Prepared in cooperation with the Albuquerque Bernalillo County Water Utility Authority

Water-Quality Trends in Surface Waters of the Jemez River and Middle Rio Grande Basin from Cochiti to Albuquerque, New Mexico, 2004–19

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Cover. Photograph showing the Rio Grande between Highway 550 and Alameda, New Mexico, with the Sandia Mountains in the background. Photograph by Justin R. Nichols, U.S. Geological Survey.

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By Allison K. Flickinger and Zachary M. Shephard

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Appendix 2. Concentration Uncertainty Analysis
Appendix 3. Non-Organic Constituents Results Summary
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Conversion Factors

U.S. customary units to International System of Units

Multiply	By	To obtain
	Length	
inch (in.)	2.54	centimeter (cm)
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
square mile (mi^2)	259.0	hectare (ha)
square mile (mi^2)	2.590	square kilometer (km^2)
	Flow rate	
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)
	Radioactivity	
picocurie per liter (pCi/L)	0.037	becquerel per liter (Bq/L)

International System of Units to U.S. customary units

Multiply	By	To obtain
	Volume	
liter (L)	33.81402	ounce, fluid (fl. oz)
liter (L)	2.113	pint (pt)
liter (L)	1.057	quart (qt)
liter (L)	0.2642	gallon (gal)
liter (L)	61.02	cubic inch (in^3)
	Flow rate	
cubic meter per second (m^3/s)	70.07	acre-foot per day (acre-ft/d)
cubic meter per second (m^3/s)	35.31	cubic foot per second (ft^3/s)
cubic meter per second (m^3/s)	22.83	million gallons per day (Mgal/d)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$°F = (1.8 \times °C) + 32.$$

Datum

Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88).

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Elevation, as used in this report, refers to distance above the vertical datum.

Supplemental Information

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius (µS/cm at 25 °C).

Concentrations of chemical constituents in water are given in either milligrams per liter (mg/L) or micrograms per liter (µg/L).

Concentrations of *Escherichia coli* (*E. coli*) and fecal coliforms in water are reported in colony forming units per 100 milliliters.

Activities for radioactive constituents in water are given in picocuries per liter (pCi/L).

Water year is defined as beginning on October 1 and continuing through September 30 of the following year.

Abbreviations

ABCWUA	Albuquerque Bernalillo County Water Utility Authority
DEET	*N,N*-diethyl-*meta*-toluamide
EPA	U.S. Environmental Protection Agency
FN	flow normalized
MCL	maximum contaminant level
MRGB	monitoring program Middle Rio Grande Basin monitoring program
NMED	New Mexico Environment Department
NMOSE/ISC	New Mexico Office of the State Engineer/Interstate Stream Commission
p-value	probability value
RSD	relative standard deviation
SKT	Seasonal Kendall Tau
SMCL	secondary maximum contaminant level
UCL	upper confidence limit
USGS	U.S. Geological Survey
VOC	volatile organic compound
WRTDS	Weighted Regressions on Time, Discharge, and Season

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By Allison K. Flickinger and Zachary M. Shephard

Abstract

Municipal water supply for Albuquerque, New Mexico, is provided, in part, through diversion of surface water from the Rio Grande by way of the San Juan-Chama Drinking Water Project diversion structure. Changes in surface-water quality along the Rio Grande and its tributaries upstream from the San Juan-Chama Drinking Water Project diversion structure are not well characterized. This study describes the methods and results of an analysis of surface-water-quality trends for selected constituents in the Rio Grande upstream from Albuquerque. Trends were evaluated for differing time periods ranging from 2004 to 2019 by using the Seasonal Kendall Tau (SKT) test and the Weighted Regressions on Time, Discharge, and Season (WRTDS) model.

Water-quality data at three long-term sites were used for the trend analyses in this study, with the Cochiti and Alameda sites along the Rio Grande and the Jemez Canyon Dam site along the Jemez River, a tributary of the Rio Grande. The proximity of the Cochiti and Jemez Canyon Dam sites to dams is a drawback to the analysis because it is difficult to differentiate between the influence of dam management and the influence of streamflow on water-quality trends. The data used also did not fully meet desired levels of seasonal sampling density and had shorter periods of record than typically used for trend analysis, and this should be considered in the interpretation of these results.

Study results indicate that concentrations, and thereby fluxes, are influenced by changes in streamflow at the Alameda site. Most trends from the WRTDS results, obtained by using flow-normalization, were downward for constituents at the Alameda site. Most constituents that were analyzed for trends by using SKT did not have a significant trend at any of the sites included in this study, indicating either that the water quality in the Middle Rio Grande Basin has been stable during the study period or that not enough samples were collected during different seasons to characterize the range of concentration variability with streamflow. The SKT test results indicate upward trends in concentrations of the following constituents: aluminum and antimony at the Alameda site, nitrate and nitrate plus nitrite at the Cochiti site, and potassium and antimony during the spring season at Jemez Canyon Dam. The SKT test results indicate a downward trend in cobalt at the Cochiti site that is subject to bias in the cobalt concentrations. SKT test results also indicate small, downward trends in Kjeldahl nitrogen at the Alameda and Cochiti sites.

Concentrations of water-quality constituents were also compared to Federal and State water-quality standards to provide context and relevance to the results. No concentrations were above the national primary or secondary drinking water standards at the Alameda and Cochiti sites, but the Jemez Canyon Dam site did have concentrations above the U.S. Environmental Protection Agency primary drinking water standard for arsenic and above the national secondary drinking water standards for dissolved solids and aluminum. The Alameda and Cochiti sites are on reaches of the Rio Grande that are listed as impaired for gross alpha particles and the Alameda site is on a reach of the Rio Grande that is listed as impaired for *Escherichia coli*, but there were no consistent changes in concentrations of these constituents at the impaired locations.

Introduction

The Albuquerque Bernalillo County Water Utility Authority (ABCWUA) provides water and wastewater services to more than 680,000 water users in the greater Albuquerque, New Mexico, metropolitan area (ABCWUA, 2019). In December 2008, the ABCWUA began diverting surface water from the Rio Grande as part of the San Juan-Chama Drinking Water Project to supplement municipal water supply. This water source has increased from composing about 20 percent of the annual supply to Albuquerque and the surrounding areas in 2009 to composing about 70 percent of the annual supply in 2019 (ABCWUA, 2016, 2020). The monitoring and analyses of surface-water quality in the Rio Grande and its tributaries upstream from the San Juan-Chama Drinking Water Project diversion structure is necessary to maintain use of this critical water source.

The U.S. Geological Survey (USGS), in cooperation with the ABCWUA, monitors groundwater and surface-water quantity and quality as part of the Middle Rio Grande Basin monitoring program (MRGB monitoring program). The purpose of the MRGB monitoring program is to improve the understanding of groundwater and surface-water resources in the Middle Rio Grande Basin, including groundwater/surface-water interaction and water quality, and assist the ABCWUA in maintaining an adequate supply of potable water for future needs.

As part of the MRGB monitoring program, the USGS has been collecting water-quality samples at several sites on the Rio Grande and its tributaries in the Middle Rio Grande Basin since 2004. These data are vital in understanding not only water-quality conditions, but also water-quality trends and how constituents' concentrations compare to the U.S. Environmental Protection Agency (EPA) standards and the New Mexico Environment Department (NMED) water-quality criteria.

Purpose and Scope

Trends in concentrations of selected constituents were evaluated for differing time periods ranging from 2004 to 2019 based on the record of sampling at each of the long-term sites along the Rio Grande and tributaries upstream from Albuquerque (table 1). The results of this study are intended to identify any observed trends in water-quality constituents that may potentially be of concern for the users. Analysis of these trends can help inform the ABCWUA on how to best monitor and manage this water supply in the future. Additionally, these results may be of regional interest for other water users and managers who rely on water in the Middle Rio Grande Basin or downstream sections of the Rio Grande and may serve as an example of water-quality trend analysis for other users and managers in other regions of the United States.

Description of the Middle Rio Grande Basin Area

The Middle Rio Grande Basin (Basin) is commonly defined as the basin encompassing the stretch of the Rio Grande between Cochiti and San Acacia (Bartolino and Cole, 2002; fig. 1). The Basin is made up of a variety of environments, including the natural landscape range from riparian forests of the river valleys to the high desert and human-altered landscapes of developed cities and irrigation canals and drainage networks for agriculture. Major population centers in the study area include Santa Fe and Albuquerque, which have seen population growths of approximately 20 and 22 percent, respectively, from 2000 to 2019 (U.S. Census Bureau, 2003, 2019). Irrigation for agriculture occurs along the Rio Grande Valley, typically as a network of irrigation canals and return drains that have the potential to introduce salinity, nutrients, and other organic compounds from agriculture to the river. The Rio Grande and its alluvial floodplain bisect the Basin, and elevations can vary from approximately 4,500 to over 8,000 feet (ft) from the river channel to the basin-bounding uplifts, with peaks of the mountains adjacent to the basin greater than 10,000 ft (Bartolino and Cole, 2002).

Precipitation in the Basin varies both seasonally and spatially. Precipitation in the winter is mostly in the form of snow, and precipitation in the summer is in the form of rain that is driven by the North American monsoonal weather pattern (New Mexico Office of the State Engineer/Interstate Stream Commission [NMOSE/ISC], 2017). Mean annual precipitation ranges from about 30 inches in mountainous regions to 8 inches in the valley along the Rio Grande (NMOSE/ISC, 2017). About 60 to 75 percent of total streamflow in the Rio Grande is generated from seasonal snowpack in the San Juan and Sangre de Cristo Mountains located in the Upper Rio Grande Basin (Rango, 2006; Llewellyn and Vaddey, 2013).

The flow regime of the Rio Grande in the Upper Rio Grande Basin, upstream from Cochiti Lake, is strongly driven by spring peak flows from the snowmelt runoff in the headwaters and mountainous regions (Rango, 2006; Llewellyn and Vaddey, 2013) (fig. 1). Summer flows are influenced by diversions for irrigation and late season monsoonal storm events and are followed by low flows in the winter. The streamflows of the Middle Rio Grande, however, are much more controlled by reservoir releases than by seasonal influences.

Rio Grande streamflows originating in the Upper Rio Grande Basin are passed through the U.S. Army Corps of Engineers flood-control reservoir, Cochiti Lake. As the most upstream point of the Basin, the releases from the Cochiti Dam control the streamflows of the Rio Grande in the Basin. Inflow also comes from tributaries in the Basin, including the Jemez River and the Rio Chama (fig. 1). Flows in the Rio Grande are also managed by the operation of the San Juan-Chama Drinking Water Project, which was authorized in 1962 to fulfill New Mexico's allocation of Colorado River rights for use in the large population centers located on the Rio Grande (Glaser, 1998). Through a series of diversions, tunnels, and reservoirs, water is moved from the San Juan River to the Chama River. Snowmelt runoff and San Juan-Chama Drinking Water Project water are stored in two reservoirs on the Chama River and released based on need and available water.

The Rio Grande is hydraulically connected with the underlying basin-fill aquifer, which consists of the Santa Fe Group and the younger post-Santa Fe Group sediments, deposited on top of the Santa Fe Group (Bartolino and Cole, 2002). Although the Rio Grande likely both gains from and loses streamflow to groundwater throughout New Mexico, it has been determined through studies and models that the Rio Grande from Cochiti Lake past Albuquerque is typically a losing stream most of the year (Thorn, 1995; McAda and Barroll, 2002; Veenhuis, 2002) but can gain flow from groundwater during winter months (S.S. Papadopulos and Associates, Inc., 2004).

Table 1. Information for selected streamflow and water-quality monitoring sites in the Jemez River and Middle Rio Grande Basin study area, Cochiti to Albuquerque, New Mexico, 2004–19.

[USGS, U.S. Geological Survey; NM, New Mexico; dates are in month/day/year format]

Short name	USGS site name	USGS site number	Number of project samples collected, by season			Total number of samples	Project water-quality sampling period of record start	Project water-quality sampling period of record end	Record length (years)	Latitude (decimal degrees)	Longitude (decimal degrees)
			Winter	Spring	Summer						
Taos Junction	Rio Grande below Taos Junction Bridge near Taos, NM	08276500	7	7	7	21	2/24/2004	8/9/2010	6.5	36.3200	−105.7544
Otowi	Rio Grande at Otowi Bridge, NM	08313000	7	6	7	20	2/23/2004	8/11/2010	6.5	35.8745	−106.1424
San Felipe	Rio Grande at San Felipe, NM	08319000	6	6	6	18	2/10/2004	8/20/2009	5.5	35.4446	−106.4398
Chamita	Rio Chama near Chamita, NM	08290000	7	7	7	21	2/24/2004	8/10/2010	6.5	36.0736	−106.1117
Cochiti[1]	Rio Grande below Cochiti Dam, NM	08317400	11	10	10	31	11/19/2009	8/19/2019	9.7	35.6180	−106.3239
Alameda[1]	Rio Grande at Alameda Bridge at Alameda, NM	08329918	16	16	16	48	2/18/2004	8/21/2019	15.5	35.1977	−106.6428
Jemez Canyon Dam[1,2]	Jemez River below Jemez Canyon Dam, NM	08329000	0	7	0	7	4/21/2004	5/7/2010	6.0	35.3904	−106.5346
Jemez Canyon Dam[1,2]	Jemez River outlet below Jemez Canyon Dam, NM	08328950	0	10	0	10	4/14/2011	3/12/2019	7.9	35.3948	−106.5453

[1]Site was considered a long-term site and was used for trend analyses.

[2]Data from both Jemez Canyon Dam sites were combined for this study.

