

Prepared in cooperation with the Idaho Department of Lands and Midas Gold Idaho, Inc.

Arsenic, Antimony, Mercury, and Water Temperature in Streams near Stibnite Mining Area, Central Idaho, 2011–17



Scientific Investigations Report 2019–5072

Cover: East Fork of South Fork Salmon River at Stibnite, Idaho. (Photograph taken by Austin Baldwin, U.S. Geological Survey, August 10, 2016.)

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By Austin K. Baldwin and Alexandra B. Etheridge

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James F. Reilly II, Director

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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		
acre	4,047	square meter (m ²)
acre	0.004047	square kilometer (km ²)
square mile (mi ²)	2.590	square kilometer (km ²)
Volume		
cubic yard (yd ³)	0.7646	cubic meter (m ³)
Flow rate		
cubic foot per second (ft ³ /s)	0.02832	cubic meter per second (m ³ /s)
Mass		
ton per day (ton/d)	0.9072	metric ton per day
ton per year (ton/yr)	0.9072	metric ton per year

Conversion Factors

International System of Units to U.S. customary units

Multiply	By	To obtain
Length		
millimeter (mm)	0.03937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
meter (m)	1.094	yard (yd)
Volume		
milliliter (mL)	0.033814	ounce, fluid (fl. oz)
liter (L)	0.2642	gallon (gal)

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

$$^{\circ}\text{F}=(1.8\times^{\circ}\text{C})+32$$

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius ($\mu\text{S}/\text{cm}$ at 25°C).

Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L) or micrograms per liter ($\mu\text{g}/\text{L}$).

Datums

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 feet above NAVD 88.

Abbreviations

BCF	bias correction factor
BFI	base-flow index
CCC	criterion continuous concentration, or “chronic” aquatic-life criterion
CMC	criterion maximum concentration, or “acute” aquatic-life criterion
EF1	East Fork of South Fork Salmon River above Meadow Creek, near Stibnite, Idaho
EF2	East Fork of South Fork Salmon River at Stibnite, Idaho
EF3	East Fork of South Fork Salmon River above Sugar Creek, near Stibnite, Idaho
EFSFSR	East Fork of South Fork of the Salmon River
HHB	human-health based criterion
MDAT-CW	maximum daily average water-temperature criterion for cold-water aquatic life
MDAT-SS	maximum daily average water-temperature criterion for salmonid spawning
MDMT-SS	maximum daily maximum water-temperature criterion for salmonid spawning
MSPE	model standard percentage error
MWMT-BT	maximum weekly maximum water-temperature criterion for bull trout habitat
NWQL	National Water Quality Laboratory
QA/QC	quality-assurance/quality-control
QBFI	streamflow weighted by base-flow index
R ²	coefficient of determination
RPD	relative percent difference
SWE	snow-water equivalent
USGS	U.S. Geological Survey

Arsenic, Antimony, Mercury, and Water Temperature in Streams near Stibnite Mining Area, Central Idaho, 2011–17

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Abstract

Mineralization and historical mining of stibnite (antimony sulfide), tungsten, gold, silver, and mercury in the headwaters of the East Fork of the South Fork Salmon River (EFSFSR) near the former town of Stibnite in central Idaho resulted in water-quality impairments related to mercury, antimony, and arsenic. Additionally, mining-related disturbances and wildfires have resulted in a lack of riparian shade in some areas, likely impacting water temperatures. In 2011, the U.S. Geological Survey, in cooperation with Midas Gold Corporation and the Idaho Department of Lands, began a study to characterize the spatial and temporal occurrence of trace metals to the EFSFSR. Five sites on the EFSFSR and its tributaries (Meadow and Sugar Creeks) were sampled about six times annually during 2011–17, during a range of streamflow conditions, for a total of 36–40 samples per location. Continuous water temperature, specific conductance, and streamflow also were measured at each site. The purpose of this report is to update previously reported information related to arsenic, antimony, mercury, and water temperature.

Concentrations of dissolved arsenic and antimony generally increased from upstream to downstream in the EFSFSR. At the upstream site, upstream of the Meadow Creek confluence, dissolved arsenic and antimony concentrations averaged 8.86 and 0.93 micrograms per liter ($\mu\text{g/L}$), respectively. Downstream, upstream from the Sugar Creek confluence, average dissolved concentrations increased to 56.5 and 27.9 $\mu\text{g/L}$, respectively. All samples from the downstream EFSFSR site exceeded the human-health based criterion for both dissolved arsenic (10 $\mu\text{g/L}$) and dissolved antimony (5.6 $\mu\text{g/L}$). The chronic aquatic life criterion for dissolved arsenic (150 $\mu\text{g/L}$) was not exceeded (the maximum sample concentration was 108 $\mu\text{g/L}$), and aquatic life criteria for antimony have not been established. The highest concentrations of both dissolved arsenic and dissolved antimony occurred during low-flow periods (July–March), suggesting the constituents are present in groundwater. In contrast, total mercury concentrations were highest during high-flow periods (April–June) and were particulate-associated, suggesting that mercury is present in surface materials. At Sugar Creek, where the highest total mercury concentrations were measured, 97 percent of samples exceeded the chronic aquatic life criterion (0.012 $\mu\text{g/L}$) and

11 percent exceeded the acute criterion (2.1 $\mu\text{g/L}$). At all sites, summertime water temperatures frequently exceeded criteria related to salmonid spawning.

Surrogate models previously developed to estimate continuous concentrations of arsenic, antimony, and mercury were reevaluated and updated, and the importance of explanatory variables on constituent concentrations is discussed. Results from this study can help guide future remediation locations and strategies, and provide a baseline against which future changes can be measured.

Introduction

The Stibnite mining area (study area) is in Boise and Payette National Forests, 14 miles southeast of Yellow Pine, Idaho (fig. 1). The area was intermittently mined for gold, silver, mercury, antimony, arsenic, and tungsten from 1919 to 1997. During World War II, the Stibnite mining area produced 90 percent of the antimony (Klahr, 1987) and the majority of the tungsten (Mitchell, 2000) for the Allied war effort. Mining operations took place at Meadow Creek Mine in the Meadow Creek valley between 1919 and 1938; at Yellow Pine Mine in the area surrounding the Glory Hole (a pit lake, also known as the Yellow Pine Pit) between 1937 and 1952; and at West End Mine in areas near West End Creek and Garnet Creek between 1982 and 1997 (fig. 1). Cinnabar Mine was operated intermittently between 1902 and 1966 in the Cinnabar Creek drainage, which is a tributary to Sugar Creek (fig. 1). Additional mineralized areas that have not yet been mined also contribute to the impairment of water quality in the study area; however, identification and characterization of the unmined mineralized areas was not the focus of this work.

The study area is the headwaters of the East Fork South Fork Salmon River (EFSFSR), which, together with the South Fork Salmon River (SFSR), supports one of the most intact assemblages of native fishes in the Columbia River Basin (U.S. Forest Service, 2017). The EFSFSR is designated critical habitat for Chinook salmon (*Oncorhynchus tshawytscha*), steelhead (*O. mykiss*), bull trout (*Salvelinus confluentus*), and westslope cutthroat trout (*O. clarki lewisi*), all of which are listed as threatened, endangered, or sensitive (Idaho Department of Environmental Quality, 2002).

