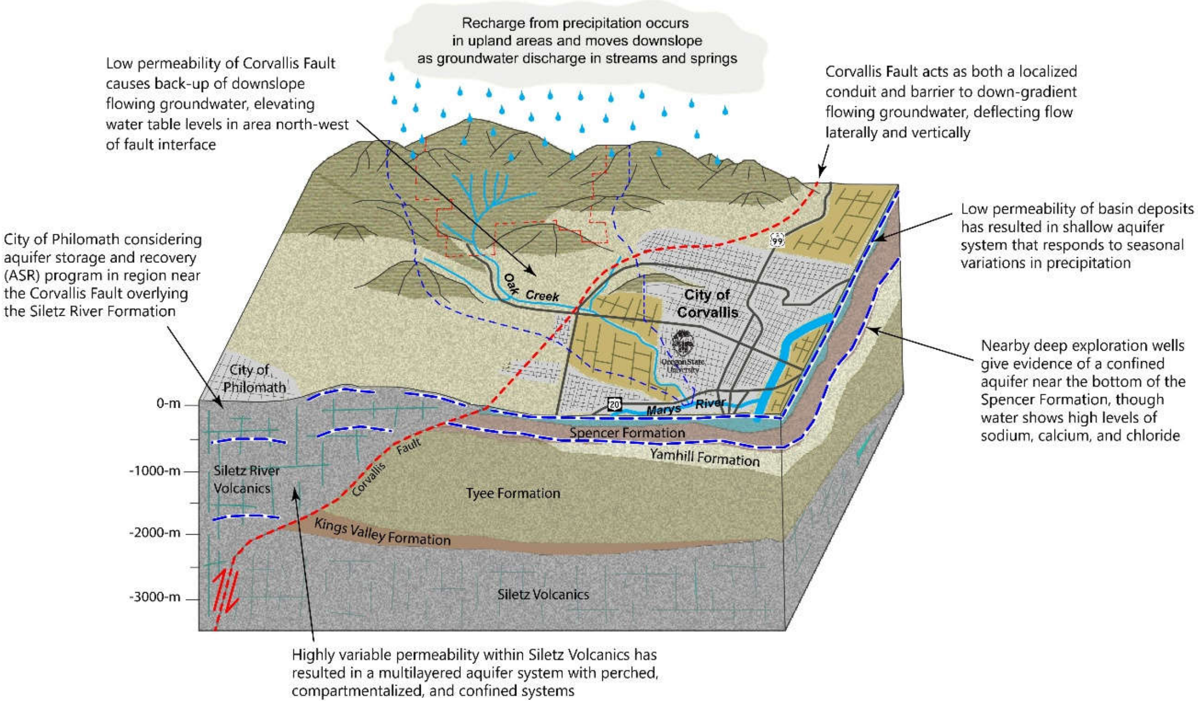


Figure 6. Potentiometric surface map of the Oak Creek Watershed and the surrounding region.

3.1.4. Conceptual Model

A conceptual (block) model, including the neighboring hydrogeologic conditions, was created to gain a three-dimensional representation of the OCW and surrounding region's hydrogeologic framework. The model developed is a synthesis of available topographic, geologic, and hydrologic data. Location, depth, and representation of geologic formations and structures were based on geologic descriptions from well logs and maps described by Goldfinger [20]. The nearby WPS and GOC deep exploration wells drilled southeast of the OCW provided data on subsurface geology structure, including lithology, potential water-bearing zones, dip and strike of geological layers, and water quality. The dip angle and distance from the exploration wells to the study area were considered when interpreting hydrogeological structure and conditions.

The Oak Creek Watershed conceptual model (Figure 7) shows that watershed hydrologic characteristics are dominated by surface topography and geology. The highly contrasting permeability architecture of the opposing geologic formations and the fault between the northern volcanic and the southern sedimentary regions influence groundwater flow through the fault interface. This groundwater compartmentalization model is consistent with other studies completed in fractured volcanic aquifers in Oregon [4,43]. Past drilling and other research data [20,30] in the OCW have indicated numerous intrusions of very low permeability volcanics throughout the region, especially along faults. The contrasting permeabilities of the northern and southern regions paired with impermeable intrusives along the fault interface have produced a zone of decreased permeability along the fault plane. As groundwater circulates hydraulically down-gradient and encounters this zone, velocities decrease, and flow is deflected along the fault strike. Groundwater gradients within the two regions of the study site are highly contrasting, with higher grades in the northern volcanic region compared to a lower grade in the southern sedimentary region. The fracture permeability, coupled with the higher surface slope in the Siletz Formation, results in a discontinuous water table with numerous perched and confined systems. In contrast, low permeability, and low gradients in the Tyee and Spencer formations of the southern region create a shallow water table circulating towards the Willamette River, which serves as a regional hydraulic sink (Figure 7).