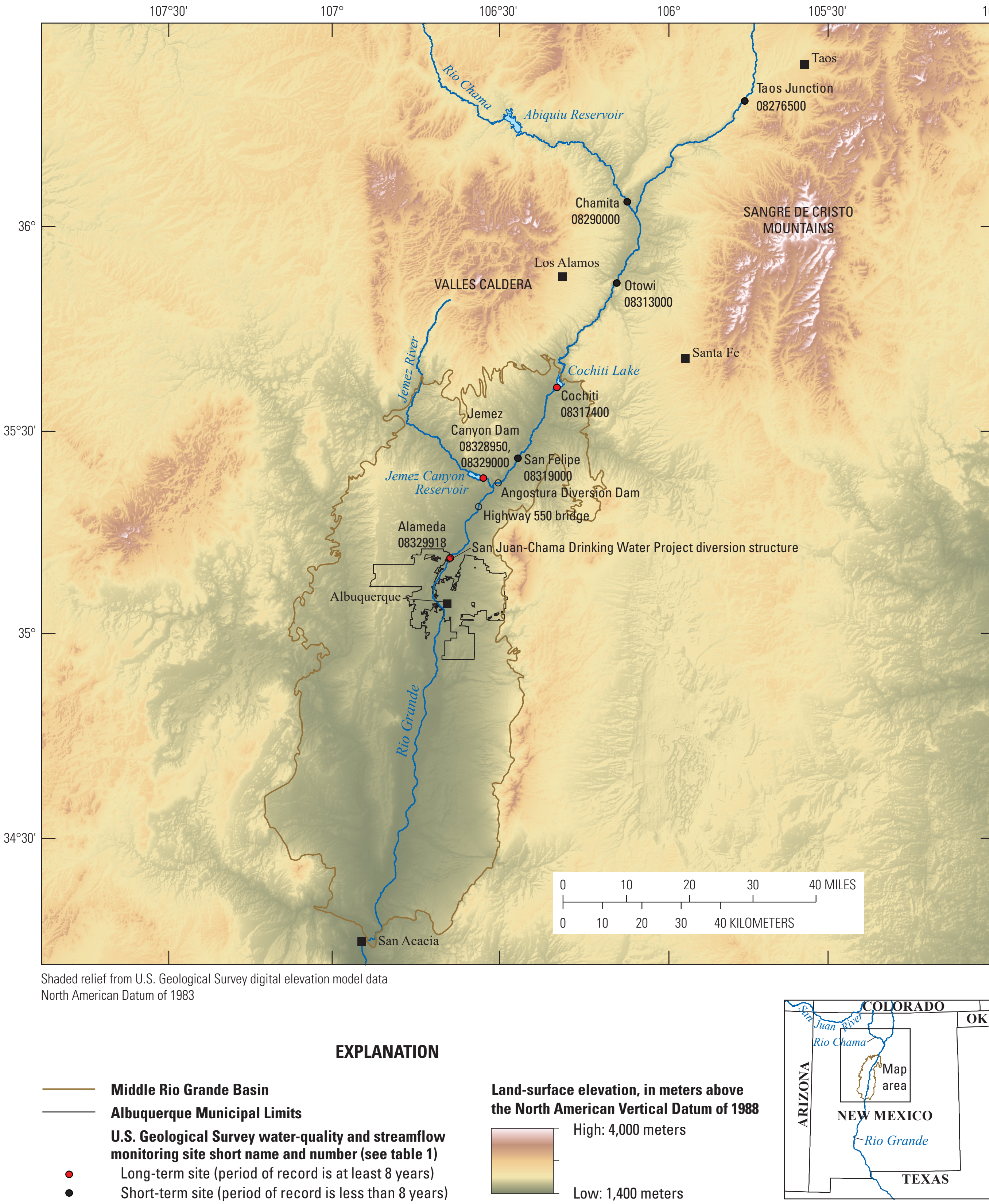

Figure 1. Location of sites used for the water-quality trend analysis of surface waters of the Jemez River and of the Rio Grande between Taos and Albuquerque, New Mexico, 2004–19.

Methods

Streamflow and Water-Quality Data Collection

Water-quality sampling for the MRGB monitoring program took place at seven sites along the Rio Grande and its tributaries in New Mexico, ranging from the farthest upstream site at Taos Junction (08276500; fig. 1) to the farthest downstream site at Alameda (08329918; fig. 1). The periods of record for the sampling locations vary, ranging between the years 2004 to 2019. Most sites were sampled three times a year, with sampling intended to capture the winter, spring, and summer flows. The sampling locations just downstream from the Jemez Canyon Dam were the exception, with samples only collected during the spring runoff season. A summary of the sampling sites and date ranges as well as the number of samples collected per season can be found in table 1. Of the seven sites, only three have records of 8 years or longer and are considered the long-term sites for inclusion in the trend analysis (table 1).

The Taos Junction and Otowi sites are located on the Rio Grande upstream from the Cochiti Dam in the Upper Rio Grande Basin. The Taos Junction and Otowi sites each have a period of record of 6.5 years (table 1), so they were not considered long-term sites for this study. The San Felipe site, located along the Rio Grande downstream from the Cochiti Dam, only has a period of record of 5.5 years, so it was not considered a long-term site for this study. The Chamita site is located on the Rio Chama, a tributary to the Upper Rio Grande. The Chamita site only has a period of record 6.5 years, so it was not considered a long-term site for this study.

The Cochiti and Alameda sampling locations (fig. 1) along the Rio Grande are located downstream from the Cochiti Dam, and the streamflow at these locations is controlled by dam releases. The Cochiti and Alameda sites have periods of records of 9.7 and 15.5 years, respectively, so they are considered long-term sites for this study (table 1). The surface-water diversion for the ABCWUA San Juan-Chama Drinking Water Project is located just south of the Alameda site (fig. 1). The drainage area for this location, about 12,700 square miles, includes the Upper and Middle Rio Grande Basin.

The Jemez Canyon Dam sampling location is on a tributary, the Jemez River, downstream from the Jemez Canyon Dam. The dam regulates streamflow for flood- and sediment-control purposes. The Jemez River enters the Rio Grande between the San Felipe and Albuquerque sites (fig. 1). The Jemez River drains an area that has geothermal activity and contains larger dissolved solids concentrations and different mineral content than the Rio Grande (Trainer and others, 2000). The sampling location along the Jemez River near the mouth of the Jemez Canyon Dam moved approximately 3,700 ft downstream in 2010 (from USGS site number 08329000 to 08328950). No major inlets nor outlets are between the two sites, so the records for the two sites were combined for the purpose of this study. When the two periods of record for the sites are combined, the Jemez Canyon Dam location has a period of record of almost 14 years (table 1), so it was considered to be a long-term site for this study. Samples were collected at the Jemez Canyon Dam location only during the spring runoff season.

Daily mean streamflows for each of the long-term sites were obtained from the USGS National Water Information System database (USGS, 2020; https://doi.org/10.5066/F7P55KJN). Water-quality samples were collected and processed by using standard equal-width isokinetic increment collection methods representative of the entire water column (USGS, 2006). Samples were analyzed for major ions, trace elements, nutrients, and organic constituents at the USGS National Water Quality Laboratory in Denver, Colorado; for radiochemical constituents at various USGS contract laboratories; and for bacterial constituents at the USGS New Mexico Water Science Center according to published USGS laboratory methods (Myers and others, 2014). Water-quality data are available in the USGS National Water Information System database (USGS, 2020) by using the USGS site number in table 1.

Quality Assurance and Uncertainty Analysis

Methods described by Mueller and others (2015) and Beisner and others (2017) were used to quantitatively characterize the bias and uncertainty in the environmental sample dataset used in this study. Several blank and replicate samples were collected in the field in conjunction with the environmental samples for use in this analysis. This bias and uncertainty analysis puts trends into context, with potential variability caused by the sampling and laboratory analysis processes. Field blanks and replicate samples were analyzed with respect to the environmental samples at the Alameda, Cochiti, and Jemez Canyon sites because these were the sites used in the trend analysis for this study. Laboratory surrogate recovery results were also reviewed.

In this study, 21 field blanks were collected between 2006 and 2019 at Alameda and Cochiti. No blank samples are available from Jemez Canyon, but the potential bias in the Alameda and Cochiti blanks is applicable to Jemez Canyon because the sampling methods were consistent among the three sites. Concentrations of constituents measured in environmental samples that are substantially above the concentrations measured in blank samples are less likely to be affected by potential bias introduced through field conditions or sampling equipment. For a given constituent, a threshold value of 10 times (hereinafter, 10x threshold) the maximum blank measurement was used to determine whether the environmental samples were likely affected by high bias introduced through sampling conditions. Environmental concentrations above this 10x threshold contain less than 10 percent influence from contamination and are not likely to be substantially affected by high bias. Additionally, upper confidence limits (UCLs) were determined for each constituent. UCLs are defined as the confidence that the maximum concentration in all of the blank samples is at least as large as the true percentile of contamination. For example, a confidence of 50 for a 90th percentile UCL indicates a 50-percent confidence that the maximum concentration of a constituent reported in a blank sample population captures the 90th percentile of the maximum contamination in the environmental sample population. UCLs are included in this report to further characterize and validate the uncertainty calculations and are presented at the 80th and 90th percentiles.

In this study, 15 replicate samples were collected between 2008 and 2019 at Alameda and Cochiti. Again, replicate samples were not collected at Jemez Canyon, but the sampling methods of the environmental samples among the three sites were consistent. Uncertainty in samples' concentration was determined by the variability in replicate-environmental pairs. For each constituent, the 80- and 90-percent confidence intervals were calculated from the mean standard deviation between all individual replicate-environmental sample pairs (Beisner and others, 2017). The intervals indicate a 90- or 80-percent confidence that the true value of any individual measurement for the listed constituent lies within the range of uncertainty above and below the measured concentration provided. Uncertainty is also expressed for each constituent as the mean relative standard deviation (RSD), which is defined as the ratio (in percent) of standard deviation to mean concentration (Mueller and others, 2015).

Streamflow Data Aggregation

Streamflow metrics at the long-term sites were aggregated to inform the discussion of the water-quality trends. Three different annual streamflow metrics were plotted: (1) annual mean daily streamflow, defined as the annual mean of the daily streamflows; (2) annual maximum daily streamflow, defined as the annual maximum of the daily streamflows; and (3) annual minimum 7-day daily streamflow, defined as the annual minimum of the 7-day mean of daily streamflows.

Water-Quality Trend Analysis

Trends were analyzed for all constituents of interest with a sufficient number of noncensored-concentration data at the three long-term sites (Alameda, Cochiti, and Jemez Canyon Dam; appendix 4, available online at https://doi.org/10.3133/sir20225062). Trend analysis was performed by using a combination of the Weighted Regressions on Time, Discharge, and Season (WRTDS) model and the Seasonal Kendall Tau (SKT) test. WRTDS was not run for the constituents at Cochiti because of its short period of record and few available samples; WRTDS was not run for the constituents at Jemez Canyon Dam because of a lack of seasonal samples (samples were only collected during the spring). Some constituents do not have reported trend results because the models did not converge or because they did not meet performance criteria described below. Not all constituent groups and constituents were considered appropriate for trend analysis. Fecal coliforms and *Escherichia coli* (*E. coli*) tend to be episodic, and the sample frequency for this project was not sufficient to account for the potential variability and to determine trends for these constituents. The individual sample concentrations were also compared to water-quality standards to add context to the results, and the magnitude of change and direction of any existing trends were taken into consideration as part of these comparisons.

Data Screening for Trend Analysis

Six constituent groups of interest were selected for this study: major ions and dissolved solids, bacteria, trace elements, nutrients, radiochemicals, and organic constituents. The trend analyses were performed on all the appropriate constituents within these groups of interest that had sufficient noncensored data. Censored data is defined as data for which the value is partially known; left-censored data (also called a nondetection or less-than value) occurs when a measurement is less than a certain value by an unknown amount. Sufficient noncensored data was defined as less than 50 percent censored data for the WRTDS model (Hirsch and De Cicco, 2015) and at least 10 detected observations for the SKT test (Schertz and others, 1991).

Seasonal Kendall Tau Test

The SKT test is a robust, nonparametric trend test commonly used to test for trends in environmental parameters (Hirsch and Slack, 1984). The test incorporates seasonality by segregating data seasonally, analyzing the data separately, and then combining the results to compute an overall trend statistic. SKT trends were computed by using the USGS EStimate TREND (ESTREND) method (Schertz and others, 1991), implemented in the R programming environment (R Core Team, 2020) using the package R-ESTREND (available at https://github.com/USGS-R/restrend/).

Concentrations of water-quality constituents are often correlated with streamflow. Concentrations of some constituents decrease with an increase in streamflow (dilution) or increase with an increase in streamflow caused by runoff, and sometimes, both situations occur. Because the detection of water-quality trends may be complicated by the presence of flow-related variability in water-quality records, a reduction in flow-related variability will increase the power of the SKT test and the chance of detecting a trend that results as an influence of something other than streamflow. To reduce the flow-related variability in the trend analysis results, the flow-adjusted SKT test was run for constituents with less than 5 percent left-censored data. Flow adjustment was performed by fitting a locally weighted regression (Loess) (Cleveland and others, 1988) to the log(concentration)-log(streamflow) relation for each combination of site, time period, and constituent. The flow-adjusted trend was calculated by removing the predicted concentration from the seasonal data prior to running the SKT test, resulting in a trend that was conceptually similar to the flow-normalized trends from WRTDS. The log flow-adjusted concentrations (LC_{FA}) were the log of the residuals of the regression, defined as

$$LC_{FA} = LC_{OBS} + LC_{PRED} \tag{1}$$

where

LC_{FA} is the log flow-adjusted concentrations,
LC_{OBS} is the log of the reported concentration, and
LC_{PRED} is the predicted log concentration from the log(concentration)-log(streamflow) Loess model.

The streamflows at all of the long-term sites are affected by dams, with the Cochiti and Jemez Canyon Dam sites located just downstream from dams, and this complicates trend analysis. It can be difficult to differentiate between the influence of dam management on the water-quality trends and the influence of streamflow. In consideration of this potential influence on the water-quality data, the relationship between constituent concentrations and streamflow was assessed to determine whether use of flow adjustment was appropriate, and the flow-adjusted models were evaluated to determine whether the fit of the regression was adequate. The flow-adjustment residuals, which are the differences between the predicted concentrations and the reported concentrations, were looked at in comparison to the sample concentrations, streamflows, and sample dates. If bias was present in the residuals, use of the flow-adjusted SKT test was not appropriate, and the non-flow-adjusted SKT test was used instead. Flow adjustment was not used in cases where constituents had greater than 5 percent of the water-quality record censored as the censored SKT test was implemented, and uncertainty associated with the censored values precludes the use of flow adjustment (Schertz and others, 1991).

Constituents with an SKT probability value (p-value) less than 0.05 were reported as having a significant trend. The trend slope, the median slope of all pairwise comparisons, was computed (Sen, 1968) and is expressed in this report as a percentage; this percentage was calculated by dividing the trend slope by the median water-quality concentration. Constituents with greater than 5 percent of the water-quality record censored do not have a reported trend percentage change because the estimate of the slope and calculation of the trend percentage are influenced by the value selected to represent nondetected concentrations.