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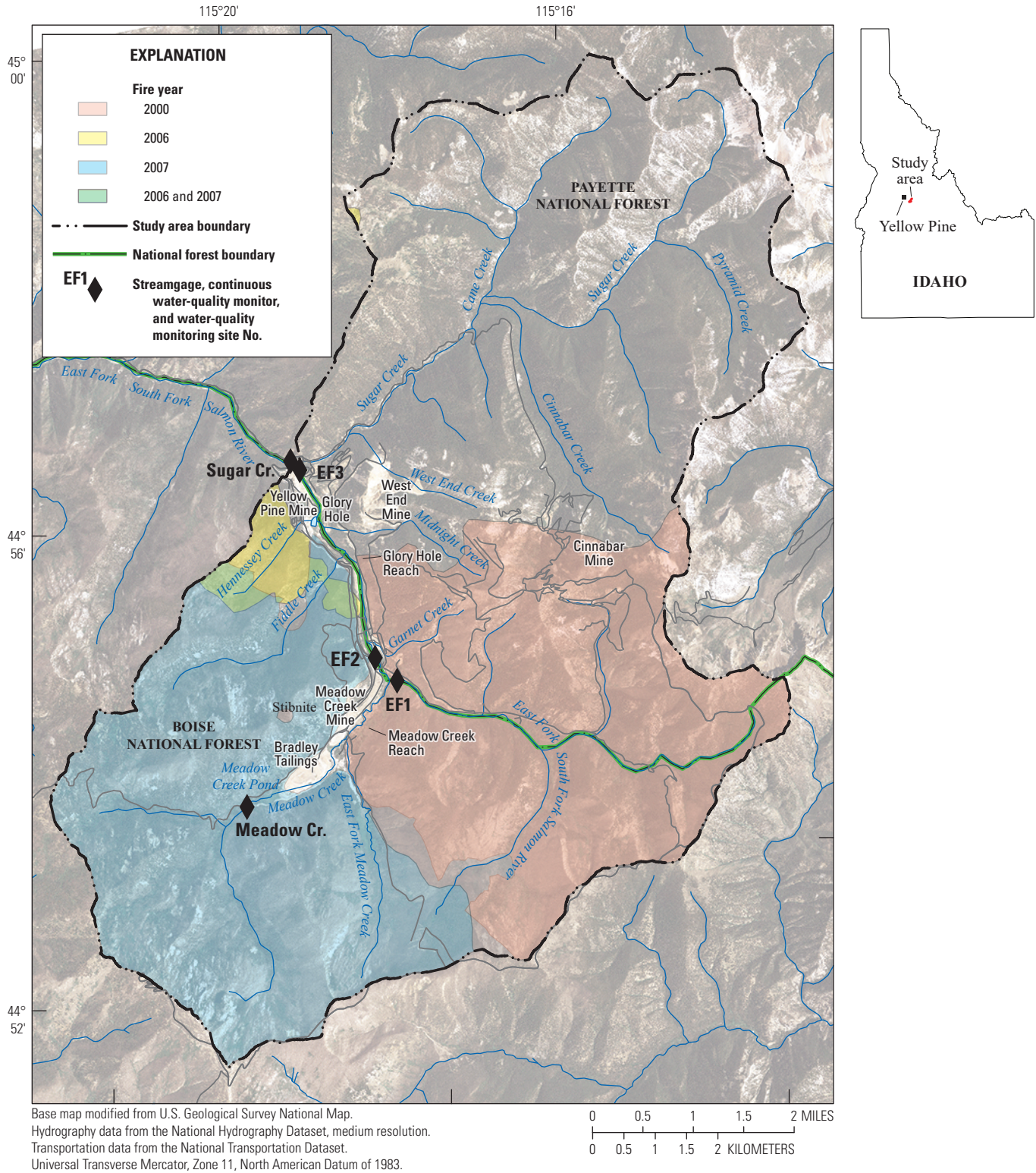


Figure 1. Streamgaging and water-quality monitoring sites near Stibnite mining area, central Idaho, 2011–17.

Historical mining-related activities have altered stream configuration and habitat in the study area. Mill tailings and spent ore were disposed directly into Meadow Creek (1930 to about 1946) and in impoundments (after about 1946) covering 100 acres of the Meadow Creek valley floor during active mining periods. The Meadow Creek stream channel was diverted around tailings and reconstructed several times to mitigate effects on water-quality from tailings. A hydroelectric dam constructed in the East Fork of Meadow Creek to power mining operations failed in 1965; the release of water transported and redistributed existing tailings and debris in the floodplain further downstream. Waste-rock dumps were generated upstream and downstream of the Glory Hole, and the EFSFSR was diverted around the Glory Hole beginning in 1938. After 1955, the Glory Hole was allowed to fill with water and remain part of the EFSFSR channel. An estimated 3,000 cubic yards of mercury-laden tailings deposited near the Cinnabar Mine on Cinnabar Creek act as a substantial source of mercury in Cinnabar and Sugar Creeks (Trainor, 2003).

These and other alterations of a strongly mineralized area resulted in water-quality impairments with implications for aquatic and human health (Etheridge, 2015). Previous water-quality investigations reported exceedances of chronic freshwater ambient water-quality criteria for arsenic, free cyanide, lead, mercury, silver, and zinc, and elevated concentrations of antimony (Trainor, 1993; URS Corporation, 2000; Etheridge, 2015).

Midas Gold Idaho, Inc., is conducting mineral exploration activities as part of the Stibnite Gold Project to better define the potential of mineral deposits in the Stibnite mining area. The exploration project identified a need to evaluate existing water-quality conditions in the study area and to identify potential source areas to target further reclamation efforts to be completed prior to or incorporated into plans for renewed mining activity. To help characterize water quality conditions, the U.S. Geological Survey (USGS), in cooperation with the Idaho Department of Water Resources (IDWR) and Midas Gold Idaho, Inc. (and later the Idaho Department of Lands), began a water-quality study in the area in 2011. Streamflow, water temperature, and specific conductance were measured continuously at five sites in the study area. Trace element and suspended-sediment concentrations were sampled at the same sites approximately every other month between October 2011 and present.

The USGS published results from the initial study period, October 2011–September 2014 (Etheridge, 2015). That report included concentration summaries of 20 trace elements from 5 sampling sites, each with 20–24 samples. The majority of the sampled trace elements were found to occur at concentrations below aquatic water-quality criteria and human-health based (HHB) criteria. Arsenic, antimony, and mercury were the only trace elements with frequent criteria exceedances. Etheridge (2015) developed surrogate models to continuously estimate

concentrations of arsenic, antimony, and mercury using specific conductance, streamflow, and other variables. Using continuous water temperature data, Etheridge showed that water temperature frequently exceeded salmonid and bull trout criteria during the initial study period.

Purpose and Scope

In the current report, additional monitoring data are used to update findings from Etheridge (2015), focusing on arsenic, antimony, mercury, and water temperature. This update is warranted because the study period has increased from 3 to 6 years, and the average number of samples at each site has increased from 22 to 38. The expanded study period includes 2017, a high water year in which mean daily streamflows at the five monitoring sites were 35–47 percent higher than in any previous study year. The expanded dataset representing more years, with more samples, improves our understanding of inter-annual variability, resulting in more relevant and robust characterizations of water quality under varying streamflow conditions. Specific objectives of this report include:

- Evaluate spatial and temporal occurrence of trace metals in the study area (that is, upstream to downstream, by watershed, and relative to streamflow and season);
- Assess exceedances of water-quality criteria;
- Update the original surrogate models and evaluate the importance of different explanatory variables on constituent concentrations.

Description of Study Area

The five monitoring sites discussed in this report are located in the Stibnite mining area in Valley County, central Idaho (fig. 1; table 1; described in detail in Etheridge, 2015). Three of the monitoring sites are located on the main-stem EFSFSR, upstream of the confluence with Sugar Creek. A fourth monitoring site is located on Sugar Creek, just upstream of the confluence with the EFSFSR. Within the Sugar Creek basin is the abandoned Cinnabar mercury mine, an area of known mercury contamination (Trainor, 2003). The fifth monitoring site is located on Meadow Creek, upstream of the historical mill tailings and spent ore disposal area. The Meadow Creek site is meant to characterize unmineralized background water-quality conditions, but is not meant to represent the pre-mining water-quality conditions of the entire study area. Even prior to mining, the extensive mineralization of the study area downstream of the Meadow Creek site may have resulted in increased trace metal concentrations in surface waters.

Table 1. Streamgaging and water-quality monitoring sites near the Stibnite mining area, central Idaho, 2012–17.

Abbreviated site name	Site No.	Streamgaging and water-quality monitoring site name
Meadow Cr.	13310850	Meadow Creek near Stibnite, Idaho
EF1	13310800	East Fork of South Fork Salmon River above Meadow Creek, near Stibnite, Idaho
EF2	13311000	East Fork of South Fork Salmon River at Stibnite, Idaho
EF3	13311250	East Fork of South Fork Salmon River above Sugar Creek, near Stibnite, Idaho
Sugar Cr.	13311450	Sugar Creek near Stibnite, Idaho

Study Methods

The methods used in this study were described previously (Etheridge, 2015). Briefly, water-quality and streamflow monitoring began at five sites between autumn 2011 and spring 2012 and is ongoing at the time of this report's publication (table 1; fig. 1). Continuous monitors were used to measure water temperature and specific conductance on a 15-minute interval at each site, operated according to USGS procedures (Wagner and others, 2006). Monitors were typically removed in winter because of limited access and to prevent ice-related damage. Stream stage height was measured at each site on a 15-minute interval throughout the study period and was used to estimate streamflow using standard USGS methods (Rantz and others, 1982).

Approximately six water-quality samples were collected annually at each site on a set interval, representative of a range of streamflow conditions and seasons. Water-quality samples were collected using cross-sectional, depth-integrating methods. Sampling equipment and procedures were consistent with those described in the USGS National Field Manual for the collection of trace metals (U. S. Geological Survey, variously dated). Samples for dissolved analyses were filtered through a pre-rinsed, 0.45-micrometer (μm) pore size, disposable capsule filter. Arsenic and antimony samples were preserved with 2 milliliters of Ultrex® nitric acid. Mercury samples were preserved with 2 mL of Omni-Trace® hydrochloric acid. Samples were shipped to the USGS National Water Quality Laboratory in Denver, Colorado.