Recharge from precipitation occurs in upland areas and moves downslope as groundwater discharge in streams and springs

Low permeability of Corvallis Fault causes back-up of downslope flowing groundwater, elevating water table levels in area north-west of fault interface

Corvallis Fault acts as both a localized conduit and barrier to down-gradient flowing groundwater, deflecting flow laterally and vertically

City of Philomath considering aquifer storage and recovery (ASR) program in region near the Corvallis Fault overlying the Siletz River Formation

Low permeability of basin deposits has resulted in shallow aquifer system that responds to seasonal variations in precipitation

Nearby deep exploration wells give evidence of a confined aquifer near the bottom of the Spencer Formation, though water shows high levels of sodium, calcium, and chloride

Highly variable permeability within Siletz Volcanics has resulted in a multilayered aquifer system with perched, compartmentalized, and confined systems

Figure 7. Conceptual model of the groundwater system of the Oak Creek Watershed and surrounding region based on a synthesis of topographic features and data from hydrologic and geologic investigations.

4. Discussion

4.1. Interaction of Geology and Hydrology

Previous sections discussed the geology and hydrology using a synthesis approach building upon rather than duplicating previous studies. Most of the physical hydrology studies summarized in this study focused on low flow conditions, given this is a time of year with competition for water use between agricultural facilities, fisheries, and rural residential land use rather than exploring the interaction of geology, direct runoff, and baseflow. Seasonal SW-GW connections are variable and can be contrasting within short distances. Gaining conditions are prominent in the watershed's northern volcanic region and transition into neutral and losing conditions in the southern sedimentary region. As depicted in Figure 8, the upper part of the watershed is a springs basin where the SRV is composed of low permeability rock mixed with higher permeability breccia and interflow zones, similar to conditions commonly found in rock quarries. Based on areal analysis, the rocks and interflow zones dip westward due to uplift along the Corvallis Fault. Losing reaches of the incised streams are more common where the topography becomes less steep. Whether the noted stream losses can be directly attributed to a function of changes in permeability architecture is uncertain and its study goes beyond the purpose of this synthesis paper.

The middle section of the watershed constitutes a generally gaining reach of the mainstem of Oak Creek. Drainage from the overlying perched aquifer to springs discharging along the stream bank can be observed downgradient of the OSU animal research facilities. Ground-truthing through observations of water levels in shallow wells confirmed the hydraulic separation of the perched aquifer from the deeper volcanic rock aquifer. Stable isotope analyses of stream water, coupled with general water chemistry analyses, indicate mixing of surface water and groundwater in the OCW exists [44]. Reconnaissance-level stream gauging using a potentiomanometer identified isolated losing reaches [33]. Whether the losing reaches are a function of an apparent increase in intercalated low permeability sedimentary rocks or volcanic rock permeability architecture or stream capture of clusters of permit-exempt wells servicing the rural residences remains unknown without further investigations.