Weighted Regressions on Time, Discharge, and Season

Another method of trend analysis used in this study is the WRTDS model (Hirsch and others, 2010; Hirsch and De Cicco, 2015). The WRTDS model was implemented by using two R packages: (1) Exploration and Graphics for RivEr Trends (EGRET) package to run the WRTDS model (Hirsch and De Cicco, 2015) and (2) EGRETci package to perform analysis of the uncertainty associated with the WRTDS model (Hirsch and others, 2015).

The EGRET package estimates daily concentration from the daily mean discharge and regression coefficients for the indicated period of record as follows:

$$\ln(c) = \beta_0 + \beta_1 t + \beta_2 \ln(Q) + \beta_3 \sin(2\pi t) + \beta_4 \cos(2\pi t) + \varepsilon(16) \tag{2}$$

where

ln is natural log,
c is concentration,
$\beta_0 \ldots \beta_4$ are fitted regression coefficients,
Q is daily mean streamflow, in cubic meters per second,
t is time, in decimal years, and
ε is the unexplained variation.

The default model parameters were used for the WRTDS models used in this study, except for the “minNumObs” parameter, which had to be reduced from 50 to 30 in order to accommodate for the smaller dataset.

Flow normalization removes the variation in concentration and load that is due to random and systematic variations in discharge but retains the influence of nonrandom seasonal variations. It is implemented in WRTDS by using the assumption of a stationary discharge regime over the period of record, and flow-normalized (FN) estimates of concentration and load are produced as estimates of concentration and load that occur at the mean discharge for each calendar day of the year over the trend period. The FN concentrations and loads are intended to describe the changing state of the system over time by integrating out the influences of variations in concentration or flux that arise from the day-to-day variations in discharge.

For the WRTDS trend analysis for this study, daily FN mean concentrations were aggregated to water-year (beginning October 1 and continuing through September 30 of the following year) means. The FN water-year concentration means were used to quantify the trend magnitude by calculating the percent change between the first and last FN water-year concentration means of the model period.

The EGRETci package uses a bootstrap method to determine a 90-percent confidence interval on the magnitude of the trend and a likelihood statistic that is the functional equivalent to the two-sided p-value. The likelihood statistic provides information on whether the null hypothesis, that there is no trend in the FN concentrations over the period of record, should be rejected and provides a measure of the strength of evidence that the trend exists. For this study, the likelihood of a net positive or negative change in concentrations or loads (in percentage) had to be greater than 0.7 for a model to have evidence of a trend.

Although the original guidance for WRTDS recommended its use at sites with long-term (at least 20 years) and dense (at least 200 samples) water-quality datasets and with continuous daily streamflow for the entire water-quality period of record (Hirsch and others, 2010), refinements to WRTDS and more recent experience with the method have indicated that a reasonable trend estimation can be achieved for datasets as short as 10 years and with as few as 60 samples (Hirsch and De Cicco, 2015; Chanat and others, 2016). In addition, an appropriate dataset should also represent the seasonal variability in constituent behavior as well as represent high-flow periods (Oelsner and others, 2017). Alameda was the only site with a potentially appropriate dataset for WRTDS because Cochiti has a period of record of only 9.7 years and 30 samples and Jemez Canyon Dam samples were only collected during spring. Although samples were collected at Alameda starting in the 2004 water year, there is a gap in the streamflow data during the 2005 water year. To get a complete record of daily streamflow data, the model was run beginning with the 2006 water year, resulting in a dataset of 42 samples over 14 years. While the dataset used for Alameda meets the minimum record length, it does fall short of the desired sample density. The samples from Alameda targeted the three main flow regimes of the Rio Grande; however, samples were only collected during three seasons (winter, spring, and summer), and samples used for the WRTDS model would ideally represent all four seasons to account for variation in concentrations caused by temperature, sunlight, and other environmental conditions. Additionally, although the Alameda site is not directly downstream from a dam, the streamflow at the site is influenced by the releases from the Cochiti Dam, meaning that fluctuations in streamflow may not necessarily correspond to changes in concentration.

Because of the shortcomings of the Alameda dataset, emphasis was placed on evaluating the performance and bias of the models. The relationship between concentration and streamflow was considered for each constituent, as required by the WRTDS model. The WRTDS output was also checked visually for fit, residual structure, and residual bias according to the procedures described in Hirsch and De Cicco (2015). In addition to the model output being visually checked, the flux bias statistic for the WRTDS models was also considered. A flux is the total mass of a constituent moving past a point over a specified unit of time. The flux bias statistic for the WRTDS model is a value representing the difference between the sum of estimated fluxes of sampled days and the sum of the measured fluxes of sampled days, where an unbiased model would have the desired flux bias statistic value of zero. Only models with a flux bias statistic less than 0.2 were considered as potentially adequate for this study.

Trend Determination and Comparison to Water-Quality Standards

For this study, only trends in concentration or flux with a likelihood greater than 0.7 (for WRTDS results) or a *p*-value less than 0.05 (for SKT results) were considered positive (upward) or negative (downward). Magnitude of trends were also used to consider the effect of a trend. Conversely, trends in concentration or flux with a likelihood less than 0.7 (for WRTDS results) or a *p*-value greater than 0.05 (for SKT results) or an overall magnitude less than 5 percent per decade were considered stable.

The constituent concentration summary statistics (appendix 4) and the results from the trend analyses were compared to water-quality standards to provide context and relevance. Concentrations of constituents were compared with EPA primary and secondary drinking water standards (EPA, 2020). Also, sampling locations where trends occurred were compared with the NMED list of impaired waters (NMED, 2020).

Results

Results from the quality assurance and uncertainty analyses, streamflow data analyses, and the water-quality trend analyses are presented. Selected results pertaining to the constituents used in the trend analyses are discussed in the text, and the complete sets of results for all parameters can be found in appendixes 1–4, available online at https://doi.org/10.3133/sir20225062.

Quality Assurance and Uncertainty Results

Comprehensive quality assurance results for all constituents are in appendixes 1 and 2 (available online at https://doi.org/10.3133/sir20225062). Most of the constituents were not detected at concentrations greater than laboratory reporting levels in the blank samples (appendix 2). Only 12 constituents had detections greater than laboratory reporting levels, including constituents that were not used for trend analysis. Of the constituents used specifically in the trend analysis, only nine were measured at concentrations greater than laboratory reporting levels in field blank samples (table 2). Seven of the constituents that were detected in blank samples also were detected in environmental samples at concentrations less than their 10x threshold (10 times greater than the maximum laboratory reporting levels concentration in all blanks for the given constituent). Most of the environmental samples, however, had concentrations that were less than the maximum blank laboratory reporting level (levels vary between blank samples), and a small percentage of the total collected blanks had detections, indicating little concern of widespread contamination bias. Low-level ammonia detections in field blanks collected during this study are in the range documented by Mueller and Titus (2005).

Table 2. Summary of blank sample bias analysis results for selected constituents with reported blank detections, Jemez River and Middle Rio Grande Basin study area, Cochiti to Albuquerque, New Mexico.

[10x threshold, 10 times the maximum laboratory reporting level; UCL, upper confidence limit; wf, water, filtered; N, nitrogen; mg/L, milligrams per liter; µg/L, micrograms per liter; wu, water, unfiltered; recov, recoverable; P, phosphorus]

Constituent	Units	Laboratory reporting level	Maximum concentration in blank sample	10x Threshold	Total blank sample count	Percentage of concentrations in blanks greater than laboratory reporting level	UCL (90th percentile)	UCL (80th percentile)	Environmental sample percentage less than 10x threshold	Trends analyzed
Ammonia, wf	mg/L as N	0.02	0.01266	0.13	6	17	47	74	100	Yes
Antimony, wf	µg/L	0.14	0.05	0.50	7	14	52	79	97	Yes
Barium, wu, recov	µg/L	0.6	0.318	3.18	6	17	47	74	0	Yes
Calcium, wf	mg/L	0.04	0.027	0.27	7	14	52	79	0	Yes
Calcium, wu, recov	mg/L	0.04	0.031	0.31	6	17	47	74	0	Yes
Cobalt, wf	µg/L	0.05	0.1947	1.95	7	57	52	79	98	Yes
Copper, wf	µg/L	1	0.41	4.10	7	29	52	79	98	Yes
Organic carbon, wf	mg/L	0.4	4.188	41.88	5	40	41	67	100	Yes
Phenol, wu	µg/L	0.44	1.34	13.40	10	10	65	89	100	No
Phosphorus, wu	mg/L as P	0.008	0.0064	0.06	7	29	52	79	19	Yes
Silica, wf	mg/L	0.05	0.028	0.28	7	14	52	79	0	No
Thallium, wf	µg/L	0.04	0.02	0.20	7	14	52	79	98	Yes

Based on the concentrations measured in the blanks and environmental samples, two constituents, organic carbon and cobalt, were determined to have concentrations that are biased high. For organic carbon and cobalt, their maximum blank sample detection was higher than their blank maximum laboratory reporting level, they were detected in more than 25 percent of the blank samples, and almost all (≥98 percent) of their environmental samples were measured at concentrations less than the 10x threshold. Some organic constituents have reported concentration values that are considered to be estimates but are less than the listed laboratory reporting level values. Estimated values less than laboratory reporting levels were rarely observed in the analysis of blank samples for this study with the exception of the following organic constituents: triclosan, 2,6-dimethylnaphthalene, *N,N*-diethyl-*meta*-toluamide (DEET), phenol, and tributyl phosphate. These organic constituents had estimated reported values that indicate some potential bias among the environmental samples; however, these estimated concentrations were all less than the laboratory reporting level but greater than the laboratory detection level (table 3). These estimated detections could be a result of variability of error in laboratory analysis, and this bias is likely minimal and may have limited influence on environmental sample concentrations.

The UCL is the confidence that the maximum concentration detected in all blanks for a given constituent meets the specified percentile of maximum contamination. The 80th percentile UCLs for organic carbon and cobalt, which are constituents subject to high bias, are 67 and 79 percent, respectively. This means that the confidence that the maximum concentration (0.19 microgram per liter [µg/L] for cobalt, 4.18 milligrams per liter [mg/L] for organic carbon) detected in blanks for these constituents captures the 80th percentile of maximum contamination is 67 and 79 percent, respectively (table 2). The reason the UCLs are not greater is because of the low quantity (5 to 7) of blank samples used to evaluate the potential bias in concentrations of these constituents. Additional blank samples would yield greater UCLs, meaning greater confidence in the extent of the possible contamination. Despite the potential bias, constituents that had minimal detected concentrations in blank samples were kept in the trend analysis for this study. The results of these trends are reported with the qualification that their concentrations may be biased high, although this may not necessarily invalidate the trends. Table 2 summarizes potential sample bias determined by blank analysis and only includes samples with reported blank detections. The remaining constituents analyzed for trends were not detected in blank samples.

Uncertainties in the constituent concentrations based on replicate analysis are listed in table 4 and appendix 2. For each constituent, the 90- and 80-percent confidence intervals and mean RSD are used to represent the uncertainty in the reported concentrations. The purpose of the uncertainty calculated with the replicate data is to put the trends into context with potential variability brought on by the sampling and laboratory analysis process. The confidence in trends may decrease for samples with higher uncertainty, although the uncertainty determined in this analysis was not evaluated to validate or invalidate any trends.

The surrogate compounds alpha-HCH-d_6, polychlorinated biphenyl (PCB) congener 207, 1,2-dichloroethane-d_4, toluene-d_8, and 1-bromo-4-fluorobenzene were commonly added to samples to determine the percentage of recoveries for organic compounds. Surrogate recoveries were similar for all samples and generally within acceptable ranges with some exceptions. Ranges for surrogate recovery values were 37.2–115 percent for alpha-HCH-d_6, 34.6–117 percent for PCB congener 207, 98.2–156 percent for 1,2-dichloroethane-d_4, 79.5–105 percent for toluene-d_8, and 74.1–106 percent for 1-bromo-4-fluorobenzene. Low recoveries (less than 70 percent) for alpha-HCH-d_6 and PCB congener 207 are singular outliers for the dataset. Although recoveries of 1,2-dichloroethane-d_4, a volatile organic compound (VOC), often tend to be high, few VOCs were detected during the study period; VOCs were not analyzed for trends.

Table 3. Summary of blank sample bias analysis results for select organic constituents, Jemez River and Middle Rio Grande Basin study area, Cochiti to Albuquerque, New Mexico.

[wu, water, unfiltered; µg/L, micrograms per liter; Values are primarily estimated values less than laboratory reporting levels]

Constituent	Minimum estimated concentration (µg/L)	Maximum estimated concentration (µg/L)	Total blank sample count	Percentage of estimates reported in blanks	Maximum laboratory reporting level (µg/L)	Minimum laboratory reporting level (µg/L)
Triclosan, wu	0.022	0.116	10	30	1	0.2
2,6-Dimethylnaphthalene, wu	0.0025	0.0025	10	10	0.5	0.04
N,N-Diethyl-*meta*-toluamide, wu	0.0082	0.0603	10	30	0.5	0.04
Tributyl phosphate, wu	0.0134	0.0134	10	10	0.5	0.04

Table 4. Summary of replicate sample uncertainty analysis results for constituents analyzed for trends, Jemez River and Middle Rio Grande Basin study area, Cochiti to Albuquerque, New Mexico.