Samples were analyzed at the USGS National Water Quality Laboratory using established analytical techniques. Arsenic and antimony concentrations were determined by atomic absorption spectrometry in conjunction with a graphite furnace and inductively coupled plasma-mass spectrometry (Fishman and Friedman, 1989; Fishman, 1993; Hoffman and others, 1996; Garbarino and Struzeski, 1998; Garbarino and others, 2006). Mercury concentrations were determined by atomic fluorescence spectrometry (Garbarino and Damrau, 2001).

Quality-assurance/quality-control (QA/QC) procedures included field blanks and field replicates, which are summarized in tables 2 and 3, respectively. Dissolved antimony was detected in 13.9 percent of field blanks ($n=36$), with a maximum concentration of 0.11 $\mu\text{g/L}$, and dissolved arsenic was detected in 8.3 percent of field blanks ($n=36$), with a maximum concentration of 0.06 $\mu\text{g/L}$. Total antimony and total arsenic ($n=36$ each) and dissolved and total mercury ($n=24$ and 27, respectively) were not detected in field blanks. The field-blank detections of dissolved but not total antimony and arsenic suggest occasional contamination during filtration of the dissolved samples, either from the pump hose or the capsule filter. Relative to environmental concentrations in this study, the detected concentrations of dissolved antimony and arsenic in field blanks are low.

Field replicate pairs of dissolved and total antimony and arsenic had median relative percent differences (RPD) of 0.7–1.55 percent ($n=22$ each). Field replicate pairs of dissolved and total mercury had median RPDs of 12.6 percent and 27.5 percent, respectively.

Table 2. Summary of results from field blanks from streams near Stibnite mining area, central Idaho, 2011–17.

[Abbreviation: $\mu\text{g/L}$, microgram per liter]

Constituent	Number of blanks	Blanks with detected concentrations (percent)	Maximum detected concentration ($\mu\text{g/L}$)
Antimony, dissolved	36	13.9	0.11
Antimony, total	36	0	ND
Arsenic, dissolved	36	8.3	0.06
Arsenic, total	36	0	ND
Mercury, dissolved	24	0	ND
Mercury, total	27	0	ND

Table 3. Summary of results from field replicate pairs from streams near Stibnite mining area, central Idaho, 2011–17.

[RPD: Relative percent difference, calculated using the absolute value of the difference between the result pair, divided by the mean of the result pair and multiplied by 100]

Constituent	Total replicate pairs	Replicate pairs with detected concentrations in both samples					Number of replicate pairs with detected concentration in only one sample	Number of replicate pairs with no detected concentrations
		Number of pairs	Median RPD	Mean RPD	Standard deviation of RPD	Maximum RPD		
Antimony, dissolved	22	22	1.20	1.54	1.28	4.3	0	0
Antimony, total	22	22	1.55	2.30	2.16	7.2	0	0
Arsenic, dissolved	22	22	0.70	1.23	1.63	5.2	0	0
Arsenic, total	22	22	1.55	2.63	2.54	8.7	0	0
Mercury, dissolved	14	8	12.6	13.0	10.8	33.3	1	5
Mercury, total	15	12	27.5	40.0	47.4	174	0	3

Summary statistics for constituents with detection frequencies less than 100 percent were computed using adjusted maximum likelihood estimation (Helsel, 2012) using the *censStats* function in the “smwrQW” package in R (R Core Team, 2015; Lorenz, 2018). Relations between constituent concentrations and streamflow were assessed using Spearman correlation with a significance level (*p*-value) of 0.05, computed using the *rcorr* function in the “Hmisc” package in R (Harrell and others, 2015). The percentage of arsenic, antimony, and mercury occurring in the dissolved phase (percent dissolved) was computed for each sample as the dissolved concentration divided by the total concentration, times 100. Percent dissolved summary statistics excluded samples with nondetects, as percent dissolved could not be computed for those samples. In some instances, in samples with virtually all of a given constituent present in the dissolved phase, the dissolved concentration was reported to be higher than the total concentration because of measurement uncertainties. In those instances, the percent dissolved was reported as 100.

Non-QA/QC data collected as part of this study are publicly available from the USGS National Water Information System (U.S. Geological Survey, 2017). QA/QC data are available upon request to the USGS Idaho Water Science Center.

Constituent concentrations and water temperatures were compared to State of Idaho water-quality criteria to assess potential harm to human health or aquatic life (Idaho Department of Environmental Quality, 2014). Criteria are listed in table 4. Criteria for arsenic and antimony are based on dissolved fractions only. Criteria for mercury are based on total mercury. None of the criteria used require corrections based on water hardness (they are not hardness-dependent).

Daily average and daily maximum water temperatures were computed from 15-minute data. Exceedances of the maximum weekly maximum water-temperature criterion for bull trout habitat (MWMT-BT) were determined by comparing the criterion value (13 degrees Celsius [°C]) to the 7-day rolling average of maximum water temperatures.

Table 4. State of Idaho water-quality criteria for arsenic, antimony, mercury, and water temperature.

[**Criterion:** HHB, human-health based criterion; CCC, criterion continuous concentration or “chronic” aquatic-life criterion; CMC, criterion maximum concentration or “acute” aquatic life criterion; MDAT-SS, maximum daily average water-temperature criterion for salmonid spawning; MDMT-SS, maximum daily maximum water-temperature criterion for salmonid spawning; MWMT-BT, maximum weekly maximum water-temperature criterion for bull trout habitat; MDAT-CW, maximum daily average water-temperature criterion for coldwater aquatic life. **Criterion value:** Dissolved arsenic, dissolved antimony, and total mercury shown in microgram per liter (µg/L); water temperature shown in degrees Celsius (°C)]

Constituent	Criterion	Criterion value (µg/L and °C)
Dissolved arsenic	HHB	10
	CCC	150
Dissolved antimony	HHB	5.6
	CMC	2.1
Total mercury	CCC	0.012
	CMC	2.1
Water temperature	MDAT-SS	9
	MDMT-SS	13
	MWMT-BT	13
	MDAT-CW	19

Surrogate Regression Models for Estimating Constituent Concentrations

Surrogate regression models can provide real-time estimates of concentrations for constituents of regulatory interest. Multiple linear regression models were developed using continuously monitored specific conductance, streamflow, hydrologic indices, and time/season variables (surrogates, collectively) to estimate continuous concentrations of dissolved arsenic, dissolved antimony, and total mercury at the five monitoring sites. Surrogate models were developed using stepwise linear regression analysis as described in Wood and Etheridge (2011). Stepwise linear regression involves testing a number of explanatory variables to determine which are the best predictors of measured concentrations.

Unlike Wood and Etheridge (2011), instantaneous values of water-quality parameters were paired with values obtained from discrete sample analytical results, rather than daily values. Predictor variables were assessed for their significance (using a p -value of less than 0.05) in estimating the constituent of interest, and the variance inflation factor (VIF) was used with a maximum threshold of 4 to detect multicollinearity (problematic correlation between variables) as additional predictor variables were assessed in the regression model. The lowest Mallows' C_p and predicted residual error sum of squares (PRESS) statistic were used to identify models for further exploration. Finally, residuals analysis was used to identify that a plot of residuals versus observed values were homoscedastic, and that residuals plotted against each predictor variable, including time, were randomly distributed. The adjusted coefficient of determination (R^2_{adj}) and mean square prediction error were also used to express the overall ability of the final model to explain the variability in observed sample results and with what degree of error.

Regression models were evaluated using the USGS R statistical programming packages “smwrQW” (Lorenz, 2018), “smwrStats,” “DVstats,” and “dataRetrieval” (Hirsch and De Cicco, 2015), all of which are provided in the public domain at U.S. Geological Survey (variously dated). Particularly, regression assumptions and methods used in R packages are described in detail in Mallows (1973), Miller (1990), and Helsel and Hirsch (2002). Methods used for regression estimation of left-censored (non-detected) mercury results in two of the models are further described in Cohn (1988), Breen (1996), and Helsel (2012). Specifically, left-censored results for mercury were re-expressed in models using the Adjusted Maximum Likelihood Estimation method.

The functional form of the surrogate models is:

$$\ln C = I + a(SC) + b(\ln SC) + c(Q) + d(SWE) + e(BFI) + f[\text{square root}(QBFI)] + g[\text{square root}(Qrange)] + h[\sin(2\pi T)] + i[\cos(2\pi T)] + j(T)(1)$$

where

\ln	is the natural logarithm;
C	is the constituent concentration in $\mu\text{g/L}$;
I	is the regression intercept;
SC	is specific conductance in microsiemens per centimeter at 25 °C;
Q	is the sampled streamflow in cubic feet per second;
SWE	is the snow water equivalent;
BFI	is the base flow index;
$QBFI$	is the streamflow divided by the BFI;
$Qrange$	is the ratio of daily range in streamflow to mean daily streamflow;
T	is decimal time expressed as a year with the decimal representing the day of that year as a fraction, for example December 31, 2011, is 2011.999, January 1, 2012, is 2012.001;
$\sin(2\pi T)$ and $\cos(2\pi T)$	are periodic time functions that describe seasonal variability;
a, \dots, j	are the regression coefficients that remain constant over time.