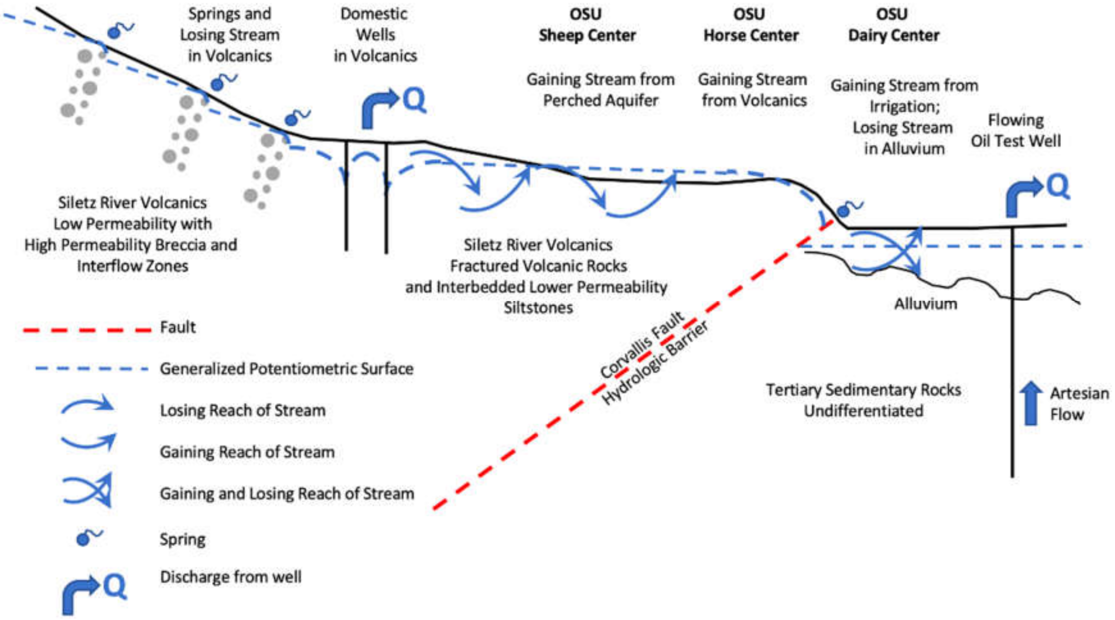


Figure 8. Summary sketch of hydrogeologic observations and interpretations. Not to scale. Generalized orientation of the section coincides with section A-A′ on Figure 1.

Study results showed that the hydrogeology of the OCW is characterized by two regions of contrasting hydrologic properties: the highly fractured and permeable Siltez Volcanic rocks in the northern region of the watershed and the low permeability sediments of the Spencer, Yamhill, and Tyee formations in the southern region. The interface between these two regions is the Corvallis Fault that serves as a hydraulic barrier to groundwater flowing downgradient from the Coastal Range to the Marys River basin to the south and the Willamette River to the east. The Corvallis Fault severs the hydraulic continuity of the volcanic rock aquifer by juxtaposing the lower permeability sedimentary rocks to the east against the higher permeability rocks on the hanging wall of the reverse fault. The hydrologic barrier interpretation of the Corvallis Fault is corroborated by previous work that mapped the Benton County area's well function, including wells located within the OCW and SRV, Tyee, and Spencer Formations. Transmissivity, specific capacity, and total yield all increased in the Oak Creek Basin area near the Corvallis Fault [30]. These findings are consistent with the hydrogeologic framework presented in this article. As groundwater encounters the less-permeable interface of the Corvallis Fault, it backs up, increasing the amount of groundwater available for withdrawal. This results in groundwater mounding in the SRV north of the fault that contributes to the compartmentalization of the volcanic rock aquifer.

Higher variability of groundwater level exists within the Siletz Formation compared to the Tyee and Spencer formations. The potentiometric surface map developed in this study provides evidence of the northern and southern regions' opposing permeability architectures and hydraulic conductivities. Water contours in the northern region generally have a higher gradient than those in the southern region due to the volcanic rock's greater effective porosity (permeability) and hydraulic conductivity. Water moves more freely through the northern portion of the watershed, and as a result, the hydraulic gradient is steeper in the northern region compared to the southern region. The potentiometric surface map developed in this project is consistent with a groundwater map completed in the Corvallis area that stopped just east of the OCW [32]. While no wells penetrate the Corvallis Fault to confirm the hydraulic separation from west to east, the decrease in hydraulic gradient in wells tapping the sedimentary rocks to the east, coupled with the reported artesian flow from a nearby oil and gas exploration well provides first order approximations of the lack of direct hydraulic continuity across the Corvallis Fault. Irrigation of fields along the lower reach of the OCW also complicate the confident assessment of gaining and losing reaches as some of the return flows from irrigation occur along the north bank of the stream. Yet, water levels measured in shallow wells located on the south bank of the stream are unchanged during the irrigation season.

4.2. Groundwater Compartmentalization

The Corvallis Fault, coupled with complex inter-layering of sedimentary rocks and volcanic rocks and groundwater mounding, has created a groundwater compartment in the OCW. Groundwater compartments have demonstrated the suitability of Columbia River Basalts for the storage of water and gas. Mansfield [45] described a compartmentalized aquifer in northwestern Oregon for aquifer storage and recovery as an affordable alternative to above-ground storage. The Mist Gas Field, located along the Oregon coast near Astoria, Oregon, has been used since the 1980s to store natural gas in compartmentalized volcanic rocks similar to the compartment described by Mansfield, precluding the need for expensive above-ground storage facilities [46]. Harpham [43] designed an Aquifer Storage and Recovery project to repurpose a dam site in the Deschutes Basin near Tumalo, Oregon, USA where a surface water reservoir constructed on the footwall of a fault in the volcanic rocks has never held water since construction in 1915. Ringrose and Meckel [47] cite the role of fault architecture as a critical element for storage site characterization elsewhere in the world, citing a global petroleum assessment where 71% of the known hydrocarbon reserves occurred in structural (i.e., faulted) traps, as opposed to stratigraphic or other traps.