[%, percent; wf, water filtered; mg/L, milligrams per liter; inflect, inflection-point titration method; $CaCO_3$, calcium carbonate; µg/L, micrograms per liter; N, nitrogen; P, phosphorous; °C, degrees Celsius; Cs, cesium; pCi/L, picocuries per liter; NO_3+NO_2, nitrate plus nitrite; Pu, plutonium; Sr, strontium; U, uranium]

Constituent	Units	Variability (90% confidence)	Variability (80% confidence)	Mean relative standard deviation (%)	Mean concentration
Alkalinity, wf, inflect, field	mg/L as $CaCO_3$	± 0.55	± 0.43	0.46	107.6
Aluminum, wf	µg/L	± 5.80	± 4.52	19.05	35.2
Ammonia + organic N, wf	mg/L as N	± 0.02	± 0.02	9.89	0.2
Ammonia, wf	mg/L as N	± 0.00	± 0.00	2.15	0.0
Antimony, wf	µg/L	± 0.04	± 0.03	11.38	0.2
Arsenic, wf	µg/L	± 0.19	± 0.15	5.53	6.0
Barium, wf	µg/L	± 3.63	± 2.83	4.10	67.8
Boron, wf	µg/L	± 2.66	± 2.07	4.70	110.5
Cadmium, wf	µg/L	± 0.00	± 0.00	8.77	0.0
Calcium, wf	mg/L	± 1.84	± 1.44	4.15	37.6
Chloride, wf	mg/L	± 0.03	± 0.02	0.50	19.5
Cobalt, wf	µg/L	± 0.12	± 0.10	37.10	0.4
Copper, wf	µg/L	± 0.40	± 0.31	30.15	1.1
Dissolved solids, dry, at 180 °C	mg/L	± 12.22	± 9.52	4.92	237.2
Fluoride, wf	mg/L	± 0.01	± 0.01	1.48	0.4
Gross beta, wf, Cs-137	pCi/L	± 0.67	± 0.52	17.48	3.6
Lead, wf	µg/L	± 0.01	± 0.01	17.67	0.1
Magnesium, wf	mg/L	± 0.36	± 0.28	4.99	5.7
Manganese, wf	µg/L	± 0.22	± 0.17	11.49	6.8
Molybdenum, wf	µg/L	± 0.23	± 0.18	3.63	4.0
Nickel, wf	µg/L	± 0.01	± 0.01	1.20	0.8
Nitrite, wf	mg/L as N	± 0.00	± 0.00	4.12	0.0
NO_3+NO_2, wf	mg/L as N	± 0.03	± 0.02	17.01	0.1
Organic carbon, wf	mg/L	± 0.05	± 0.04	1.32	3.6
Phosphorus, wf	mg/L as P	± 0.00	± 0.00	2.52	0.0
Potassium, wf	mg/L	± 0.14	± 0.11	4.23	3.6
Pu-238, wf	pCi/L	± 0.01	± 0.00	22.95	0.0
Pu-239 + Pu-240, wf	pCi/L	± 0.01	± 0.01	51.59	0.0
Selenium, wf	µg/L	± 0.02	± 0.01	5.09	0.2
Sodium, wf	mg/L	± 1.09	± 0.85	4.51	29.7
Sr-90, wf	pCi/L	± 0.03	± 0.02	8.05	0.2
Sulfate, wf	mg/L	± 0.08	± 0.06	0.14	48.0
Tritium, wu	pCi/L	± 0.80	± 0.62	3.78	17.3
U-234, wf	pCi/L	± 0.06	± 0.05	6.11	1.1
U-235, wf	pCi/L	± 0.01	± 0.01	38.48	0.0
U-238, wf	pCi/L	± 0.07	± 0.05	9.46	0.6
Uranium, wf	pCi/L	± 0.16	± 0.13	6.23	1.9
Vanadium, wf	µg/L	± 0.34	± 0.27	6.14	4.2

Streamflow Summarization

The annual maximum daily, annual mean daily, and annual minimum 7-day daily streamflows were plotted for Alameda and Cochiti, and the same streamflow metrics were plotted for the spring season (March–May) for Jemez Canyon Dam because samples were only collected there during spring (fig. 2). Overall, the flow metrics for each of the three long-term sites have large ranges of values and fluctuations in values between sequential water years.

The streamflow data for Alameda and Cochiti indicate lower flows during the 2011–14 water years than during the 2015–19 water years (fig. 2). The maximum daily and mean daily flows were higher before and after the 2011–14 water years, and the 7-day minimum remained fairly low after this period, with the exception of the 2018 water year. The maximum daily and mean daily streamflow data for each site follow a similar pattern overall. Although trends in streamflows at Alameda and Cochiti were not consistent over the period of record for this study, the maximum and mean daily flows were mostly increasing from 2013 to 2019, with the exception of the 2018 water year. The spring and summer seasonal streamflow data for Alameda and Cochiti are similar to the annual streamflow data for each site. The winter streamflow metrics at Alameda are mostly unchanging throughout the period. At Cochiti, maximum daily and mean daily streamflow for the 2016 and 2018 winter seasons were higher than during the other years.

The flow metrics for the spring flows at Jemez Canyon Dam have less of a pronounced pattern than the flow metrics at the other two sites, but all flow metrics reflect that 2006, 2011, 2013, 2014, and 2018 had low-flow spring seasons. The metrics indicate variability between years and no consistent pattern over the period of record for this study.

Water-Quality Trend Results and Comparison to Water-Quality Standards

Water-quality trend results for the WRTDS models and SKT tests are summarized in tables 5–8 and are discussed by site in the following sections. The results for Alameda and Cochiti were obtained by using samples collected during three seasons (winter, spring, and summer) to represent annual trends. The results for Jemez Canyon Dam, however, were obtained from samples that were only collected during the spring season; therefore, the trends for Jemez Canyon Dam are only representative of trends during the spring season and are not representative of conditions during any other time of year. All constituents that were originally considered for trend analyses, including those that did not have sufficient data and those that did not have adequate model performance, are summarized in appendix 3, available online at https://doi.org/10.3133/sir20225062. Most concentrations for organic constituents were either nondetections or left-censored, so the organic constituents did not meet the trend analysis criteria. Organic constituents are summarized in appendix 4.

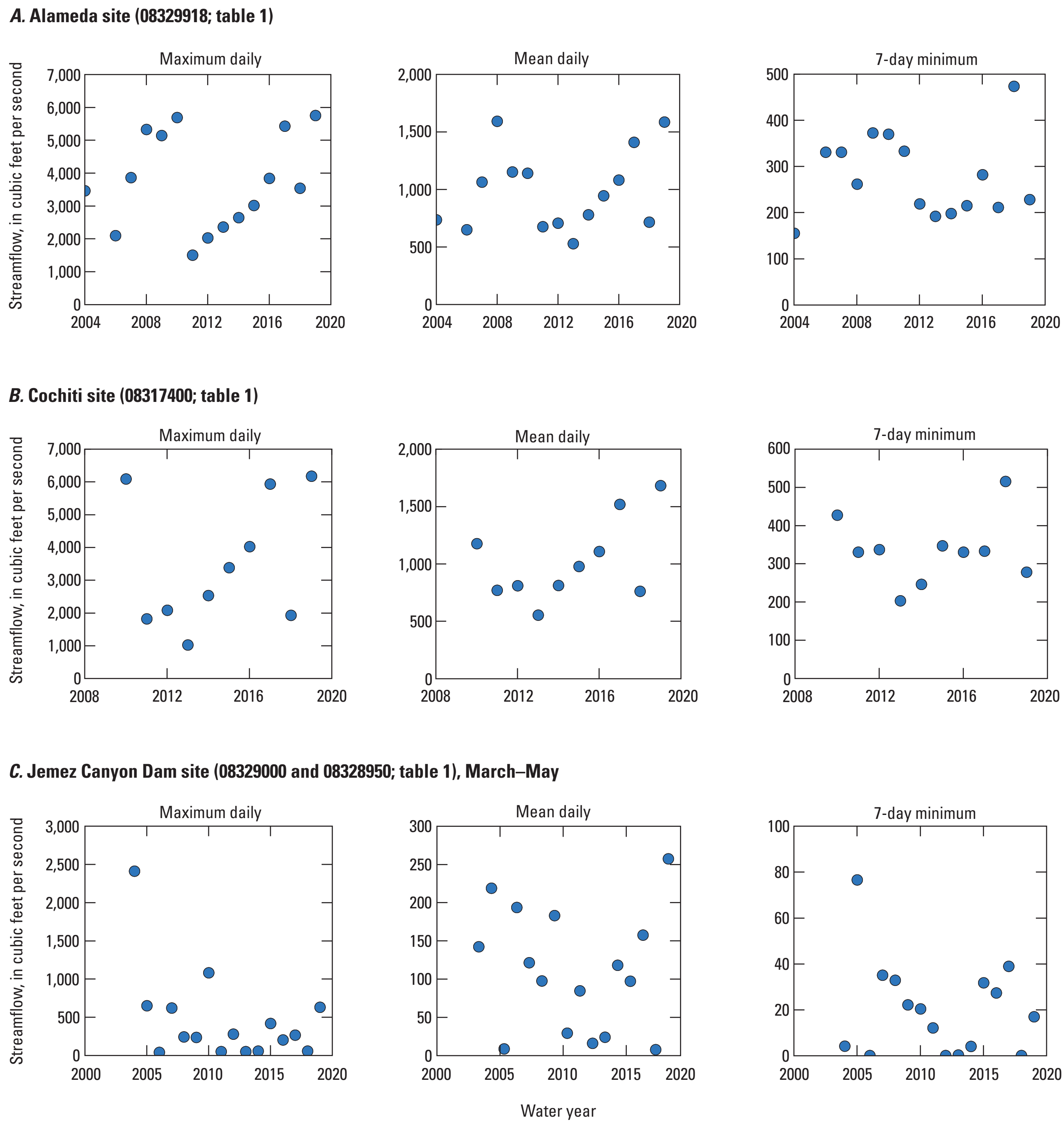

Figure 2. *A–B*, Annual streamflow metrics for the Alameda (Rio Grande at Alameda Bridge at Alameda, New Mexico, 08329918) and Cochiti (Rio Grande below Cochiti Dam, New Mexico, 08317400) sites, *C*, spring streamflow metrics for the Jemez Canyon Dam site (Jemez River below Jemez Canyon Dam, New Mexico, 08329000, and Jemez River outlet below Jemez Canyon Dam, New Mexico, 08328950), and *D–I*, seasonal (winter, spring, and summer) streamflow metrics for the Alameda and Cochiti sites, New Mexico, 2005–19.

