

Water Resources Research



RESEARCH ARTICLE

10.1029/2020WR028581

Key Points:

- Substantial and widespread declines in dissolved-solids loads and concentrations were observed across the Upper Colorado River Basin during the 20th century
- Majority of the downward trends are estimated to be the result of watershed-related processes, not changes in the streamflow regime
- Decreases in the 1940s indicate watershed processes other than salinity control substantially affected dissolved-solids transport

Supporting Information:

Supporting Information may be found in the online version of this article.

Correspondence to:

C. A. Rumsey,
crumsey@usgs.gov

Citation:

Rumsey, C. A., Miller, O., Hirsch, R. M., Marston, T. M., & Susong, D. D. (2021). Substantial declines in salinity observed across the Upper Colorado River Basin during the 20th century, 1929–2019. *Water Resources Research*, 57, e2020WR028581. <https://doi.org/10.1029/2020WR028581>

Received 12 AUG 2020
Accepted 22 MAR 2021

Published 2021. This article is a U.S. Government work and is in the public domain in the USA.
This is an open access article under the terms of the [Creative Commons Attribution](https://creativecommons.org/licenses/by/4.0/) License, which permits use, distribution and reproduction in any medium, provided the original work is properly cited.

Substantial Declines in Salinity Observed Across the Upper Colorado River Basin During the 20th Century, 1929–2019

Christine A. Rumsey¹ , Olivia Miller¹ , Robert M. Hirsch² , Thomas M. Marston¹ , and David D. Susong¹ 

¹U.S. Geological Survey, Utah Water Science Center, Salt Lake City, UT, USA, ²U.S. Geological Survey, National Center, Reston, VA, USA

Abstract Salinity in the Colorado River Basin causes an estimated \$300 to \$400 million per year in economic damages in the United States. To inform and improve salinity-control efforts, this study quantifies long-term trends in salinity (dissolved solids) across the Upper Colorado River Basin (UCRB), including time periods prior to the construction of large dams and preceding the implementation of salinity-control projects. Weighted Regressions on Time, Discharge, and Season was used with data sets of dissolved-solids and specific-conductance measurements, collected as early as 1929, to evaluate long-term trends in dissolved-solids loads and concentrations in streams from 1929 to 2019 ($n = 14$). Results indicate that large, widespread, and sustained downward trends in dissolved-solids concentrations and loads occurred over the last 50–90 years. For 12 of the 14 stream sites with significant downward change, median declines of -38% (range of -14% to -57%) and -40% (range of -9% to -65%) were observed for flow-normalized concentration and load, respectively. Steepest rates of decline occurred from 1980 to 2000, coincident with the initiation of salinity-control efforts in the 1980s. However, there was a consistent slowing or reversing of downward trends after 2000 even though salinity-control efforts continued. Significant decreases in salinity occurred as early as the 1940s at some streams, indicating that, in addition to salinity-control projects, changes in land cover, land use, and/or climate substantially affect salinity transport in the UCRB. Observed dissolved-solids trends are likely the result of changes to watershed-related processes, not due to changes in the streamflow regime.

Plain Language Summary Salinity, or dissolved salt, in the Colorado River Basin causes an estimated \$300 to \$400 million per year in economic damages in the United States. The Colorado River Basin Salinity Control Program implements and manages projects to reduce salinity, investing millions of dollars per year in improved irrigation systems, vegetation recovery, and other mitigation strategies. To inform and improve mitigation efforts, there is a need to better understand changes in salinity that occurred prior to the implementation of salinity-control projects in the 1980s. This study uses decades of water-quality measurements, collected as early as 1929, to explore salinity trends in the Upper Colorado River Basin from 1929 to 2019. Findings indicate that large, widespread, and sustained downward trends in salinity occurred over the last 50–90 years. Further, the timing and amount of salinity reductions suggest changes in land cover, land use, and/or climate, in addition to salinity control, substantially affect how salinity is transported to streams in the basin. Identifying the causes of dropping salinity levels will be important for water managers in the basin so they can anticipate future changes in salinity, develop more efficient salinity-control practices, and capitalize on natural processes that reduce salinity.