Beyond the trace of the Corvallis Fault, the outline of the OCW groundwater compartment closely coincides with watershed boundary. As depicted in Figure 6, the northern boundary is defined hydraulically by the groundwater mounding or "divide" just beyond the watershed boundary. Likewise, the aquifer compartment's southern boundary closely coincides with the watershed boundary, which is hydraulically defined by groundwater mounding or divide.

4.3. Land and Watershed Management Practices

An isotope and geochemical investigation by Johnson [44] identified a connection between agricultural land use in the middle reaches of the OCW. The preponderance of forest and agricultural use within the OCW, coupled with the related rural residences served by individual permit-exempt water wells, underscores the importance of protecting drinking water supplies, especially given that permit-exempt water wells are typically paired with onsite wastewater systems that are well known for contributing nitrates and phosphorus to groundwater [48]. While the Agricultural Improvement Act (Farm Bill) of 2018 primarily focuses on building partnerships between water utilities and agricultural producers in watersheds to limit nutrient and sediment runoff, the Conservation Partnership Programs rarely address groundwater and much less groundwater supplying drinking water derived from private wells [49].

This paradox is not unique to the Farm Bill of 2018. The Wellhead Protection and Source Water Protection Programs associated with the Safe Drinking Water Act, as amended in 1986, served as the impetus behind states to develop programs to protect groundwater supplying public water systems. However, the Wellhead Protection and Source Water Protection Programs continued to overlook protection for areas serviced by private wells. The Sole Source Aquifer program authorized by the Safe Drinking Water Act of 1974 allowed citizens to petition the Environmental Protection Agency for Sole Source Aquifer Designation when an aquifer supplies at least 50% of the drinking water for a “service area” and if there are no reasonably available alternative drinking water sources should the aquifer become contaminated. This study provides an example of the type of information needed to develop a Source Water Protection program in mixed agriculture, forestry, and rural residential land use through the Sole Source Aquifer petition process.

4.4. Framework Adds to Role of Geology and Potential Groundwater Supplies

One of the ancillary goals of the study was to evaluate the substitution for surface water supplies used by research farms that require temporary dams on Oak Creek which inhibit fish passage. The OCW study offers an unusual alignment of supplemental water needs, hydrogeologic framework that contributes to water supply options, current favorable land use for water quality protection, and aquifer compartmentalization for subsurface storage. The nearby communities of Philomath and Dallas, Oregon, USA are experiencing growth and desire to develop new groundwater supplies and underground water storage options in the same aquifers underlying the OCW. The findings of our study contribute options to supplement these and other local municipal water supplies [50] and can inform proposed aquifer storage and recovery (ASR) projects in the surrounding region. ASR is a proven technology used by many municipalities where water is injected and stored in subsurface aquifers for future extraction. Use of ASR may be possible within the volcanic rocks underlying the OSU agricultural research facilities. Comparable hydrogeologic compartmentalization of the volcanic rocks in the OCW were proven to be instrumental in ASR in the volcanic rocks underlying Warren, Oregon, USA as described by Mansfield [44].

While proximity does not guarantee viability, the OCW is located near other ASR projects utilizing the volcanic rocks for subsurface storage. Aquifer storage and recovery techniques have been shown to be successful within the volcanic rocks serving as water supplies for the City of Dallas, Oregon, USA. The Dallas ASR system is unique in Oregon because the naturally brackish groundwater found in the volcanic rocks mixes with the injected water. However, recovery of injected water has been sufficient to supplement municipal water supplies during periods of water scarcity.

The neighboring City of Philomath is also pursuing ASR for supplementing municipal drinking water supplies and has generated plans to develop two existing drinking water wells. Philomath water managers expect the ASR project to reduce water demands on the nearby Marys River, which has experienced record low flows during the summer months. Preliminary pump tests and other performance analyses conducted on the City's wells have shown promising results.

The minimal overburden overlying the SRV in the OCW contribute to the viability of managed recharge. Likewise, constructed wetlands could be used to treat water before infiltration. The general lack of land development, except for small agricultural research facilities, research forest, and the few rural residences, make the OCW a valuable location for managed aquifer recharge.