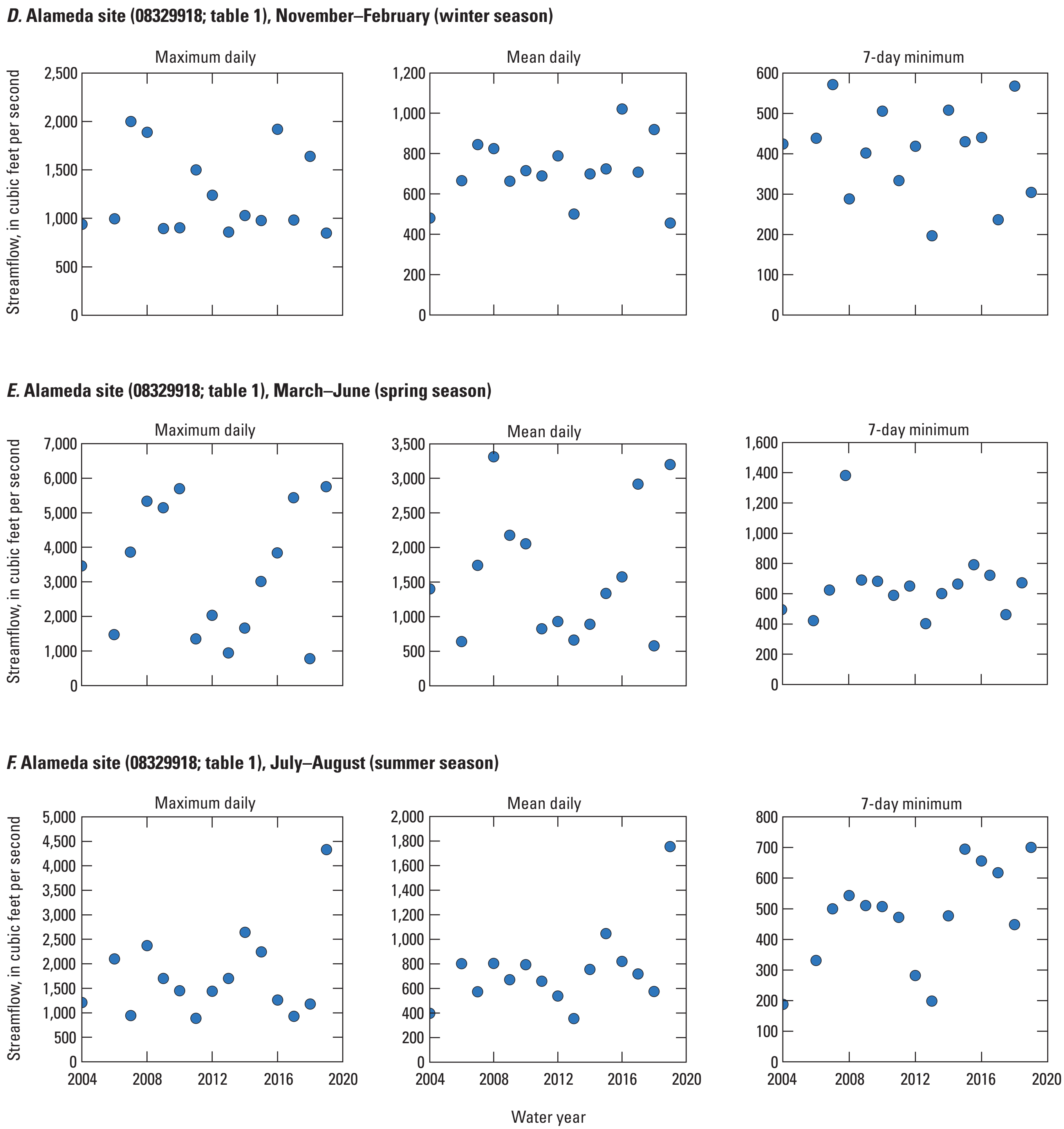

Figure 2.—Continued

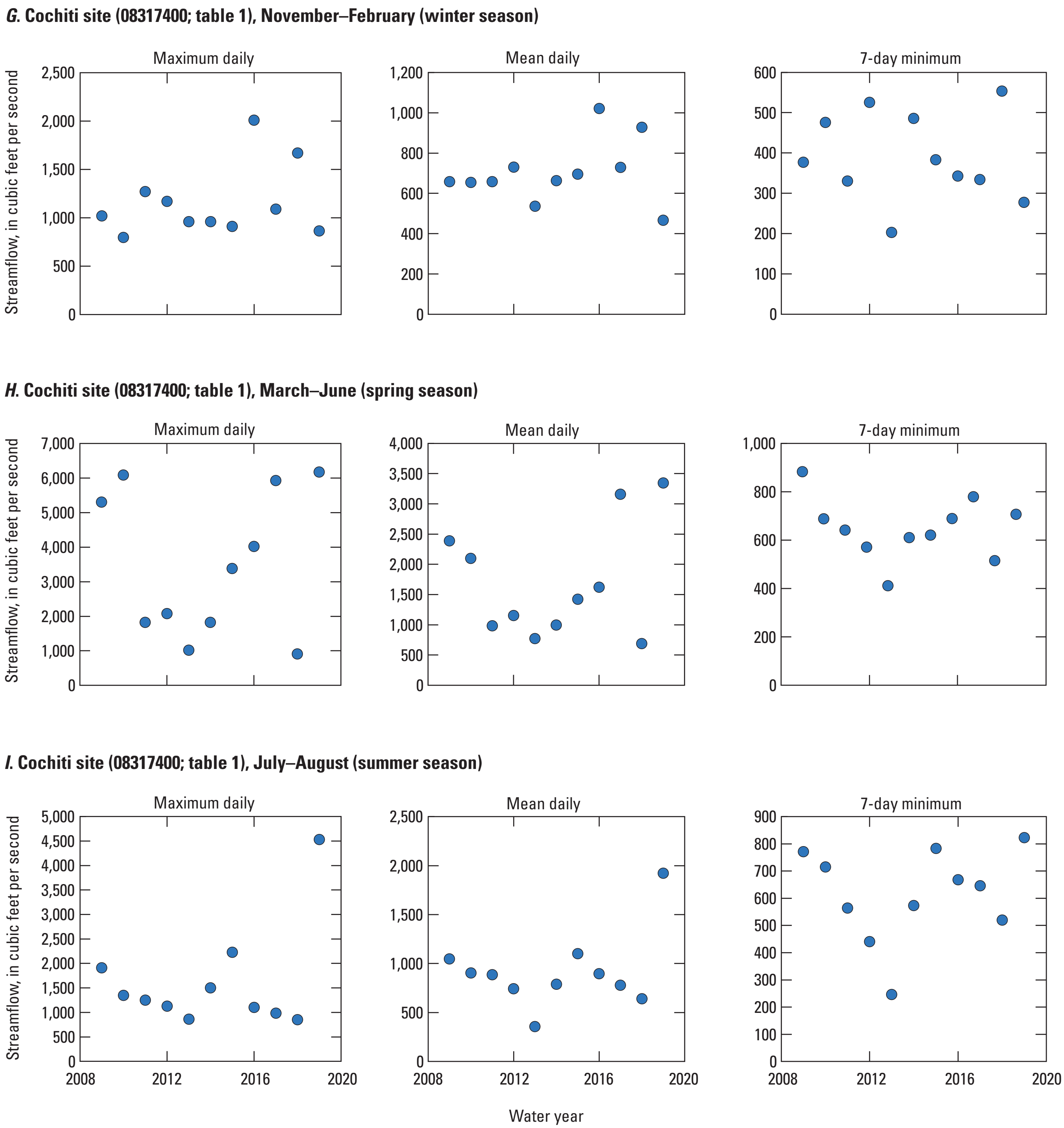

Figure 2.—Continued

Alameda

WRTDS models that had sufficient model performance indicated downward trends for most constituents (table 5; fig. 3). The only constituent with no concentration trend was sulfate, and the only constituent with no trend in flux was phosphorus. Models for most trace elements, both radiochemicals, several nutrients, dissolved solids, and suspended sediment had inadequate model performance; therefore, the results were not reported. All constituents with downward trends in concentration have changes in FN-modeled annual means that are greater than 5 percent, except for alkalinity and barium (table 5).

Very few constituents at Alameda had significant trends when using the non-flow-adjusted SKT test (table 6). Detected trends include upward trends in aluminum and antimony and a downward trend in Kjeldahl nitrogen that has a non-flow-adjusted change per year of −3.50 percent. No flow-adjusted results were reported for constituents at Alameda because flow adjustments were either considered inappropriate for the censored nature of the dataset or the fit of the flow adjustment was not adequate.

Gross alpha particles are listed as an impairment for the reach of the Rio Grande that begins at the Highway 550 bridge and ends at the Alameda site (NMED, 2020). Alpha radioactivity was not analyzed for trends, but the concentrations appear to be consistent during the study period (fig. 4).

The bacteria data for Alameda were plotted by date for a visual determination of general changes in concentration (fig. 5). A change in sampling method occurred during the study period. For fecal coliforms, the filter size decreased from 0.7 to 0.45 micrometer, and for *E. coli*, the agar changed from m-TEC to modified m-TEC. The concentrations of these two constituents did not significantly change, with the potential exception of fecal coliforms in the summer, where the concentrations measured by using the first sampling method had a larger range than those obtained by using the smaller filter. Fecal coliforms concentrations were relatively consistent during the study period for the three seasons sampled. *E. coli* concentrations were stable during spring and summer, but potentially decreased over the study period during the winter. *E. coli* is listed as a known impairment for the reach of the Rio Grande that begins at the Highway 550 bridge and ends at the Alameda location (NMED, 2020).

Table 5. Water-quality trends obtained by using the WRTDS model for constituents at the Alameda site (Rio Grande at Alameda Bridge at Alameda, New Mexico, 08329918), 2006–19.

[WRTDS, Weighted Regressions on Time, Discharge, and Season; FN, flow normalized; --, model results not reported due to model nonconvergence, model performance, or a flux bias statistic greater than 0.2]

Constituent	FN concentration trend	FN flux trend	Modeled change in FN annual concentration for full period of record, in percent
Major ions and dissolved solids			
Alkalinity	Down[1]	Down[1]	−1.92
Calcium	Down[1]	--	−10.86
Chloride	Down[1]	--	−17.18
Fluoride	Down[1]	Down[1]	−16.15
Hardness	Down[1]	Down[1]	−10.24
Magnesium	Down[1]	--	−7.61
Potassium	Down[1]	Down[1]	−11.66
Sodium	Down[1]	Down[1]	−21.85
Sulfate	none	--	−18.52
Trace elements			
Barium	Down[1]	--	−0.47
Nutrients			
Kjeldahl nitrogen	Down[1]	Down	−32.56
Organic carbon	Down[1]	--	−20.43
Phosphorus	Down[1]	None	−8.45

[1]Significant trend (likelihood greater than 0.7).

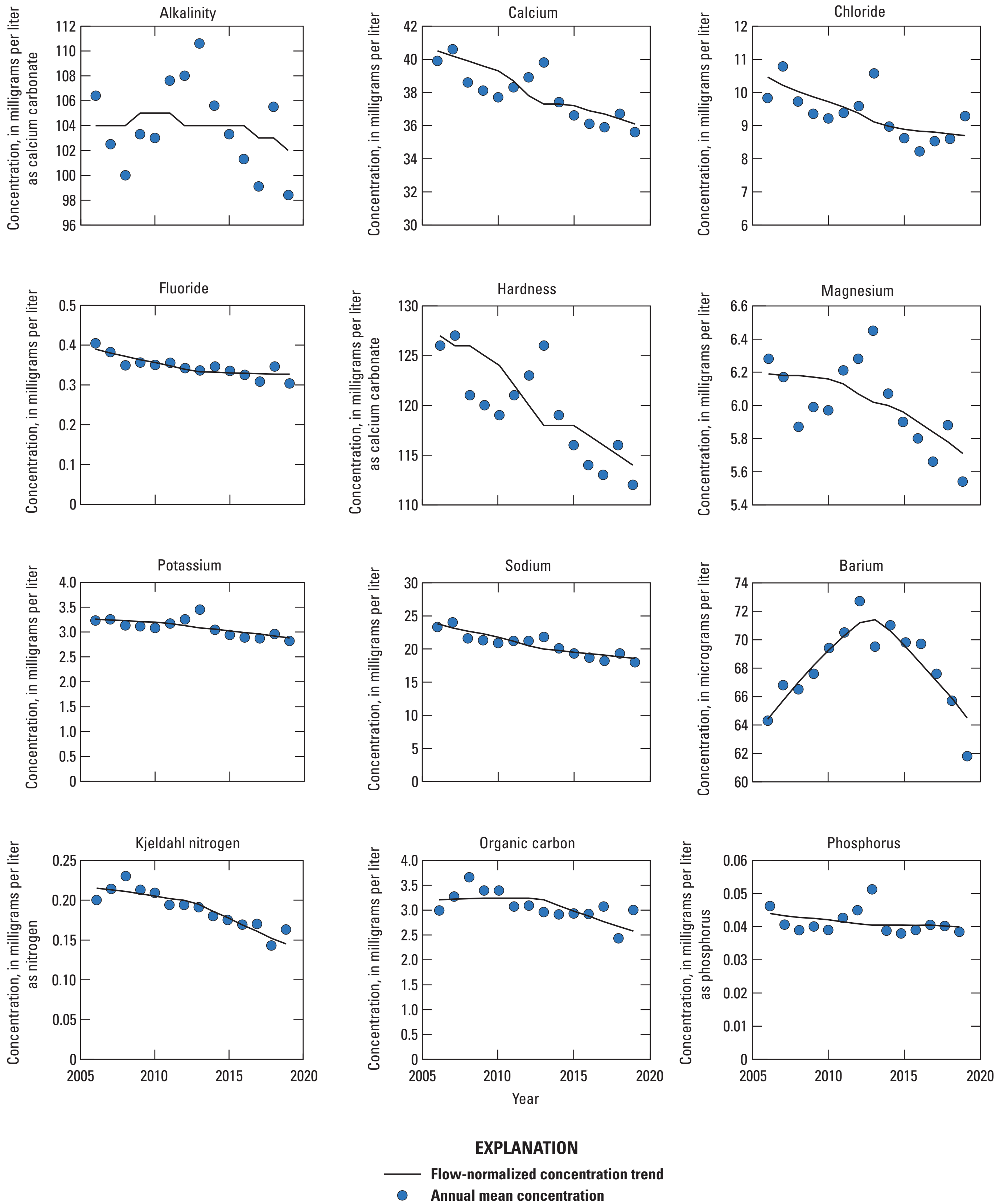

Figure 3. Modeled annual mean concentrations and flow-normalized trend lines for the constituents in surface-water samples from the Alameda site (Rio Grande at Alameda Bridge at Alameda, New Mexico, 08329918) that were determined to have trends by using the Weighted Regressions on Time, Discharge, and Season model.

Table 6. Water-quality trends obtained by using the SKT test for constituents at the Alameda site (Rio Grande at Alameda Bridge at Alameda, New Mexico, 08329918), 2004–19.

[SKT, Seasonal Kendall Tau; FA, flow adjusted; --, test results not reported due to poor flow-adjustment fit; *, trend change not reported when greater than 5 percent of values are censored]

Constituent	Non-FA trend change per year, in percent	Non-FA trend direction	FA trend change per year, in percent	FA trend direction
Major ions and dissolved solids				
Alkalinity	0.16	None	--	--
Calcium	−0.25	None	--	--
Chloride	−0.39	None	--	--
Dissolved solids	0.15	None	--	--
Fluoride	−0.98	None	--	--
Hardness	−0.13	None	--	--
Magnesium	0.15	None	--	--
Potassium	−0.44	None	--	--
Sodium	−0.69	None	--	--
Sulfate	0.70	None	--	--
Suspended sediment	−2.23	None	--	--
Trace elements				
Aluminum	*	Up[1]	--	--
Antimony	*	Up[1]	--	--
Arsenic	*	None	--	--
Barium	*	None	--	--
Boron	*	None	--	--
Cobalt	*	None	--	--
Lead	*	None	--	--
Manganese	*	None	--	--
Molybdenum	*	None	--	--
Nickel	*	None	--	--
Selenium	*	None	--	--
Uranium	*	None	--	--
Vanadium	*	None	--	--
Nutrients				
Kjeldahl nitrogen	−3.50	--	Down[1]	--
Nitrate	*	None	--	--
Nitrite plus nitrate	*	None	--	--
Organic carbon	−1.24	None	--	--

[1]Significant trend (p-value less than 0.05).

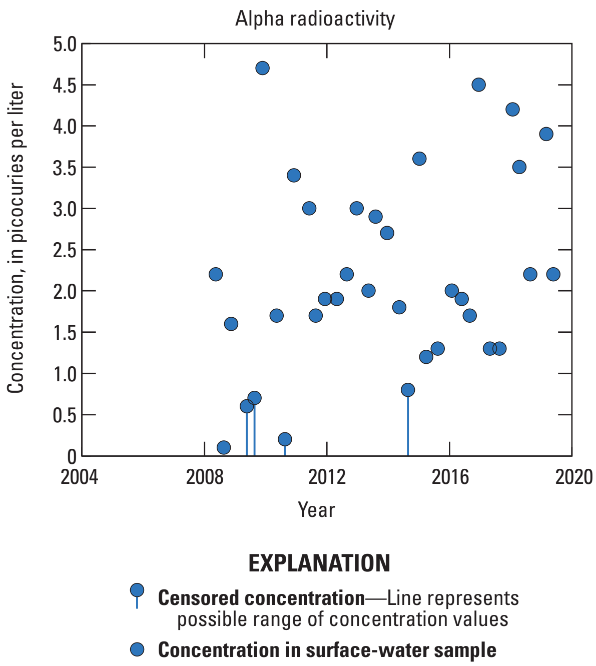

Figure 4. Alpha radioactivity concentrations in surface-water samples from the Alameda site (Rio Grande at Alameda Bridge at Alameda, New Mexico, 08329918).

Most of the organic constituents analyzed were not detected in the samples from the Alameda site (appendix 3). Of the 211 organic constituents analyzed for at the Alameda site, 117 did not return any noncensored concentrations greater than the laboratory reporting level. Five or more noncensored concentrations greater than the laboratory reporting level of the following constituents occurred at Alameda: DEET, *p*-cresol, triclosan, isophorone, and hexahydrohexamethyl cyclopentabenzopyran (HHCB). Of these constituents, DEET had estimated concentrations within the blank sample population; however, these estimates are less than the listed laboratory reporting level and may be an artifact of laboratory analysis, meaning the potential bias in the environmental sample population is likely minimal. Phenol had 1 detection above laboratory reporting levels; however, only 1 out of 10 phenol blanks had detections of any sort, indicating that the potential bias is likely not widespread within the environmental sample population. The numbers of detections were not sufficient to determine trends for these constituents. Time series of these concentrations were plotted (fig. 6), but no patterns were visible in the timing or magnitude of the detected, noncensored concentrations at each site. The concentrations were also plotted against streamflows, but there was no correlation between detections of the organic constituents and streamflows.