1. Introduction

The southwestern United States (US) relies on the Colorado River to sustain its ecosystems, communities, and economies. The Colorado River provides irrigation water to nearly 4.5 million acres of land, generates over 4,200 megawatts of hydroelectric power, and supplies water to over 35 million people in the United States and 3.3 million people in Mexico each year (U.S. Bureau of Reclamation, 2012, 2017). Furthermore, it is estimated that the Colorado River supports over 16 million jobs, with an annual economic benefit of over \$1.4 trillion (James et al., 2014).

Salinity, or dissolved solids, is an important water-quality constituent of concern in the Colorado River, causing an estimated \$300 to \$400 million per year in economic damages in the United States (U.S. Bureau of Reclamation, 2017). Increased salinity affects agricultural, municipal, and industrial sectors by reducing crop production, increasing water treatment costs, and damaging water supply infrastructure. Additionally, higher salinity makes it difficult for the United States to meet its legal water quality obligations for water delivered to Mexico (U.S. Bureau of Reclamation, 2017).

To address these challenges, the Colorado River Basin Salinity Control Forum was established in 1973 (Colorado River Basin Salinity Control Forum [CRBSCF], 2014) to enhance and protect the quality of water in the Colorado River for use in the United States and Mexico, in accordance with the 1972 Clean Water Act and the Salinity Control Act of 1974 (U.S. Bureau of Reclamation, 2017). To reduce or prevent salinity loading to streams, the Salinity Control Forum implements a variety of salinity-control measures, such as well fields designed to intercept brine from entering rivers (Chafin, 2003), improved irrigation infrastructure, vegetation management, land retirement, canal lining, and water-efficient irrigation systems that reduce surface runoff and infiltration through saline soils (Anning et al., 2010). To inform their mitigation strategies, previous studies have investigated dissolved-solids sources (Miller et al., 2017), transport processes (Cadaret et al., 2016; Rumsey et al., 2017), and the effectiveness of salinity-control efforts (Schaffrath, 2012; Thiros, 2017).

Dissolved solids in water occur naturally due to the weathering and dissolution of minerals in soils and rocks; however, various anthropogenic activities can increase dissolved-solids loading above natural levels (Anning et al., 2010). Geology, land cover, land-use practices, and climate are factors known to affect dissolved-solids loading to streams (Kenney et al., 2009). It is estimated that a significant portion of dissolved solids in the Colorado River are generated in the Upper Colorado River Basin (UCRB) due to the prevalence of geologic units that contain high amounts of salts (U.S. Environmental Protection Agency, 1971). Multiple studies conclude that the largest natural source of dissolved solids to streams in the UCRB is sedimentary rocks (Anning et al., 2010; Kenney et al., 2009; Miller et al., 2017), as soluble minerals within sedimentary rocks are easily eroded, dissolved, and transported to surface waters (Anning et al., 2010; CRBSCF, 2014). It is estimated that 62% of UCRB dissolved-solids loads originate from geologic sources (Miller et al., 2017). The primary anthropogenic source of dissolved solids in the UCRB is the runoff and groundwater discharge from irrigated agricultural lands, especially those occurring over marine sedimentary rocks. Between 32% and 45% of dissolved-solids loading in the UCRB is estimated to originate from irrigated agricultural lands, even though they cover only 2%–3% of the total land area (Anning et al., 2010; Iorns et al., 1965; Kenney et al., 2009; Keum & Kaluarachchi, 2015; Liebermann et al., 1989; Miller et al., 2017). The flow path water takes prior to reaching a stream also affects dissolved-solids loading, with the majority of dissolved-solids loads being generated in the subsurface when water picks up soluble minerals as it travels through sedimentary rocks prior to discharging to streams (Rumsey et al., 2017).

To improve understanding of dissolved-solids transport in the basin, and to inform salinity mitigation efforts now and in the future, it is necessary to understand dissolved-solids trends and underlying drivers of change, both natural and anthropogenic, that control dissolved-solids loading in the basin. Drivers of changing dissolved solids are complex and multiple agents of change have been active in the UCRB during the 20th century, including agriculture, grazing, wildfire, water development, resource extraction, climate, urbanization, and recreation (Copeland et al., 2017; Dennison et al., 2014; National Research Council, 2007). Furthermore, the UCRB continues to experience large increases in human population, expansion of energy development, changes in agricultural practices and land cover, and substantial climate variability (Buto et al., 2010; Cole et al., 1997; Copeland et al., 2017; Xiao et al., 2018). Because streams integrate and reflect upstream landscape conditions, measurements of water quality and streamflow over time can provide evidence of landscape-scale change. Thus, an important step toward understanding changing landscape drivers of dissolved-solids loading is to use multi-decadal stream water-quality records to quantify and characterize dissolved-solids change over time.