Hydrogeological information in the OCW region is limited as deep drilling in the area has been sparse. The 650-m deep oil exploration well drilled east of the OCW encountered a sufficient water flow that well casing was installed to shut off the flow. The old well records do not report whether the water was fresh or saline, or any estimated flow rates. However, the available deep well data from nearby oil and gas exploration wells suggests that a well approaching 500 m in depth located east of the Corvallis Fault may encounter fine-grained sandstones capable of yielding sufficient water quantities to facilities located in the lower portion of the OCW, though the produced water quality remains unknown. Further investigation, including the drilling of a small diameter test hole followed by geophysical logging, would provide information on the number and thickness of the interbedded sandstones encountered in the test hole. Geophysical logging would also provide a first-order approximation of the salinity of water stored in the sandstones.

5. Conclusions

This case study contributes critical information toward enhancing understanding of local hydrogeologic features and improved groundwater resources management in areas near coastal ranges such as those found in the Pacific Northwest, USA. While this study's goal was to characterize the hydrogeologic framework within and near the Oak Creek Watershed (OCW) and enhance base knowledge of groundwater flow and aquifer fluid dynamics in the region, we were able to link this information for current and future water uses. First, the interaction of geology and hydrology are assessed with particular focus on the hydrologic role of the Corvallis Fault. The hydraulic barrier-conduit role of the fault contributes to the compartmentalization of the volcanic rock aquifer, creating two hydraulically separated groundwater compartments. The complex intrinsic and secondary permeability architecture of the volcanic rock aquifers imposed through flows and later structural geologic history link two adjacent watersheds through inter-basin groundwater flow. These findings provide an opportunity for proactive land and watershed management practices given the OCW and nearby mountainous regions are used, or are anticipated to be used, for residential and municipal drinking water supplies, either through direct capture through wells or secondary capture through managed aquifer recharge storage and recovery. Deep drilling frontiers for underutilized groundwater and aquifer storage exist, as shown by the limited hydrogeologic data reported for a smattering of past oil and gas exploration wells near the OCW.

Author Contributions: C.G.O., W.T.J. and J.H. developed the study design. J.H. conducted field data collection. All co-authors contributed to data analyses and the writing of the manuscript. All authors have read and agreed to the published version of the manuscript.

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Conflicts of Interest: The authors declare no conflict of interest.

Appendix A

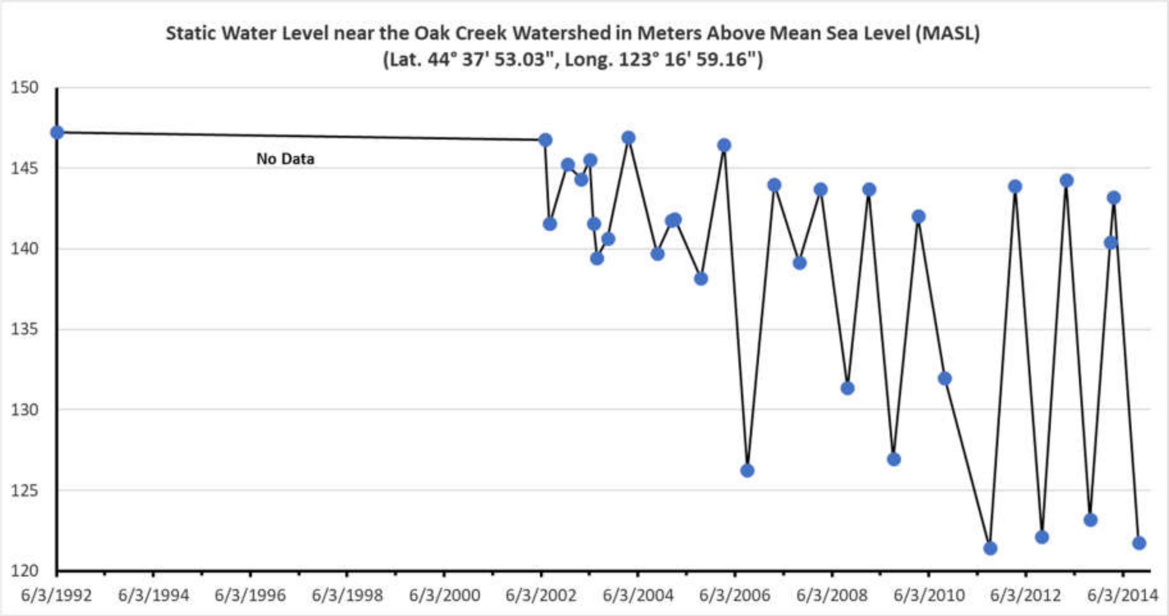

Figure A1. Example of Static Water Level Measurements at Observation Well near the Oak Creek Watershed (OCW), Corvallis, Orgon, USA. Data source (https://www.oregon.gov/owrd (accessed on 30 January 2022)).

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