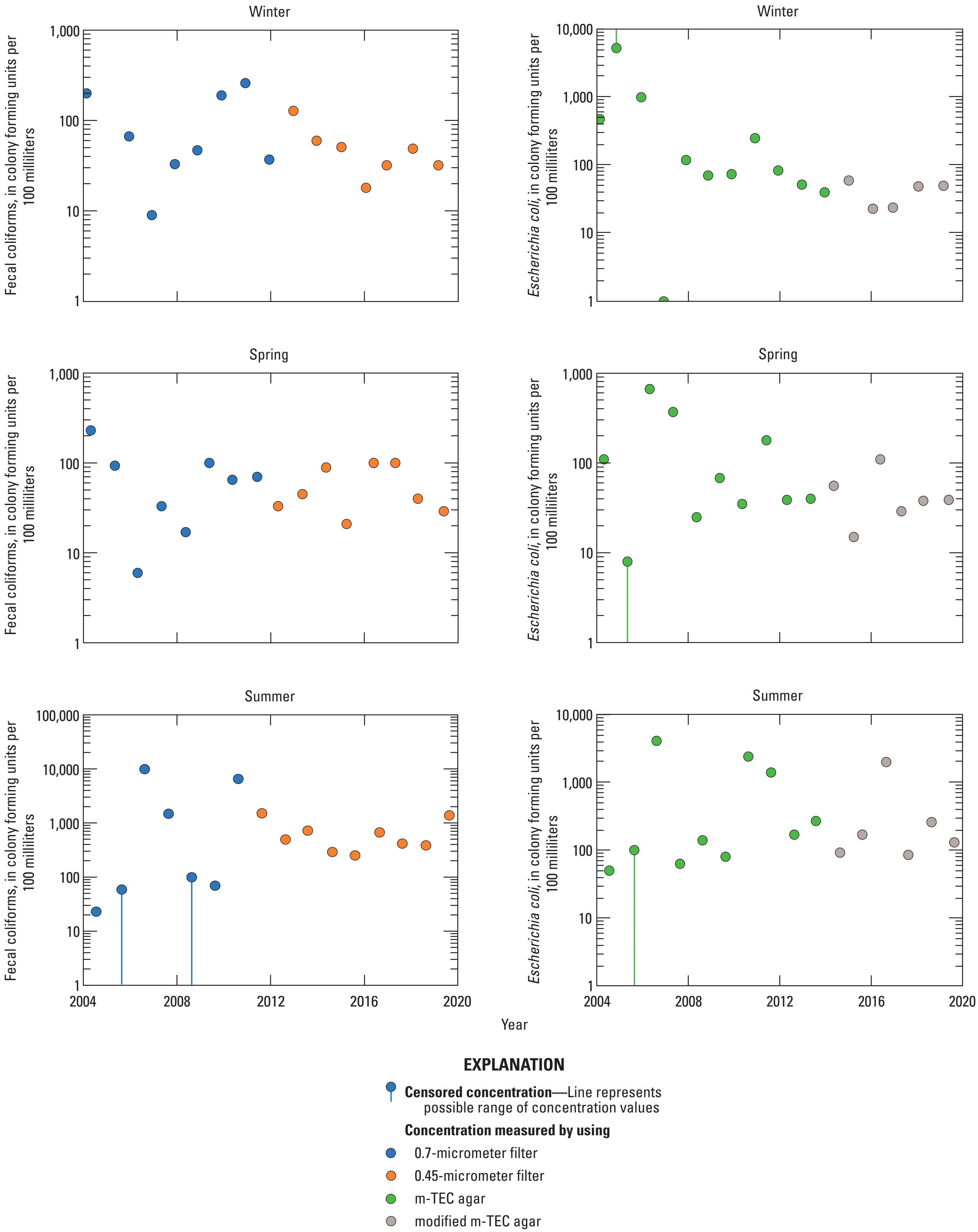

Figure 5. Fecal coliforms and *Escherichia coli* concentrations in surface-water samples from the Alameda site (Rio Grande at Alameda Bridge at Alameda, New Mexico, 08329918), by season.

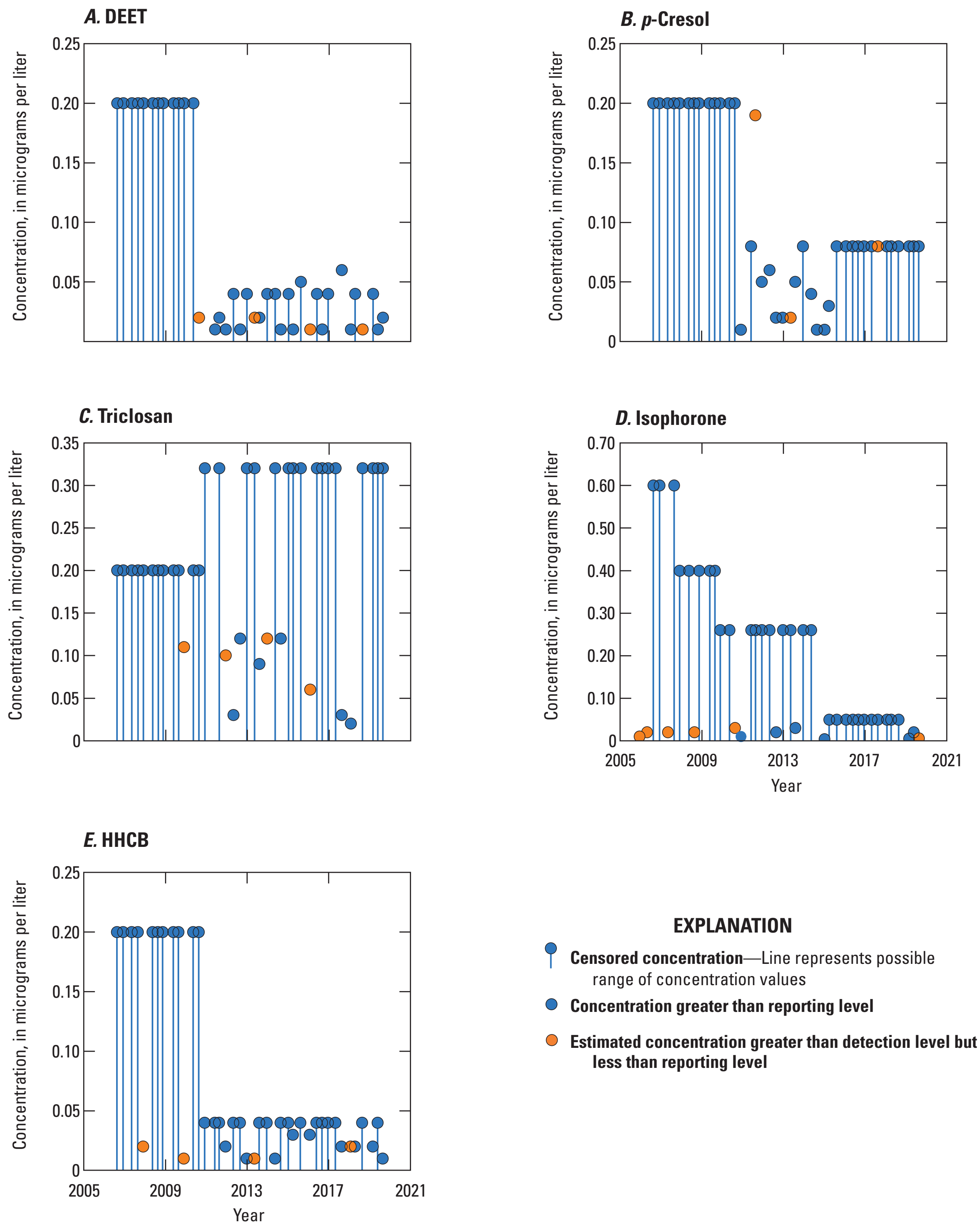

Figure 6. Concentrations of the following organic constituents in surface-water samples from the Alameda site (Rio Grande at Alameda Bridge at Alameda, New Mexico, 08329918): *A*, *N*,*N*-diethyl-*meta*-toluamide (DEET), *B*, *p*-cresol, *C*, triclosan, *D*, isophorone, and *E*, hexahydrohexamethyl cyclopentabenzopyran (HHCB).

Cochiti

No constituents in the major ions or dissolved solids group had trends detected by the SKT test (table 7). The only trace element with a detected trend was cobalt, with a detected downward trend and non-flow-adjusted trend change per year of −15.91 percent (table 7). More trends were detected for the nutrient group, with a downward trend in Kjeldahl nitrogen and upward trends in nitrate and nitrate plus nitrite (table 7). Concentrations of Kjeldahl nitrogen appear to only have a small change over time (with a non-flow-adjusted trend change per year of −3.20 percent), and the concentrations of nitrate and nitrate plus nitrite have larger changes over time (with non-flow-adjusted trend changes of 9.71 and 8.58 percent, respectively; table 7).

Gross alpha particles are listed as an impairment for the reach of the Rio Grande that begins at Cochiti Lake and ends at the Angostura Diversion Dam (approximately 1.5 miles

Table 7. Water-quality trend SKT results for constituents at the Cochiti site (Rio Grande below Cochiti Dam, New Mexico, 08317400), 2009–19.

[SKT, Seasonal Kendall Tau; FA, flow adjusted; --, test results not reported due to poor flow-adjustment fit; *, trend change not reported when greater than 5 percent of values are censored]

Constituent	Non-FA trend change per year, in percent	Non-FA trend direction	FA trend change per year, in percent	FA trend direction
Major ions and dissolved solids				
Alkalinity	−1.34	None	--	--
Calcium	−0.73	None	--	--
Chloride	0.20	None	--	--
Dissolved solids	−0.65	None	--	--
Fluoride	−1.17	None	--	--
Hardness	−0.63	None	--	--
Magnesium	−0.59	None	--	--
Potassium	−0.62	None	--	--
Sodium	−0.28	None	--	--
Sulfate	−0.53	None	--	--
Trace elements				
Aluminum	4.33	None	--	--
Antimony	−0.14	None	--	--
Barium	−2.14	None	--	--
Boron	−2.13	None	--	--
Cobalt	−15.91	Down[1]	--	--
Copper	*	None	--	--
Lead	*	None	--	--
Manganese	−3.23	None	--	--
Molybdenum	−0.82	None	--	--
Nickel	0.33	None	--	--
Selenium	−0.66	None	--	--
Uranium	−0.50	None	--	--
Nutrients				
Kjeldahl nitrogen	−3.20	Down[1]	--	--
Nitrate	9.71	Up[1]	--	--
Nitrite	*	None	--	--
Nitrate plus nitrite	8.58	Up[1]	--	--
Organic carbon	−2.15	None	--	--
Phosphorus	−1.60	None	--	--

[1]Significant trend (*p*-value less than 0.05).

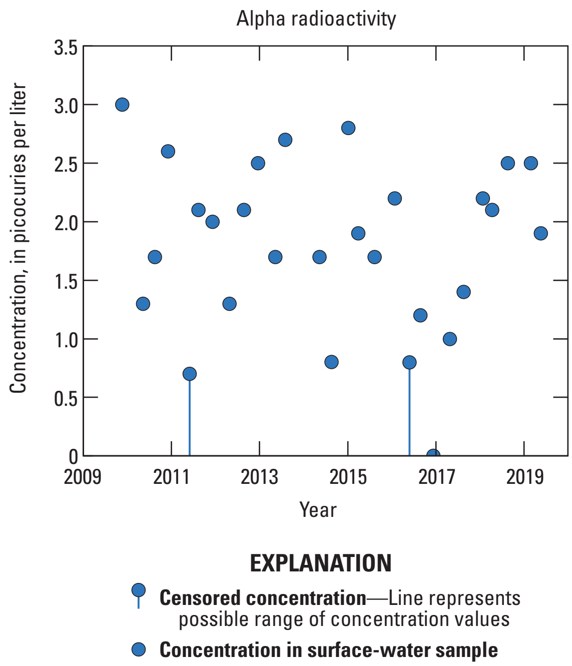

Figure 7. Alpha radioactivity concentrations in surface-water samples from the Cochiti site (Rio Grande below Cochiti Dam, New Mexico, 08317400).

upstream from where the Jemez River meets the Rio Grande; fig. 1) (NMED, 2020). Alpha radioactivity was not analyzed for trends, but the concentrations appear to be consistent during the sample period (fig. 7).

The bacteria data for Cochiti were plotted by date for a visual determination of general changes in concentration (fig. 8). The samples collected by using the first sampling method for fecal coliforms are one to two magnitudes lower than the maximum concentrations detected in samples collected by using the second sampling method; however, only seven samples were collected by using the first sampling method. Fecal coliforms concentrations did not consistently change during the study period when the second sampling method was used. *E. coli* concentrations did not consistently change between sampling methods nor over time.

Most of the organic constituents analyzed were not detected in the samples from the Cochiti site (appendix 3). Of the 197 organic constituents analyzed for at the Cochiti site, 178 did not return any noncensored concentrations greater than the laboratory reporting level. Five or more noncensored concentrations above the laboratory reporting level of the following constituents occurred at Cochiti: DEET and triclosan. Of these constituents, DEET had estimated concentrations within the blank sample population; however, these estimates are less than the listed laboratory reporting level and may be an artifact of laboratory analysis, meaning the potential bias in the environmental sample population is likely minimal. The numbers of detections were not sufficient to determine trends for these constituents. Time series of these concentrations were plotted (fig. 9), but no patterns were visible in the timing or magnitude of the detected, noncensored concentrations at each site. There was also no correlation in a comparison of concentrations to streamflows.