Downward trends in dissolved-solids concentrations and loads have been observed in rivers and streams across the UCRB from as early as the 1930s to the 1990s, where studies postulated that observed trends may have been affected by transbasin diversions, changes in land and water use, salinity-control activities, climate, and reservoir development (Kircher et al., 1984; Liebermann et al., 1989; Moody & Mueller, 1984;

Vaill & Butler, 1999). Butler (1996) identified decreasing trends in dissolved-solids loads and concentrations near Grand Valley, CO from 1970 to 1993 and concluded that trends were, in part, caused by salinity-control projects, but that natural or other anthropogenic effects in the UCRB likely also played a role in decreasing salinity. Decreases in dissolved solids observed upstream of salinity-control projects from 1970 to 1993 (Bauch & Spahr, 1998) and from 1986 to 2003 (Leib & Bauch, 2007), indicated that various watershed processes may have led to observed decreases, such as stream channel evolution, hydrologic variation, changing land-use practices, or fluctuations in groundwater discharge and quality. Additional trend analyses have specifically investigated the effects of salinity-control projects in the UCRB (Schaffrath, 2012; Thiros, 2017), finding that salinity reductions coincide with areas where projects have been implemented upstream. Overall, previous research has consistently shown decreasing dissolved solids in various parts of the UCRB during the 1900s. While some of the decrease is likely to be the result of salinity-control activities, evidence suggests that other processes in the basin also affect dissolved-solids loading and transport.

By analyzing long time scales, this study presents dissolved-solids trends that span fluctuations in climate, streamflow, land use, land cover, water development, and salinity-control practices in UCRB watersheds, with the overarching objective of using patterns in long-term trends to provide clues about possible drivers of dissolved-solids change. This study builds upon previous trend analyses by evaluating long-term trends in UCRB dissolved solids over the last 50–90 years, covering periods that precede the construction of large reservoirs and the implementation of salinity-control projects, and extend into the 2000s during the onset of widespread drought. Specific research questions were: (1) What are the directions and magnitudes of trends in dissolved-solids loads and concentrations in the UCRB during the 20th century? (2) What are the spatial and temporal patterns in regional trends? and (3) Are observed changes related to watershed processes or to changes in streamflow?

We employ a recently developed method for water-quality trend analysis: Weighted Regressions on Time, Discharge, and Season (WRTDS; Hirsch et al., 2010). This method identifies water-quality trends through time and removes the influence of interannual streamflow variability on trends, allowing for improved interpretation of results that can be used to begin parsing apart drivers of observed change. Results of this analysis provide new understanding of long-term dissolved-solids trends, their variability through time, and potential drivers of change that can be used to inform salinity mitigation efforts in the UCRB.

2. Methods

2.1. Study Area

The UCRB includes portions of Wyoming, Colorado, Utah, Arizona, and New Mexico (Figure 1), covering an area of roughly 294,000 km². The basin's headwaters are located in the Wind River Range of southwest Wyoming, the Rocky Mountains of central Colorado, and the Wasatch and Uinta Mountains of Utah. The outlet of the basin is located at Lee's Ferry, Arizona, downstream of Lake Powell on the Colorado River. Major rivers in the UCRB include the Colorado, Dolores, Duchesne, Green, Gunnison, San Juan, White, and Yampa Rivers.

Physiography and climate are diverse in the UCRB. Elevations range from 940 to 4,200 m (Liebermann et al., 1989; Figure 1d). The northern, eastern, and western borders of the basin are dominated by snow-covered mountain ranges, while the central and southern parts of the UCRB include semiarid intermontane basins and high elevation plateaus. The Rocky Mountains to the east receive large quantities of snow and are estimated to contribute as much as 70% of the annual streamflow to the Colorado River (Christensen et al., 2004). Snowmelt from these mountainous areas during the spring and early summer causes significant runoff, while baseflows dominate from late summer to early spring (Warner et al., 1985). Lower elevation, semiarid, or arid watersheds receive much less precipitation and do not generate significant amounts of streamflow (Lieberman et al., 1989). Mean annual precipitation is roughly 380 mm, ranging from 130 mm/yr in the Colorado Plateau to 1490 mm/yr in the Rocky Mountains (for water years 1930–2014; Figure 1c). Temperatures vary from −3 °C to 15 °C (Figure 1b), with a mean annual temperature of 7 °C (for water years 1930–2014; extended temperature and precipitation data set as presented in Xiao et al., 2018, based on methods from Hamlet & Lettenmaier, 2005).