Surrogate models were calibrated using results from discrete samples and concurrent streamflow and water-quality parameters. During the winter when in-situ water-quality sondes were removed, discrete measurements of water-quality parameters were collected along with discrete samples.

Three streamflow indices were assessed for their potential as explanatory variables in regression models for total mercury. These were the base flow index (BFI), the sampled streamflow divided by the BFI (“QBFI”), and the ratio of daily range in streamflow to mean daily streamflow (“Qrange”). These indices were not routinely assessed as explanatory variables for dissolved arsenic and antimony because the dissolved fraction of arsenic and antimony is the dominant form found in the study area and dissolved arsenic and antimony loading and transport is well-explained by instream changes in specific conductance.

The method for computing BFI is described in Gustard and others (1992) and incorporated into the “DVstats” R package cited above. A BFI closer to 1 indicates a larger percentage of streamflow derived from baseflow (groundwater) as opposed to snowmelt or rainfall runoff (surface water). In some regression models, BFI was used to weight streamflow by dividing the sampled streamflow by the BFI for “QBFI.” QBFI computed using a very low BFI is indicative of a large runoff event and results in a larger weighted streamflow using this weighting technique. Any “Qrange” close to 1 indicates a larger range in streamflow relative to the mean daily streamflow and is useful for estimating system response to short-duration summer storms. In some cases, QBFI and Qrange were transformed by taking the square root to improve linear fit with the response variable.

In addition to BFI, QBFI, and Qrange, snow-water equivalent (SWE) was assessed as an explanatory variable for estimating dissolved antimony concentration because of the timing of peak antimony concentrations (discussed further in section, “Results”). SWE was obtained in inches as a daily time-series from the Natural Resources Conservation Service Deadwood Summit Snow Telemetry site (site number 439, Natural Resources Conservation Service, 2018); all values were adjusted by +0.1 to eliminate zeros and negatives in preparation for data transformation.

Results

The majority of arsenic occurred in the dissolved phase, from an average of 82 percent dissolved at Meadow Creek to 96 percent dissolved at EF1, suggesting a groundwater source. Arsenic concentrations generally increased from upstream to downstream along the main stem of the EFSFSR (table 5; fig. 2A). Dissolved arsenic concentrations averaged 8.86 µg/L at EF1 and increased to 56.5 µg/L at EF3. The lowest arsenic concentrations were measured at Meadow Creek, upstream of the historical Bradley tailings (mean 1.03 µg/L, dissolved). The highest concentrations of arsenic typically occurred during low-flow periods (July–March), indicating a groundwater source (figs. 3A and 4A). Dissolved arsenic and streamflow were significantly inversely related at all sites, with Spearman correlation coefficients between -0.70 at EF2 to -0.91 at EF1 (table 6).

Like arsenic, concentrations of antimony typically increased from upstream to downstream along the main stem of the EFSFSR (table 5; fig. 2B). Dissolved antimony concentrations averaged 0.930 µg/L at EF1 and increased to 27.9 µg/L at EF3. Antimony concentrations were lowest at Meadow Creek (mean 0.320 µg/L, dissolved). As with arsenic, antimony primarily occurred in the dissolved phase (site means of 94–100 percent dissolved), and concentrations were lowest during high flow periods (fig. 3B), suggesting a groundwater source. Concentrations of dissolved antimony were significantly inversely related to streamflow at EF1, EF3, and Sugar, but were not significantly related at Meadow Creek or EF2 (table 6). Unlike arsenic, peak antimony concentrations were generally observed during the first flush of spring snowmelt on the rising limb of the hydrograph (fig. 4B) rather than during low flow conditions. This first flush phenomenon has been observed at other mine sites and has been attributed to the dissolution of soluble salts and the flushing out of waters that were concentrated by evaporation (Nordstrom, 2009). The observance of this phenomenon with dissolved antimony but not dissolved arsenic may be related to differences in adsorption behavior with minerals and (or) organic matter in the soil (Dousova and others, 2015).

Mercury concentrations were consistently highest at Sugar Creek, where the average total mercury concentration was 1.19 µg/L (table 5; fig. 2C). Mercury concentrations were lowest at Meadow Creek, where it was typically not detected. In contrast to arsenic and antimony, mercury primarily occurred in the particulate phase, especially at Sugar Creek, where particulate mercury averaged 82 percent of total mercury (EF1, EF2, and EF3 averaged 52–63 percent particulate). The association with particles indicates that the mercury is coming from erosion and (or) resuspension of surface material, rather than groundwater. Also in contrast to dissolved arsenic and antimony, total mercury concentrations were highest during high-flow periods (April–June; figs. 3C and 4C), another indicator of an erosion/surface material source. Total mercury concentrations were significantly positively related to streamflow at EF1, EF2, EF3, and Sugar Creek, with Spearman correlation coefficients between 0.64 and 0.84 (table 6).

Exceedances of Water-Quality Criteria

Exceedances of water-quality criteria for arsenic, antimony, and (or) mercury were frequent at all sampling sites except in the unmineralized Meadow Creek site, where only one exceedance was observed. Samples from sites EF1, EF2, EF3, and Sugar Creek regularly exceeded the HHB criterion for dissolved arsenic (10 µg/L), with exceedance frequencies as high as 98 and 100 percent at EF2 and EF3, respectively (fig 2A). The chronic aquatic life criterion for dissolved arsenic (150 µg/L) was not exceeded at any site (the maximum sample concentration was 108 µg/L).

Exceedances of the HHB criterion for dissolved antimony (5.6 µg/L) were common at EF2 and EF3, with exceedance frequencies of 75 and 100 percent, respectively (fig 2B). An aquatic life criterion for antimony has not been established.

The mercury chronic aquatic life criterion (0.012 µg/L) was exceeded in at least one sample at all sites, with exceedance frequencies ranging from 4 percent at Meadow Creek to 97 percent at Sugar Creek (fig. 2C). The acute aquatic life criterion (2.1 µg/L) was only exceeded at Sugar Creek, with a frequency of 11 percent.

At all sites, on the basis of continuous data, summer water temperatures frequently exceeded water temperature criteria related to salmonid spawning and (or) bull trout (fig. 5). Exceedances occurred June–September, but were most common in July and August. Sites EF1 and Meadow Creek had the fewest days with temperature exceedances, with a maximum of 15 percent of days per month exceeding the maximum daily maximum water-temperature criterion for salmonid spawning (MDMT-SS, 13 °C) and the MWMT-BT (13 °C). The maximum daily average water-temperature criterion for salmonid spawning (MDAT-SS, 9 °C), a criterion more sensitive than the MDMT-SS or MWMT-BT, was exceeded up to 83 percent of days at Meadow Creek and up to 59 percent of days at EF1.

Table 5. Summary of concentrations of select constituents from five monitoring locations in the Stibnite mining area, central Idaho, 2011–17.

[Summary statistics for constituents with detection frequency less than 100 percent were computed using the censStats function in the “smwrQW” package in R. All concentrations in microgram per liter. Site abbreviations are defined in table 1 and site locations are shown in figure 1. **Abbreviation:** NA, not applicable]

Constituent	Number of samples	Detection frequency (percent)	Mean	Standard deviation	Median	Maximum
Meadow Creek						
Arsenic, dissolved	38	100	1.03	0.268	0.900	1.50
Arsenic, total	38	100	1.57	1.71	1.25	11.3
Antimony, dissolved	35	100	0.320	0.098	0.290	0.570
Antimony, total	38	100	0.340	0.195	0.260	1.32
Mercury, dissolved	23	0	NA	NA	NA	NA
Mercury, total	24	8	0.002	0.007	0.000	0.014
EF1						
Arsenic, dissolved	36	100	8.86	2.94	9.55	12.8
Arsenic, total	36	100	9.18	2.69	9.70	13.4
Antimony, dissolved	36	100	0.930	0.361	0.910	1.86
Antimony, total	36	97	0.928	0.343	0.900	1.72
Mercury, dissolved	22	41	0.005	0.001	0.005	0.010
Mercury, total	23	87	0.012	0.011	0.007	0.051
EF2						
Arsenic, dissolved	40	100	23.7	9.83	24.4	52.6
Arsenic, total	40	100	25.7	10.3	26.2	59.2
Antimony, dissolved	39	100	10.9	10.6	7.39	50.0
Antimony, total	39	100	11.6	11.2	8.13	57.6
Mercury, dissolved	28	11	0.004	0.001	0.004	0.006
Mercury, total	30	60	0.017	0.039	0.006	0.304
EF3						
Arsenic, dissolved	39	100	56.5	28.6	57.2	108
Arsenic, total	39	100	63.4	35.5	62.9	150
Antimony, dissolved	38	100	27.9	15.0	26.8	71.9
Antimony, total	38	100	27.9	14.7	25.3	72.8
Mercury, dissolved	31	16	0.004	0.001	0.003	0.007
Mercury, total	32	50	0.008	0.009	0.005	0.037
Sugar Creek						
Arsenic, dissolved	38	100	12.1	5.31	10.85	31.1
Arsenic, total	38	100	14.5	6.56	14.1	35.1
Antimony, dissolved	37	100	3.35	2.84	2.25	12.9
Antimony, total	37	100	3.41	2.68	2.70	11.7
Mercury, dissolved	35	94	0.014	0.011	0.010	0.302
Mercury, total	36	100	1.19	4.45	0.070	26.3