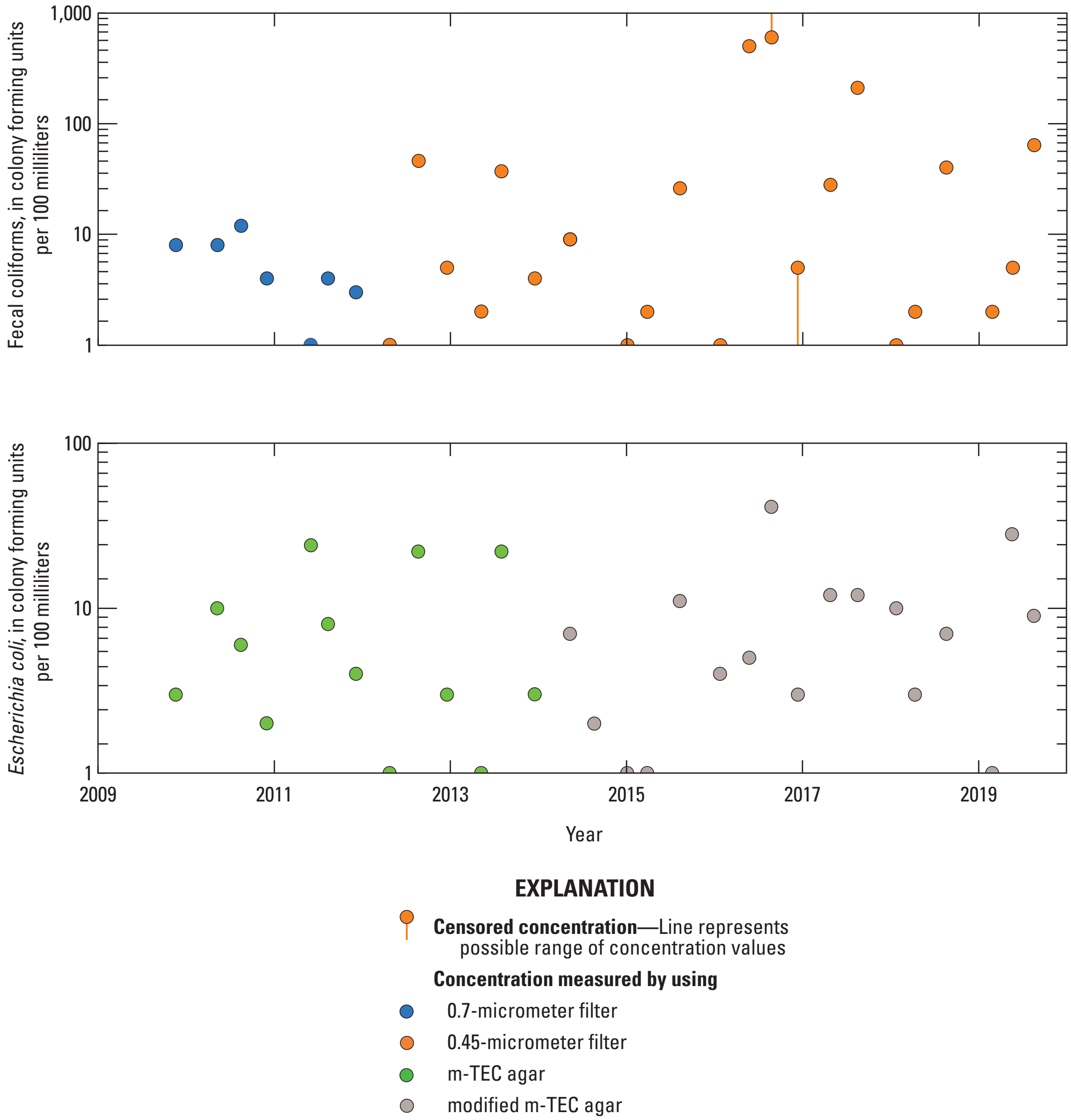

Figure 8. Fecal coliforms and *Escherichia coli* concentrations in surface-water samples from the Cochiti site (Rio Grande below Cochiti Dam, New Mexico, 08317400).

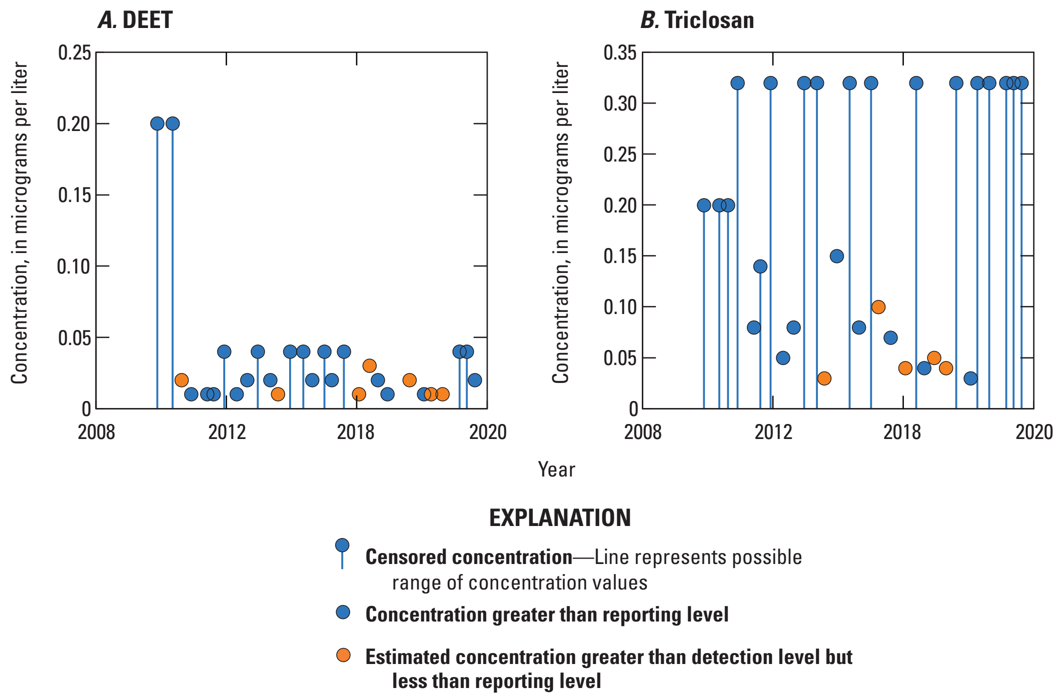

Figure 9. Concentrations of *A*, *N,N*-diethyl-*meta*-toluamide (DEET) and *B*, triclosan in surface-water samples from the Cochiti site (Rio Grande below Cochiti Dam, New Mexico, 08317400).

Jemez Canyon Dam

Most of the trace elements and nutrients at Jemez Canyon Dam have greater than 5 percent censored data, so most of those constituents do not have reported change percentages. These constituents do, however, have trend direction results. The only two spring-season trends detected at Jemez Canyon Dam are an upward flow-adjusted trend in potassium and an upward trend in antimony (table 8). The flow-adjusted trend change per year for potassium is only 2.21 percent.

The maximum concentration of arsenic at Jemez Canyon Dam, 58.5 µg/L is greater than the drinking water maximum contaminant level (MCL) of 10 µg/L (appendix 4; EPA, 2020). Of the 15 samples analyzed for arsenic at Jemez Canyon Dam, 13 of those exceeded the MCL of 10 µg/L (fig. 10).

The maximum concentration of aluminum at Jemez Canyon Dam, 202 µg/L, is greater than the drinking water secondary maximum contaminant level (SMCL) of 200 µg/L (appendix 4; EPA, 2020); this was the only occurrence of aluminum concentration greater than the SMCL out of the 15 samples analyzed for aluminum at Jemez Canyon Dam (fig. 10). Of the 15 samples analyzed for dissolved solids at Jemez Canyon Dam, 4 had concentrations greater than the SMCL of 500 mg/L (fig. 10; EPA, 2020), with a maximum concentration of 838 mg/L (appendix 4).

The bacteria data for Jemez Canyon Dam were plotted by date for a visual determination of general changes in concentration (fig. 11). No consistent changes over time were evident for either fecal coliforms or *E. coli*.

Table 8. Water-quality trend SKT results for constituents at the Jemez Canyon Dam site (Jemez River below Jemez Canyon Dam, New Mexico, 08329000, and Jemez River outlet below Jemez Canyon Dam, New Mexico, 08328950), 2004–19.

[SKT, Seasonal Kendall Tau; FA, flow adjusted; --, test results not reported due to poor flow-adjustment fit; *, trend change not reported when greater than 5 percent of values are censored]

Constituent	Non-FA trend change per year, in percent	Non-FA trend direction	FA trend change per year, in percent	FA trend direction
Major ions and dissolved solids				
Alkalinity	--	--	0.20	None
Calcium	--	--	−0.69	None
Chloride	--	--	0.87	None
Fluoride	--	--	1.00	None
Magnesium	--	--	−0.24	None
Potassium	--	--	2.21	Up[1]
Sodium	--	--	0.67	None
Sulfate	--	--	0.29	None
Suspended sediment	5.90	None	--	--
Trace elements				
Aluminum	*	None	--	--
Antimony	*	Up[1]	--	--
Arsenic	*	None	--	--
Barium	*	None	--	--
Boron	*	None	--	--
Cobalt	*	None	--	--
Copper	*	None	--	--
Manganese	*	None	--	--
Molybdenum	*	None	--	--
Nickel	*	None	--	--
Uranium	*	None	--	--
Nutrients				
Kjeldahl nitrogen	*	--	None	--
Organic carbon	--	--	2.45	None
Phosphorus	4.06	None	--	--

[1]Significant trend (*p*-value less than 0.05).

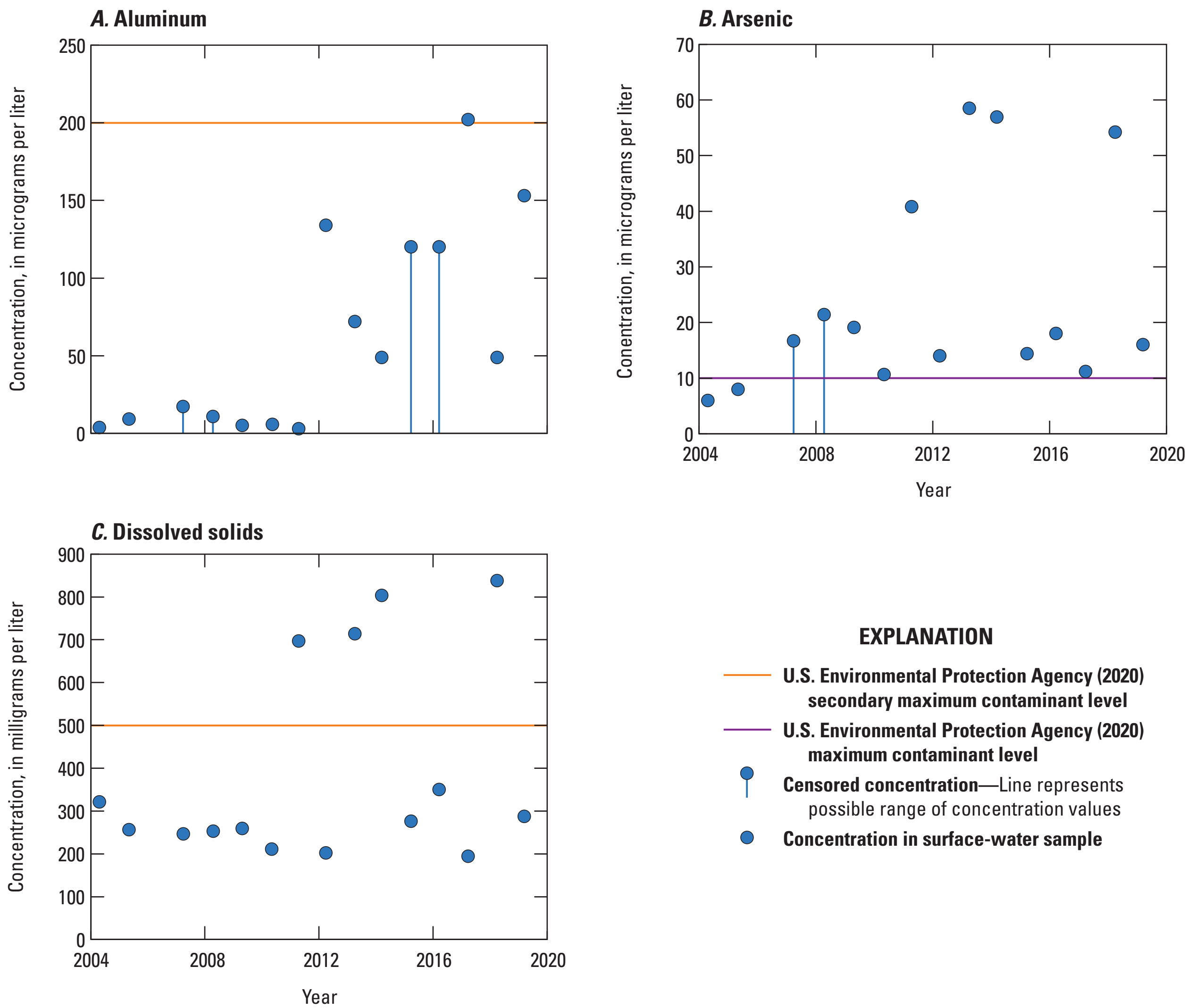

Figure 10. Spring-season concentrations of *A*, aluminum, *B*, arsenic, and *C*, dissolved solids used for the Seasonal Kendall Tau tests for the Jemez Canyon Dam site (Jemez River below Jemez Canyon Dam, New Mexico, 08329000, and Jemez River outlet below Jemez Canyon Dam, New Mexico, 08328950).