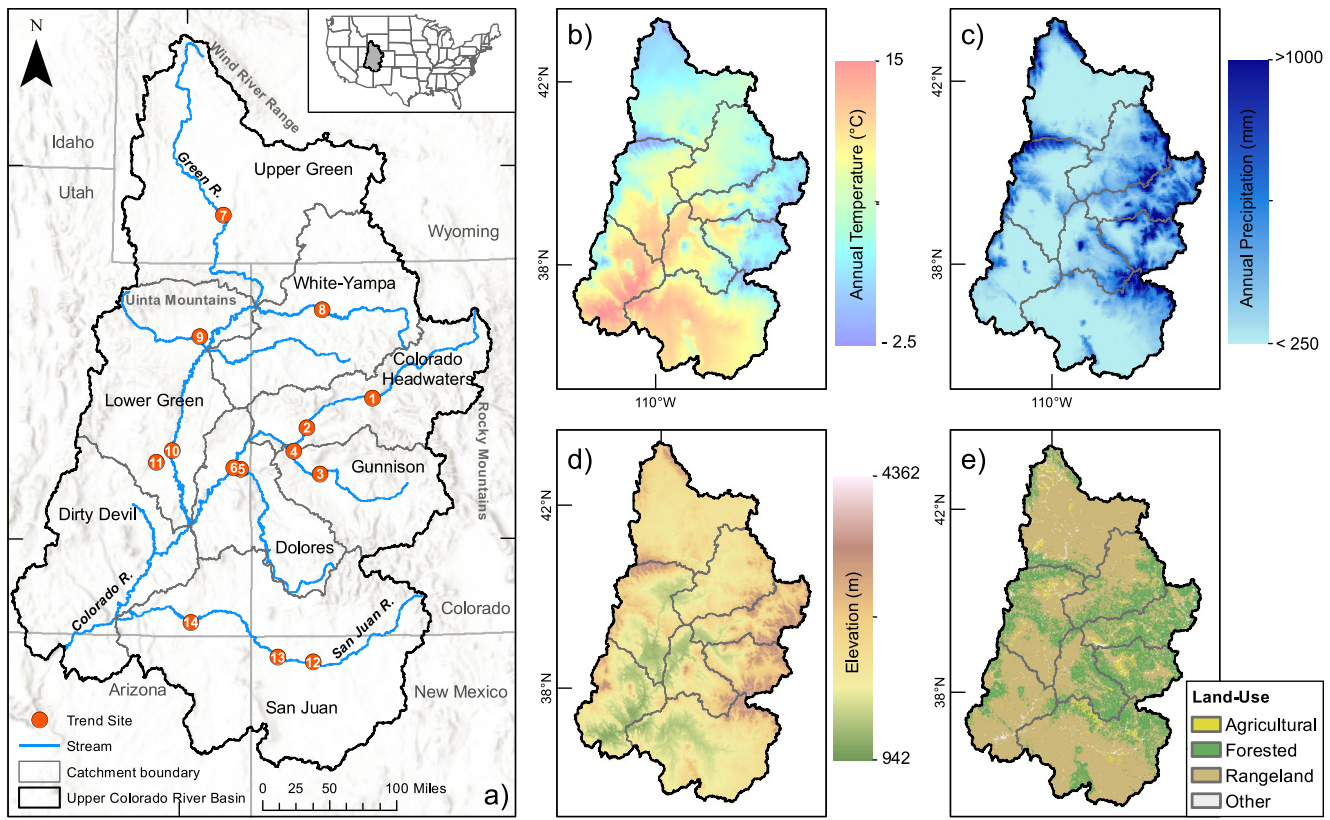


Figure 1. Study area of UCRB, including (a) locations of 14 sites selected for long-term trend analysis (numbers correspond to sites listed in Table 1), (b) average annual temperature, (c) average annual precipitation, (d) elevation (Farr & Kobrick, 2000), and (e) land-use. Credits for base map in panel (a): Map image is the intellectual property of Esri and is used herein under license. Copyright © 2014 Esri and its licensors. All rights reserved. UCRB, Upper Colorado River Basin.