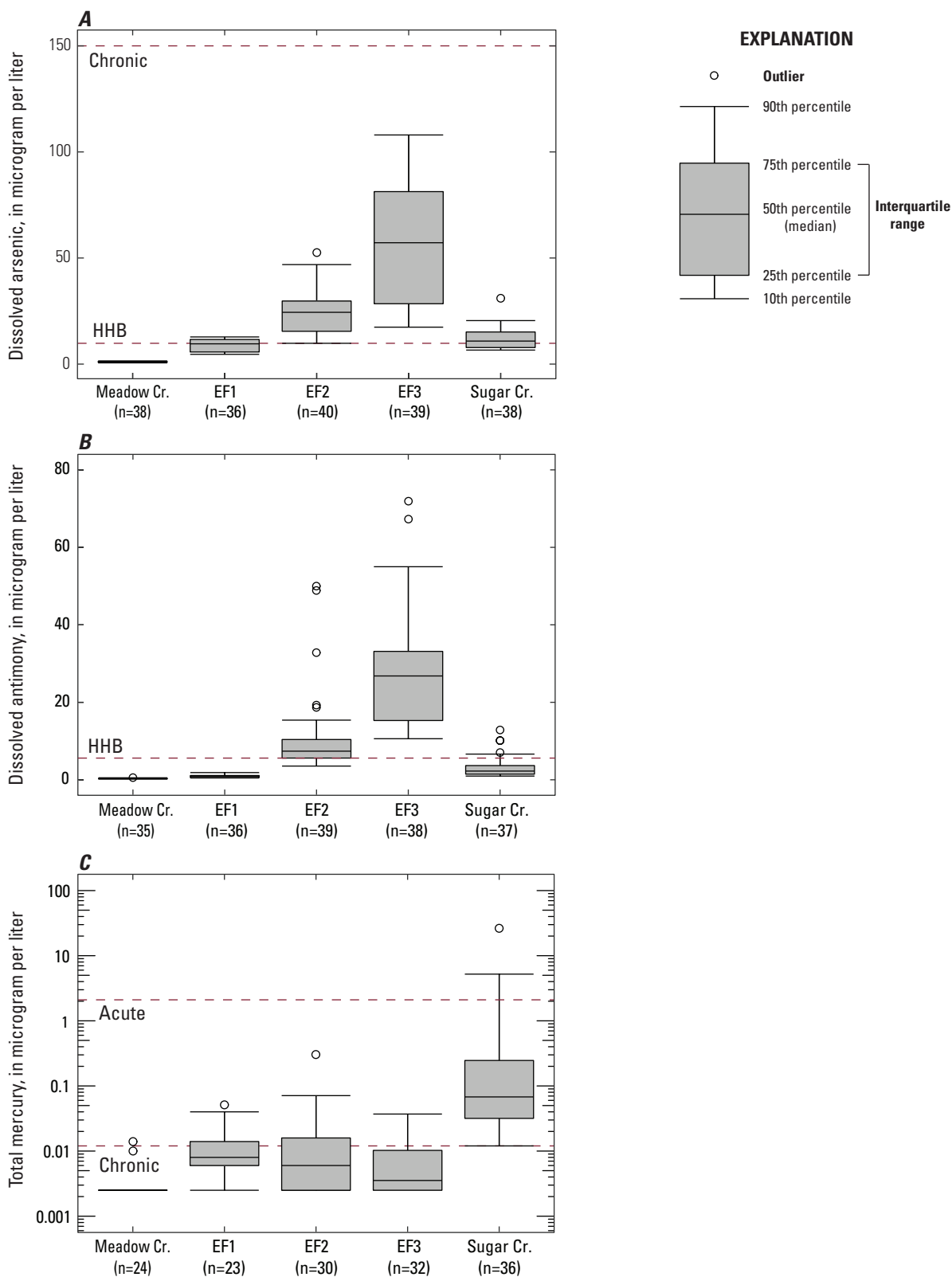


Figure 2. Concentrations of dissolved arsenic (A), dissolved antimony (B), and total mercury (C) relative to water-quality criteria at five monitoring sites in the Stibnite mining area, central Idaho, 2011–17. Concentrations below the reporting level are plotted at half the reporting level for visualization purposes. Site names and locations are shown in [table 1](#) and [figure 1](#), respectively. Water-quality criteria are defined in [table 4](#). HHB, human-health based criterion.

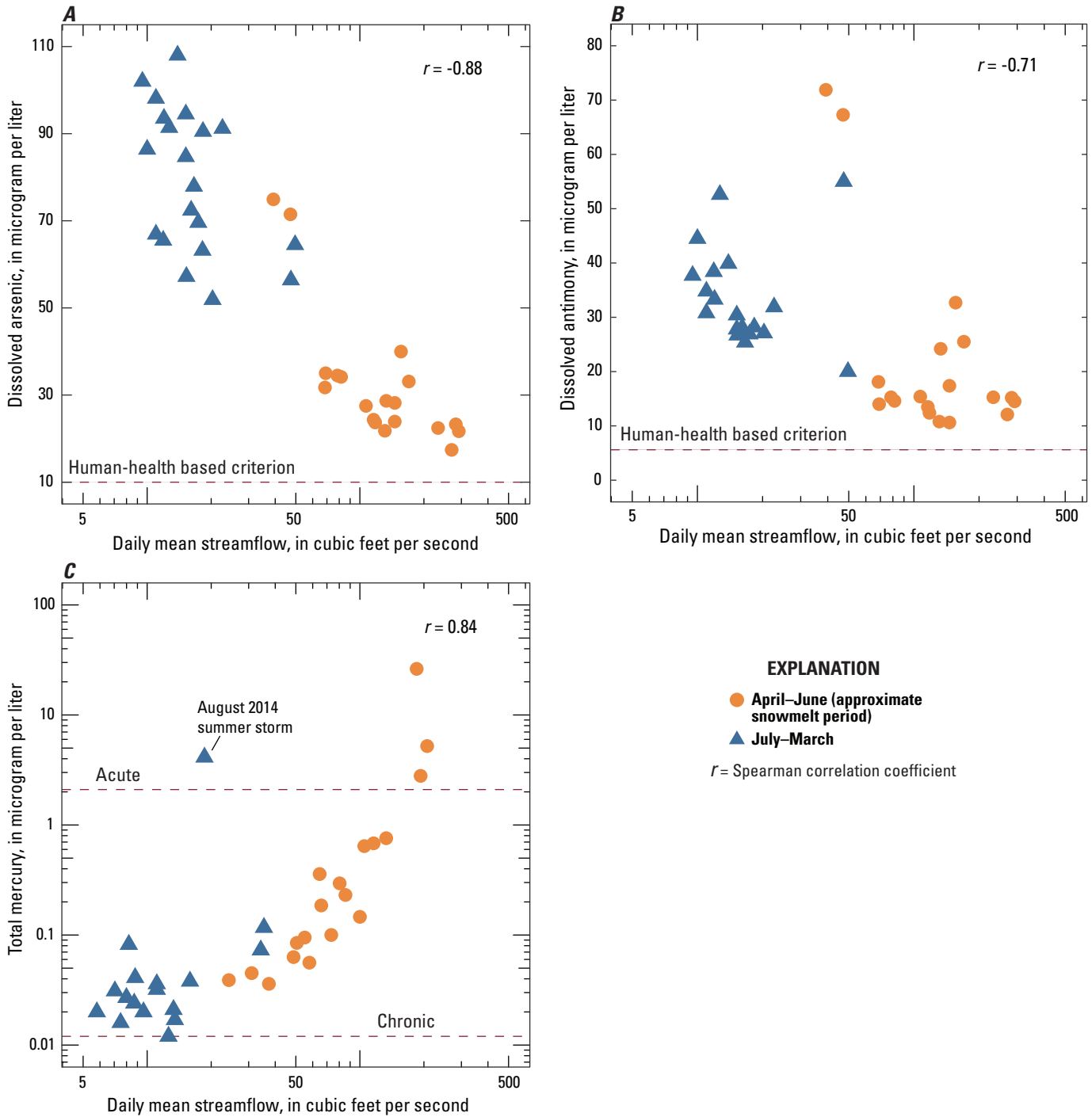


Figure 3. Concentration versus daily mean streamflow for dissolved arsenic at EF3 (A), dissolved antimony at EF3 (B), and total mercury at Sugar Creek (C), 2011–17. Concentrations below the reporting level are plotted at half the reporting level. Site names and locations are shown in table 1 and figure 1, respectively. Water-quality criteria are defined in table 4. HHB, human-health based criterion.