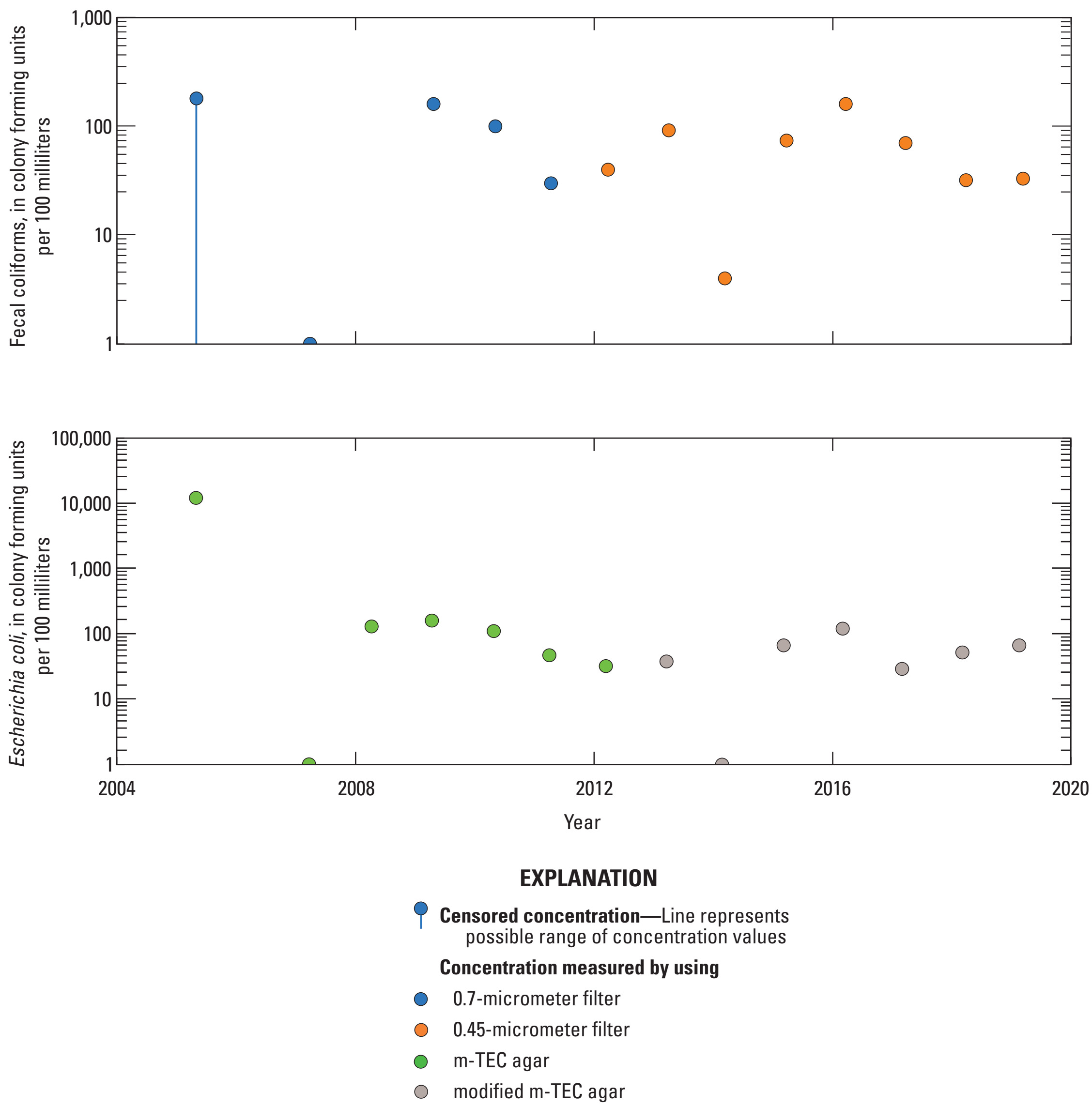

Figure 11. Fecal coliforms and *Escherichia coli* concentrations in surface-water samples from the Jemez Canyon Dam site (Jemez River below Jemez Canyon Dam, New Mexico, 08329000, and Jemez River outlet below Jemez Canyon Dam, New Mexico, 08328950).

Discussion

Analysis of blank concentrations indicates that most of the constituents, with the exception of organic carbon and cobalt, used in the trend analysis portion of this study were not subject to widespread high bias in concentrations. Organic carbon and cobalt were determined to have high bias because their maximum blank sample detection was greater than their blank maximum laboratory reporting level; more than 25 percent of the blank samples recorded detections, and almost all (more than 97 percent) of their environmental samples were measured at concentrations less than the 10x threshold (10 times the maximum blank concentration). The 80th percentile UCLs for organic carbon and cobalt are 67 and 79 percent, respectively, meaning that the confidence that the maximum blank concentration (0.1947 µg/L for cobalt, 4.188 mg/L for organic carbon) detected for these constituents captures the 80th percentile of maximum contamination is 67 and 79 percent (table 2). SKT test results indicate that cobalt concentrations had a downward trend at Cochiti and no trend at Alameda or Jemez Canyon (spring season only). The WRTDS model results indicate that organic carbon concentrations had a downward trend at Alameda; the SKT test results indicate no trend at Alameda, Cochiti, or Jemez Canyon Dam (spring season only). These trends, or lack of, are not necessarily invalidated, but they do come with the caveat that the concentrations are generally biased high and that the extent of the bias may be variable over time.

Analysis of replicates gives additional context to the variability of the sample population which is important for the interpretation of the trend analyses results. Constituents with higher mean relative standard deviations (RSDs), defined as the ratio (in percent) of the standard deviation to mean concentration, may increase uncertainty about the validation of the trend. Trends were not particularly evaluated to be invalidated with the replicate datasets; however, constituents such as cobalt and copper had RSDs greater than 25 percent (appendix 1; table 4), indicating high variability in the concentrations introduced through the sample collection and analysis process. These high RSDs potentially indicate that the reported concentrations have a variability too high to confirm trends. These trends are still reported with the caveat that they are fairly uncertain. The replicate sample size was small, and the RSD has the potential to decrease if additional replicate samples were collected.

Streamflow is extremely variable annually and seasonally at the long-term sites (fig. 2). There were periods of increasing and decreasing annual flow metrics within the periods of study, but streamflow did not increase or decrease during the entire periods of record for each site. Although the periods of record are too short to be used to determine trends in streamflow, the lack of consistent increase or decrease in streamflow indicates that trends in concentrations and fluxes are not primarily driven by systematic changes in streamflow. Streamflow metrics not considered during this study, such as timing of high and low flows, also have the potential to affect concentrations. Both the WRTDS model, which is a flow-normalized trend analysis method, and the non-flow-adjusted SKT test were used to determine trends in water quality at the Alameda site, but some constituents had differing results between the two methods. Of the 11 constituents that had reported results for both methods, 10 constituents had trend results that differed in either trend direction or significance (tables 5 and 6). This indicates that changes in streamflow at the Alameda site influence the concentrations of the constituents analyzed. Of the six constituents with trend results for concentration and flux at Alameda, five of the constituents have consistent concentration and flux trend results, with either significant trends of the same direction for concentration and flux results of a constituent or no likely trend for concentration and flux results of a constituent (table 5). The flow-adjusted trends for concentration and the flux trends were in the same direction, indicating that changes in concentration were not affected by a systematic change in streamflow that counter-acted the changes in concentration.

The SKT tests for all constituents at Alameda, all constituents at Cochiti, and some constituents at Jemez Canyon Dam did not have satisfactory fits for flow adjustment, so the non-flow-adjusted results were reported. Using the Alameda dataset with WRTDS had several shortcomings, including length of and density of data, sampling during only three of the four seasons, and dam influence on streamflows. The shortcomings with the SKT tests at Alameda include unsatisfactory fits for the flow-adjusted models, so the non-flow-adjusted results were reported instead. Flow-adjusted results are preferred because they remove the year-to-year noise, and non-flow-adjusted trend results may be influenced by trends in streamflow. Given the shortcomings for both methods, it is difficult to consider one model more appropriate to use than the other.

All reported WRTDS model results for the major ions, aside from sulfate, at Alameda indicated downward trends (table 5). The modeled annual mean concentrations of most major ions peaked during water year 2013 (fig. 3). The lowest maximum daily and mean daily streamflows occurred at Alameda during water year 2013 (fig. 2), so it is expected that concentrations would be higher for any constituent that has a negative relationship between streamflow and concentration. Alkalinity at Alameda was stable over the trend period. Although the WRTDS trend result for alkalinity is downward, the change between the 2006 and 2019 FN concentrations is minimal (−1.92 percent; table 5). Similarly, the WRTDS model results for barium, the only reported trace element result at Alameda, indicate an increase in the first half of the model period followed by a decrease (fig. 3) and very little difference between the 2006 and 2019 FN concentration (−0.47 percent; table 5), so barium can also be considered stable. All the other constituents have changes in trend magnitude of greater than 5 percent.

SKT results indicated few trends for Alameda and Cochiti (tables 6 and 7) and few spring-season trends for Jemez Canyon Dam (table 8). The SKT results indicated upward trends in aluminum and antimony at Alameda. Cobalt had a downward trend at Cochiti, and although concentrations did decrease substantially around 2015, the blank analysis indicates that cobalt was generally biased high and that the extent of the bias may be variable over time; therefore, uncertainty is associated with this trend. SKT results also indicated upward trends in nitrate and plus nitrite at Cochiti (table 7). Downward trends in Kjeldahl nitrogen occurred at both Alameda and Cochiti, although it should be noted that the non-flow-adjusted trend change per year was less than 5 percent at both sites (−3.50 and −3.20, respectively), indicating that although the SKT test indicated significant downward trends, they may not actually affect changes in the concentrations. The only spring-season trends determined by using SKT at the Jemez Canyon Dam site were an upward trend in potassium and antimony (table 8). When the SKT test was used to analyze flow-corrected monthly means of water-quality data from 1975 to 1999, dissolved solids, sodium, magnesium, potassium, chloride, and fluoride had downward trends in the reaches downstream from Cochiti and through Albuquerque (Passell and others, 2004). Note that the sites used in the Passel and others study and the sites used in this study were not the same; the sites in this study generally were upstream from the larger changes in constituents. Although the results from the Passel and others study are not directly comparable to the results from this study because of the use of monthly means, a greater use of flow correction, and a difference in sampling locations, this study did not find any evidence for the continuation of the trends in the Passel and others study. Another study of trends in nutrients along the Rio Grande (Passell and others, 2005) used monthly medians for SKT analysis, but the only constituents analyzed at a comparable site, San Felipe (between the Cochiti and Alameda sites, fig. 1), were for total constituents, not dissolved constituents as used in this study; therefore, comparisons between the two studies are not appropriate. The lack of trends in the constituents analyzed in this study could indicate that water quality in the Basin has stabilized despite the population growths of about 20 percent in the major metropolitan areas (U.S. Census Bureau, 2003, 2019). However, the sampling density may not be large enough to adequately characterize the range of concentration variability with streamflow, especially given the evidence that the concentrations are influenced by changes in streamflow at Alameda.

Comparing the concentration and trend results to water-quality criteria is important to understand the relevance of the results. *E. coli* is listed as a known impairment for the reach of the Rio Grande that begins at the Highway 550 bridge and ends at the Alameda site (NMED, 2020). *E. coli* concentrations appear stable during spring and summer, but potentially decrease in the winter. Streamflow statistics did not consistently change at Alameda (fig. 2), so a potential decrease of *E. coli* concentrations during the winter season could be a result of a change in management, land use, or other driving factors such as public education and outreach on stormwater impacts that the City of Albuquerque has implemented as part of its stormwater management program (City of Albuquerque, 2020). Another possibility is that a change in a winter streamflow statistic that could be driving this potential decrease in winter *E. coli* measurements was not considered during this study. Additionally, with the high variability of *E. coli*, three samples per year is not adequate to accurately characterize the relationship between concentration and streamflow, so no conclusions can be confidently drawn.

Gross alpha particles are listed as an impairment for reaches of the Rio Grande that include the Alameda and Cochiti sites. The SKT results and plotting of concentrations showed either no trend or no overall change in alpha radioactivity at the impaired locations. Aluminum and selenium are listed as impairments at the Cochiti site, but the SKT test indicated no trends for either constituent.

The comparison of constituents' concentrations to drinking water standards is meant only to give context to the trend results because the water being sampled is untreated water directly from the streams. Of the three sites, only the Jemez Canyon Dam site had concentrations that exceeded the standards. The Jemez River has much smaller streamflow rates than the Rio Grande, and the Jemez River input to the Rio Grande is diluted by the difference in streamflow. A trace element of concern is arsenic; almost all the concentrations at the Jemez Canyon Dam site exceeded the EPA drinking water MCL. Arsenic is a natural constituent of the geothermal reservoir of the Valles Caldera, which has mixed with the groundwater in the area, resulting in elevated levels of arsenic in nearby springs, wells, and surface waters (Trainer and others, 2000; Reid and others, 2003). No trends were identified for arsenic concentrations at any of the sites. Concentrations of aluminum and dissolved solids also exceeded the EPA drinking water SMCL, in one and four samples, respectively (fig. 10). Although the SKT test indicated no trend for either constituent, aluminum concentrations over time indicate an increase in aluminum concentrations, particularly since 2012, and this should be taken into consideration.

Summary

Trends were analyzed for selected constituents at three sites along the Jemez River and the Rio Grande upstream from Albuquerque, New Mexico, for differing time periods ranging from 2004 to 2019 by using the Seasonal Kendall Tau (SKT) test and the Weighted Regressions on Time, Discharge, and Season (WRTDS) model. Water-quality data at three long-term sites were used for the trend analyses in this study, with the Cochiti and Alameda sites along the Rio Grande and the Jemez Canyon Dam site along the Jemez River, a tributary of the Rio Grande.

Many constituents were not suitable for trend analysis due to inadequate periods of record or high occurrences of censored data. The data used for the trend analyses did not fully meet desired sample density within seasons. Continuation of sampling to extend the periods of record and increasing sample density could result in a more complete and robust trend analysis. The proximity of the Cochiti and Jemez Canyon Dam sites to dams is also a drawback to the analysis because it is difficult to differentiate between the influence of dam management and the influence of streamflow on water-quality trends.

Both WRTDS, a flow-normalized trend analysis, and a non-flow-adjusted SKT test were performed for constituents at the Alameda site. Most reportable WRTDS trends were downward for constituents at the Alameda site. The SKT results for constituents at the Alameda site, however, showed few trends. The difference between flow-normalized and non-flow-adjusted trend results indicates that concentrations, and thereby fluxes, are influenced by changes in streamflow at this site.

Most constituents that were analyzed for trends did not have a significant trend, indicating either that the water quality in the Middle Rio Grande Basin has been stable during the sampling period or that the sampling temporal density was not adequate to characterize the range of concentration variability with streamflow. SKT test results indicate upward trends in concentrations of aluminum and antimony at the Alameda site, nitrate and nitrate plus nitrite at the Cochiti site, and potassium and antimony during the spring season at the Jemez Canyon Dam site. SKT test results also indicate small, downward trends in Kjeldahl nitrogen at the Alameda and Cochiti sites and a downward trend in cobalt at the Cochiti site that is subject to bias in the cobalt concentrations.

Concentrations of water-quality constituents were also compared to Federal and State water-quality standards to provide context and relevance to the results. No concentrations were above the national primary or secondary drinking water standards at the Alameda and Cochiti sites, but the Jemez Canyon Dam site did have spring-season concentrations above the U.S. Environmental Protection Agency primary drinking water standard for arsenic and above the national secondary drinking water standards for dissolved solids and aluminum. *Escherichia coli* (*E. coli*) is listed as an impairment for a reach of the Rio Grande that includes the Alameda site, but seasonal data for *E. coli* indicate that concentrations are mostly stable in spring and summer, with a potential decrease in concentrations in samples collected during the winter. Gross alpha particles are listed as an impairment for reaches of the Rio Grande that include the Alameda and Cochiti sites, but the time series of concentrations showed no overall change at the impaired locations.

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