UCRB hydrogeologic conditions are complex. Groundwater recharge occurs primarily in high elevation areas (Warner et al., 1985), which are dominated by igneous and metamorphic rocks. Sedimentary rocks occur throughout the rest of the basin and include sandstone, siltstone, shale, and occurrences of evaporite. Several widespread formations that were deposited in marine or brackish environments contain saline minerals that today are easily eroded, dissolved, and transported to surface waters (Anning et al., 2010; CRBSCF, 2014). Marine shales, salt anticlines, exposed evaporite deposits, coal-bearing formations, and lacustrine deposits are all geologic features known to contribute high amounts of dissolved solids to UCRB streams (Liebermann et al., 1989).

2.2. Long-Term Streamflow and Dissolved-Solids Data Sets

Long-term trends in annual dissolved-solids loads and concentrations were evaluated using daily streamflow and discrete dissolved-solids concentration data from the US Geological Survey (USGS) National Water Information System (NWIS; U.S. Geological Survey, 2021, <http://waterdata.usgs.gov/nwis>). “Daily” and “discrete” refer to data collected every day and periodically, respectively, within a given period of record. To maximize dissolved-solids records, long-term dissolved-solids data sets combined all available dissolved-solids measurements determined using a variety of methods at each site, including the sum of dissolved constituents (SUM; USGS parameter code 70301), residue on evaporation at 180 °C (ROE; USGS parameter code 70300), and specific conductance (SC; USGS parameter codes 00095 and 90095). SUM is the summation of concentrations of major cations and anions in a water sample; ROE is obtained by drying filtered (0.45 μm) water samples at 180 °C and measuring the mass of dried residue per known volume of water; and SC is obtained using a sensor to measure the capacity of water to conduct an electrical current, which is a function of the amount and type of dissolved species in a water sample (Radtke et al., 2005). Because ROE

measurements may be skewed high due to trapped water in minerals, and because measurements of SC are in units of $\mu\text{S}/\text{cm}$, both parameters were adjusted to SUM values using linear regression models in order to ensure consistency among dissolved-solids measurements. SUM concentrations were used as reported in NWIS. Dissolved-solids concentrations below the method detection limit were not included.

Site-specific linear regression models estimated SUM as a function of ROE or SC and decimal time (year + fraction of the year [day of the year from January 1/365]). Prior to generating models, dissolved-solids data were trimmed to match the period of trend analysis (see Section 2.3 for criteria used to determine period of analysis; Table 1). Dissolved solids outliers were identified using ratios of SUM/SC and SUM/ROE and removed prior to developing linear regression models to avoid skew from extreme measurements. Outliers were defined for SUM/SC as values greater than 1 or less than the first percentile of values falling below 1.5 times the interquartile range. Outliers for SUM/ROE were values greater than 1.2 (Tillman & Anning, 2014) or less than the first percentile of values falling below 1.5 times the interquartile range. The percent of paired SUM/SC or SUM/ROE values removed was less than 1.2% and 2%, respectively. Residual standard errors of all models ranged from 0.03 to 0.07 (model coefficients and fit statistics are included in the supporting information [SI], Tables S1a and S1b). After SC and ROE were converted to SUM concentrations, all dissolved-solids concentrations were combined to build long-term dissolved-solids data sets at each site. If multiple dissolved-solids measurements were available on a given day, they were selected preferentially as SUM values first (direct measurement of dissolved solids), ROE values second (direct measurement, but can be skewed by mineral composition), and SC values third (indirect measurement).

In addition to combining multiple dissolved-solids parameters to generate long-term data sets, it was also necessary to include dissolved-solids concentrations collected using composite sampling, which was a method of sample collection used in the UCRB prior to 1978. Values reported for this type of sampling are a single concentration representing a multi-day time period, ranging from 2 to 10 days. Trend results from WRTDS simulations comparing composite and discrete dissolved-solids data sets showed that annual estimates of load and concentration were generally within 10% of each other, verifying that the difference between the two sampling methods was low enough to provide comparable estimates from WRTDS (Table S2 and Section S2.1 of SI).