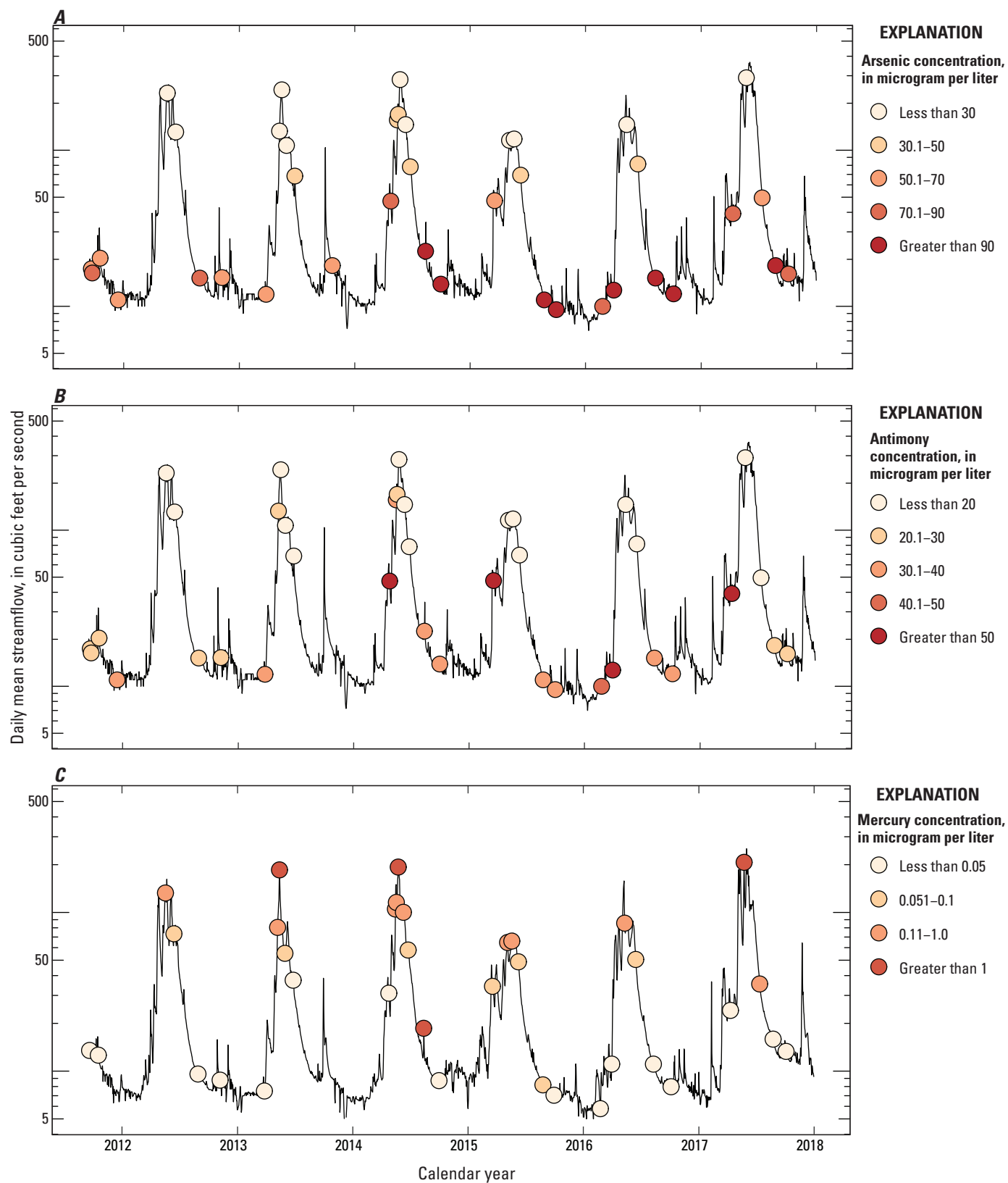


Figure 4. Daily mean streamflow, sample timing, and concentration of dissolved arsenic at EF3 (A), dissolved antimony at EF3 (B), and total mercury at Sugar Creek (C), 2011–17. Site names and locations are shown in [table 1](#) and [figure 1](#), respectively. Water-quality criteria are defined in [table 4](#).

Table 6. Statistically significant ($p < 0.05$) Spearman correlation coefficients between streamflow and constituent concentrations at each site.

[Positive coefficients indicate a direct (positive) relationship. Negative coefficients indicate an inverse relationship. Site names and locations are shown in [table 1](#) and [figure 1](#), respectively. **Abbreviations:** NS, not significant.]

	Arsenic		Antimony		Mercury	
	Dissolved	Total	Dissolved	Total	Dissolved	Total
Meadow Cr.	-0.88	NS	NS	NS	NS	NS
EF1	-0.91	-0.86	-0.69	-0.69	0.63	0.79
EF2	-0.70	-0.57	NS	NS	NS	0.64
EF3	-0.88	-0.85	-0.71	-0.69	0.47	0.69
Sugar Cr.	-0.85	-0.51	-0.88	-0.81	0.73	0.84

Water temperature criteria exceedances were much more frequent at EF2 than at EF1 ([fig. 5](#)), likely because mining disturbances and forest fires ([fig. 1](#)) have left much of Meadow Creek without shade. The MDMT-SS, MDAT-SS, and MWMT-BT were exceeded up to 83, 94, and 82 percent of days, respectively, at EF2, and up to 59, 100, and 59 percent of days, respectively, at EF3. The MDAT-CW was not exceeded at any site during the study period.

Surrogate Regression Models

Surrogate regression models that use input variables from real-time in-stream sensors can be used to assess changes in stream quality in near-real-time. Surrogate models also can be used to estimate temporal variability in constituent concentration and load over any desired time frame of interest, from hours to years. The population of samples used to compute regression estimates of constituents of interest is larger and more varied than the sample population used in Etheridge (2015). Because of this more varied dataset, coefficients for explanatory variables changed in many cases. Changes to model coefficients do not necessarily indicate changes in drivers of constituent transport, nor do they indicate a change in the conclusion that continuously measured in-stream specific conductance and streamflow are good explanatory surrogates that can be used to estimate constituent concentrations in near real time. As more conditions are sampled, surrogate model validation and assessment for changes in explanatory variable coefficients is recommended practice (Rasmussen and others, 2009). Changes in explanatory variable coefficients during model recalibration efforts reflect the fact that the population of samples used to calibrate models is more varied. For example, specific conductance remains positively correlated and significant in linear or log space in all the dissolved arsenic and antimony models, and streamflow remains positively correlated with total mercury concentrations, but the coefficients and (or) transformations may have been changed to improve model fit with a more diverse population of calibration samples ([table 7](#)). In some cases, explanatory

variables previously used to fit models for seasonal change were no longer significant or were replaced with a more suitable explanatory variable to capture a seasonal component of constituent loading and transport, such as SWE ([table 7](#)). However, because the updated models are based on a larger and more varied dataset, they are considered to better reflect the interactions between constituent concentrations and explanatory variables compared to the original models.

Generally, surrogate regression models for dissolved arsenic, dissolved antimony, and total mercury at EF3 and Sugar Creek are the most useful models for ongoing monitoring and assessment of trends in constituent concentration and loads because together these two sites represent water-quality conditions leaving the study area. As such, models from these two sites can be used to estimate total mercury, dissolved arsenic, and dissolved antimony loading to points downstream. Model archive summaries are provided in the appendix.

Meadow Creek

The updated surrogate regression model for dissolved arsenic in Meadow Creek agrees with the conclusion of the original model that arsenic is derived from groundwater upstream of the monitoring site. Etheridge (2015) described how increases in streamflow due to rainfall or snowmelt runoff coincided with decreases in dissolved arsenic and specific conductance. Streamflow remains negatively correlated with dissolved arsenic at Meadow Creek and elsewhere ([table 6](#); [fig. 3A](#)), but was not included in the Meadow Creek regression model because specific conductance was a better predictor, and the model did not appreciably improve with the inclusion of streamflow.

In contrast to arsenic, regression analysis using a larger sample population indicated that dissolved antimony concentrations are not significantly related to changes in streamflow or specific conductance in Meadow Creek ([tables 6 and 7](#)). In the absence of explanatory variables, a regression model for dissolved antimony at Meadow Creek cannot be published.

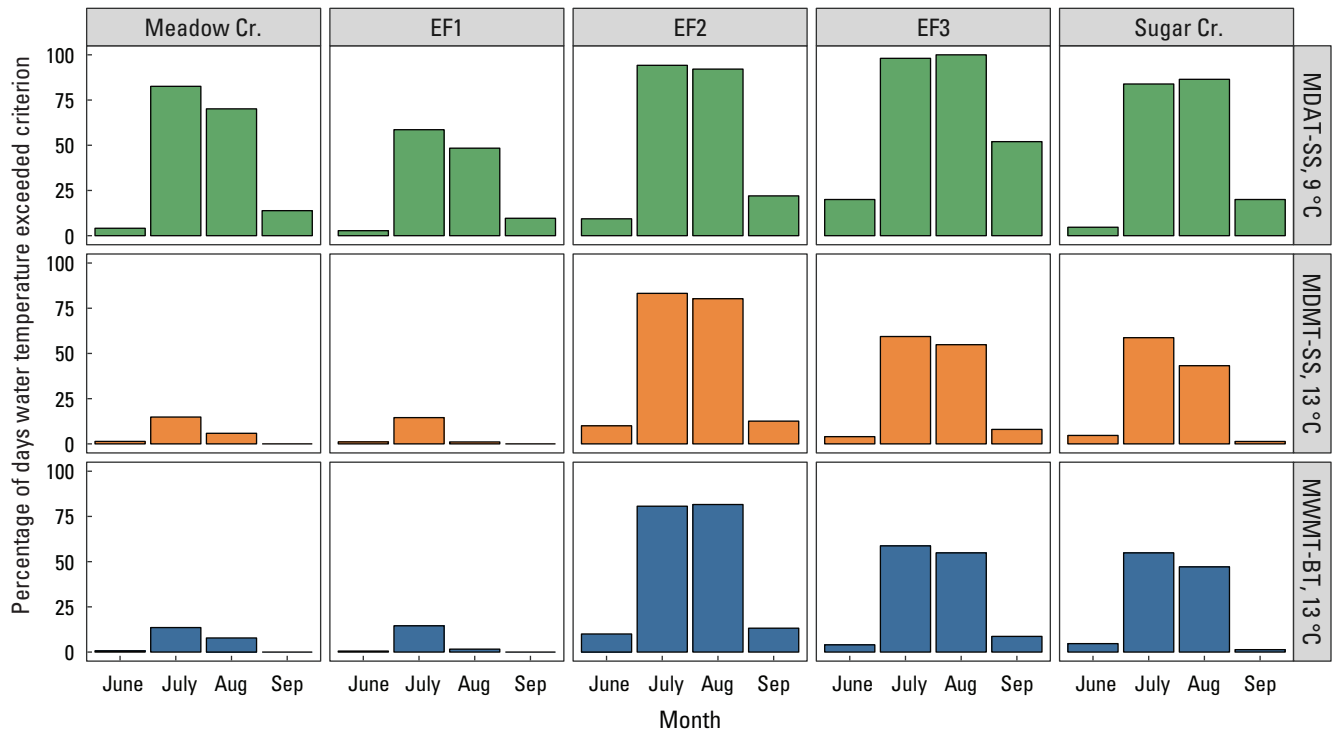


Figure 5. Percentage of days that water temperature exceeded criteria at five monitoring sites in the Stibnite mining area, central Idaho, June–September, 2011–17. Site names and locations are shown in [table 1](#) and [figure 1](#), respectively. Water-temperature criteria are defined in [table 4](#). [Aug, August; Sep, September]

Table 7. Regression coefficients and model diagnostics for surrogate models used to estimate concentrations of selected constituents at monitoring sites in the Stibnite mining area, central Idaho, 2011–17.

[Site names and locations are shown in table 1 and figure 1, respectively. Each regression model was calibrated using the available data collected at each site for water years 2012–17. The regression equation is $\ln C = I + a(SC + b(\ln SC) + c(Q) + d(SWE) + e(BFI) + f[\text{square root}(QBF)] + g[\text{square root}(Orange)] + h[\sin(2\pi T)] + i[\cos(2\pi T)] + j(T)$. Where \ln is the natural logarithm transformation; C is the constituent concentration, in micrograms per liter; I is the regression intercept; SC is specific conductance in millisiemens per centimeter; Q is the sampled streamflow, in cubic feet per second; SWE is the snow water equivalent; T is decimal time expressed as a year with the decimal representing the day of that year as a fraction; QBF , $Orange$, and BFI are terms describing streamflow variability; square root is the square root transformation; and $a, b, c, d, e, f, g, h, i,$ and j are regression coefficients that remain constant over time. R^2 (coefficient of determination), represents the amount of variance explained by the model in percent. **Abbreviations:** BCF, bias correction factor; MSPE, model standard percentage error; –, no regression coefficient]

Constituent	number	intercept	SC	lnSC	Q	SWE	BFI	QBF	Orange	sin	cos	decime	BCF	MSPE (percent)	Surrogate concentration model adjusted R ²
N	I	a	b	c	d	e	f	g	h	i	j				
Meadow Creek															
Arsenic, dissolved	37	0.252	0.014	–	–	–	–	–	–	–	–	–	NA	9.64	87
Antimony, dissolved	34	Model discontinued, no significant relation with continuously measured parameters													
EF1															
Arsenic, dissolved	35	-2.937	–	1.241	–	–	–	–	–	–	–	–	1.01	10.3	92
Antimony, dissolved	35	-5.544	–	1.279	–	0.013	–	–	–	–	–	–	1.01	17.0	81
Mercury, total	22	-5.228	–	–	0.022	–	–	–	–	–	–	–	1.07	49.1	75
EF2															
Arsenic, dissolved	38	-1.364	–	1.049	–	–	–	–	–	–	–	–	1.03	24.5	69
Antimony, dissolved	38	-1.970	–	0.855	–	0.030	–	–	–	–	–	–	1.07	36.3	71
Mercury, total	29	-5.089	–	–	0.066	-1.430	–	–	-0.638	-0.703	–	–	1.26	97.6	67
EF3															
Arsenic, dissolved	37	-275	74.71	–	–	–	–	–	-5.33	-10.93	–	–	na	17.8	88
Antimony, dissolved	37	-2.477	–	1.249	–	0.013	–	–	–	–	–	–	1.01	16.9	89
Mercury, total	31	-4.089	–	–	0.004	–	-1.622	–	–	–	–	–	1.08	50.1	67
Sugar Creek															
Arsenic, dissolved	36	108	0.012	–	–	–	–	–	–	–	–	-0.053	1.01	14.3	88
Antimony, dissolved	36	301	–	1.725	–	–	–	–	0.113	0.323	–	-0.153	1.01	17.3	94
Mercury, total	35	-354	–	–	–	–	–	0.179	4.368	–	–	0.173	1.37	78.5	85

East Fork of South Fork Salmon River Above Meadow Creek (EF1)

With the addition of 15 new samples, seasonality terms became insignificant as explanatory variables for dissolved arsenic concentration at EF1. The dissolved antimony model for EF1 remained the most similar to the originally published model, but explained less of the variation captured in the sample population used to calibrate the model. The dissolved antimony and arsenic surrogate regression models for EF1 are useful in real time to assess changes in baseline conditions in the study area.

The total mercury model developed for EF1 showed that streamflow alone is a good surrogate for estimating total mercury concentrations (table 7). There are not yet enough calibration samples to warrant additional explanatory variables in the EF1 total mercury surrogate regression model so only streamflow was used as an explanatory variable. Three out of 22 sample results for total mercury at EF1 were left-censored (13.6 percent). Additional model calibration samples and (or) a sample population containing less than 5 percent left-censored (Theresa Rasmussen, USGS Surrogates Workgroup, personal communication) total mercury results are needed before a linear regression model is used to estimate total mercury at EF1 in near real time, thus a model archive summary is not included in the appendix for the EF1 total mercury model.

East Fork of South Fork Salmon River at Stibnite (EF2)

Relations between continuously measured specific conductance and dissolved arsenic and antimony remained significant at EF2 with the addition of new samples. Surrogate regression models at EF2 accounted for less of the sampled variability in dissolved arsenic and antimony concentrations compared to surrogate regression models at other streamgages in the study area. Seasonality variables were no longer significant in either regression model at EF2. SWE was a significant explanatory variable in the dissolved antimony regression model because the highest dissolved antimony concentrations in EF2 were observed soon after snowpack began to melt, rather than at low flow (fig. 4B). SWE and unmeasured streamflow (streamflow at EF2 minus streamflow at EF1 minus streamflow at Meadow Creek) were evaluated as explanatory variables to improve the dissolved arsenic model fit at EF2, but were ultimately not included in the model because they resulted in non-normally distributed model residuals, violating an assumption of linear regression.

A total mercury surrogate regression model was developed for EF2 using 29 samples (table 7). The total mercury surrogate regression model for EF2 is not summarized in a model archive (appendix) because more samples should be collected before it is used to estimate total

mercury concentrations in near-real-time with computed and (or) telemetered explanatory variables from the streamgage. Out of 29 total mercury results, 11 (38 percent) were left-censored. Additional model calibration samples and (or) a sample population containing less than 5 percent left-censored total mercury results are needed before a linear regression model is used to estimate total mercury at EF2 in near real time. Streamflow and two hydrologic indices are positively correlated to mercury concentrations at EF2 (table 7). Although Sugar Creek contributes approximately 98 percent of the mercury load transported downstream of the study area (Etheridge, 2015), the EFSFSR is also a contributor of mercury upstream and downstream of the Glory Hole. Holloway and others (2017) suggested that the historical Fern mine was a source of mercury to the EFSFSR, attributable to increased stream sediment mercury concentrations at EF1.

The total mercury sample collected on April 11, 2017, represents the highest total mercury concentration (0.304 µg/L) sampled at EF2 (fig. 2C). This sample is an outlier because it corresponds to a relatively low streamflow of 30.4 ft³/s. The sample was collected during a brief period of relatively warm weather, which triggered a period of snowmelt before cooler weather returned (Natural Resources Conservation Service, 2018; University of Utah, 2018). Because mercury transport may occur as snowmelt initiates, SWE was particularly useful as an explanatory variable to fit the model to the 0.304 µg/L outlier. BFI was also a significant negative correlate to mercury concentrations at EF2 (table 7). The surrogate regression model for mercury at EF2 may improve with additional samples collected during brief increases of snowmelt runoff, early in the snowmelt runoff season, and during summer storms.

East Fork of South Fork Salmon River Above Sugar Creek (EF3)

Surrogate regression models remained relevant and useful for estimating real-time concentrations of dissolved arsenic and antimony at EF3. Like EF2, SWE was positively correlated with dissolved antimony at EF3.

The total mercury surrogate regression model at EF3 was improved with the addition of 16 new samples. Although the total mercury surrogate regression model at EF3 explained less of the variability in sampled total mercury than its predecessor, it achieves a 50.1 percent model standard percentage error around a detection limit of 0.005 µg/L. Out of 31 total mercury results at EF3, 15 (48 percent) are left-censored. The total mercury surrogate regression model for EF3 is not summarized in a model archive (appendix) because more samples and (or) a sample population containing less than 5 percent left-censored total mercury results are needed before a linear regression model is used to estimate total mercury concentrations in near-real-time with computed and (or) telemetered explanatory variables from the streamgage.

Sugar Creek

Decimal time was a significant explanatory variable in all surrogate regression models for Sugar Creek, indicating trend significance in constituent transport in Sugar Creek. Regression models developed for Sugar Creek during stepwise regression analysis exhibited a trend in residuals plotted against time unless decimal time was included as an explanatory variable in the regression equation. A negative coefficient for decimal time indicated a decreasing trend in concentrations of dissolved arsenic and antimony in Sugar Creek, though a nonparametric Mann-Kendall test for temporal trend in dissolved arsenic and antimony was insignificant. A positive coefficient for decimal time indicated an increasing trend in total mercury concentrations in Sugar Creek, and again, a nonparametric Mann-Kendall test for temporal trend in total mercury was insignificant. In each case where decimal time was used as an explanatory variable, it removed bias in residuals over time. Use of decimal time as an explanatory variable in each case also resolved violations of regression assumptions such as the assumption that residuals are homoscedastic and normally distributed. In all three cases, decimal time should be verified or removed as an explanatory variable in any future iterations of each model, especially when there are enough samples to subset the calibration dataset and validate models with hold-out samples.

Dissolved arsenic and antimony model standard percentage errors improved at Sugar Creek with 15–16 additional samples collected since 2015. Together with regression models at EF3, the surrogate regression models at Sugar Creek can be used to estimate concentration and flux of dissolved arsenic and antimony transported downstream of the study area with a high degree of statistical significance and quantifiable uncertainty in the form of real-time prediction intervals (U.S. Geological Survey, 2018). Overall statistical significance of each regression model is assessed using the model standard percentage error (table 7) and was assessed during stepwise regression using an F-statistic and associated p-value, the PREDiction Error Sum of Squares (PRESS) statistic, and k-fold cross-validation (Fushiki, 2011) (appendix).

The total mercury model at Sugar Creek was improved with the addition of 16 more samples since 2015. The sample population for both the previously published and the revised total mercury models included a summer storm sample from August 14, 2014 (Etheridge, 2015). With a concentration of 4.13 $\mu\text{g/L}$ and a corresponding streamflow of 24.2 ft^3/s , the August 14, 2014, sample remains an outlier (fig. 3C). Including the outlier in the regression model results in non-normally distributed residuals, a basic violation of regression model assumptions. The model currently includes results from one summer storm to use in assessing residuals distribution. Removing the summer storm sample outlier from August 14,

2014, improves the model, but limits its ability to provide reasonable estimates of mercury concentration or flux during summer storms. Summer storms may account for relatively large mass flux of mercury out of the study area, so the summer storm sample was kept in the calibration dataset.

Mercury transport in the study area is positively correlated to streamflow events regardless of their seasonal timing (fig. 4C). Streamflow and (or) hydrologic indices derived from continuous streamflow were significant as predictors of total mercury concentration at EF1, EF2, EF3, and Sugar Creek (table 7). Two hydrologic index terms were used with decimal time as explanatory variables in the revised total mercury model for Sugar Creek. The first index term used BFI to weight streamflow by dividing streamflow by BFI (table 7; QBFI). The most significant improvement of total mercury model fit came with the addition of a hydrologic index computed using the ratio of mean daily streamflow to daily range in streamflow (table 7; Qrange). This results in a Qrange index close to or greater than 1 for any sudden increase in streamflow such as a summer storm or any surge of snowmelt runoff.

Continuous precipitation data could improve models used to estimate mercury concentrations at Sugar Creek and elsewhere, especially during localized summer storms and brief periods of warm weather as snowmelt runoff season begins. These data are currently collected at a site near the Stibnite air strip but large gaps in the time-series precluded their inclusion in the models.

Summary

The Stibnite mining area, in the headwaters of the EFSFSR, was intermittently mined for most of the 20th century. Results from this study show that, decades after mining ceased, water quality in the area continues to be impaired. Concentrations of dissolved arsenic and antimony generally increased from upstream to downstream in the EFSFSR. Based on the majority of arsenic and antimony occurring in the dissolved phase, and inverse relationships with streamflow at most sites, the primary route of arsenic and antimony to surface waters is likely from groundwater. Mercury, in contrast, was directly related to streamflow and was associated with particulates, suggesting erosion of surface materials to be the primary route to surface waters. Mercury concentrations were highest in Sugar Creek.

Arsenic and antimony concentrations regularly exceeded human-health based criteria, and mercury concentrations frequently exceeded the chronic aquatic life criterion at some sites. Water temperatures, potentially affected by a combination of mining disturbances, loss of shade from forest fires, and (or) climate change, commonly exceeded salmonid spawning and bull trout criteria.

The additional 3 years of monitoring since the initial study report (Etheridge, 2015) improved our understanding of inter-annual variability, resulting in more relevant characterizations of water quality under varying streamflow conditions. A number of the surrogate models changed with the inclusion of the additional monitoring data. Because they represent a greater diversity of environmental conditions, the revised surrogate models should be more robust than the initial models. Streamflow, specific conductance, SWE, and hydrologic indices derived from streamflow were all important explanatory variables in surrogate regression models.

Future changes in the EFSFSR watershed are likely. These changes may be related to additional mining and (or) remediation, forest fires and (or) regrowth of fire-impacted areas, or climate change. All of these factors have the potential to impact water quality. Given the importance of the EFSFSR (and the SFSR downstream) as critical habitat for Chinook salmon, steelhead, bull trout, and westslope cutthroat trout, the remediation of the Stibnite mining area is a priority for many government and non-government organizations. Results from this study can inform future remediation and monitoring efforts by identifying the stream reaches with the highest contaminant concentrations and water temperatures, identifying contaminant pathways into the surface water (groundwater versus erosion/surface material), prioritizing contaminants based on water quality criteria exceedances, and providing valuable baselines against which future changes may be measured.

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Appendix. Surrogate Regression Model Archive Summaries

Appendix can be accessed at <https://doi.org/10.3133/sir20195072>.

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