2.3. Site Selection

To ensure a robust trend analysis, selected long-term water-quality records were required to meet a set of criteria, many of which are based on Oelsner et al. (2017). To analyze salinity change over long time scales, dissolved-solids data were required to begin in water year 1960 or earlier and needed at least 50 years of dissolved-solids data. To ensure robust representation of trends throughout the period of analysis, no data gaps larger than 6 years were allowed and at least 70% of years in the period of record needed at least quarterly sampling. To reduce uncertainty in the trends at the start and end of the record, at least 6 years of uninterrupted annual dissolved-solids data were required at the beginning and end of the record, and at least quarterly sampling was required for the first and last 2 years of the record. Finally, because streamflow cannot be used as a predictor of water-quality fluctuation immediately downstream of reservoirs, sites were located at least 10 km downstream of a dam. After applying these criteria, 14 sites had adequate data sets to perform WRTDS analysis (Figure 1a). The maximum period of record meeting the above criteria was used as the period of trend analysis at each site. Site-specific periods of record ranged from 54 to 91 years, with data spanning from 1929 to 2019 (Table 1). All years referred to in this paper are water years (1 October of the previous calendar year to 30 September). All 14 sites used discrete dissolved-solids concentration data (SUM or converted SC or ROE) and all water-quality and streamflow monitoring stations were collocated. While daily SC data were available at many sites, it often started several decades after the period of discrete data began. In an effort to avoid model bias that may have been introduced by inconsistent sampling frequencies, we did not use daily SC in the long-term trend analysis.

2.4. WRTDS Trend Analysis of Dissolved-Solids Loads and Concentrations

The recently extended methodology of WRTDS (EGRET v3.0; Hirsch, De Cicco, Watkins, et al., 2018) was used to evaluate long-term trends in dissolved-solids loads and concentrations in the UCRB. WRTDS uses

Table 1
Long-Term Dissolved-Solids Trends for 14 Sites in the UCRB

Map ID (Figure 1)	USGS station ID	USGS station name	Site ID	Drainage area, in km ²	Start year of trend	End year of trend	Starting year FN concentration, in mg/L	Trend in FN concentration, in mg/L (percent)	Starting year FN load, in tonnes/yr	Trend in FN load, in tonnes/yr (%)
1	09085000	Roaring Fork River at Glenwood Springs, CO	Roaring Fork	3,763	1960	2019	331	8 (2.4)*	244,000	14,100 (5.7)***
2	09095500	Colorado River near Cameo, CO	CR-Cameo	20,684	1934	2019	603	-86 (-14.3)***	1,352,000	-123,300 (-9.2)***
3	09149500	Uncompahgre River at Delta, CO	Uncompahgre	2,885	1960	2019	1,752	-827 (-47.2)***	350,000	-108,400 (-31.1)***
4	09152500	Gunnison River near Grand Junction, CO	Gunnison	20,520	1932	2019	1,108	-593 (-53.5)***	1,449,000	-599,300 (-41.3)***
5	09180000	Dolores River near Cisco, UT	Dolores	11,862	1953	2019	1,897	-1083 (-57)***	428,000	-234,800 (-54.9)***
6 ⁺	09180500	Colorado River near Cisco, UT	CR-Cisco	62,419	1929	2019	1,015	-409 (-40.3)***	4,115,000	-1,588,200 (-38.6)***
7	09217000	Green River near Green River, WY	GR-Green River WY	36,260	1952	2018	421	-154 (-36.3)***	475,000	-113,400 (-23.8)***
8	09251000	Yampa River near Maybell, CO	Yampa	8,762	1951	2019	246	19 (7.7)*	181,000	38,900 (21.5)*
9	09302000	Duchesne River near Randlett, UT	Duchesne	9,816	1957	2019	1,065	-368 (-34.6)***	337,000	-200,700 (-59.6)***
10 ⁺	09315000	Green River at Green River, UT	GR-Green River UT	116,161	1929	2019	553	-161 (-29.1)***	2,159,000	-661,100 (-30.6)***
11	09328500	San Rafael River near Green River, UT	San Rafael	4,217	1951	2018	3,283	-1374 (-41.9)***	205,800	-132,900 (-64.5)***
12	09364500	Animas River at Farmington, NM	Animas	3,522	1956	2011	495	-151 (-30.3)***	203,000	-54,800 (-26.6)***
13	09368000	San Juan River at Shiprock, NM	SJ-Shiprock	33,411	1959	2012	562	-258 (-45.9)***	605,000	-284,900 (-47.1)***
14 ⁺	09379500	San Juan River near Bluff, UT	SJ-Bluff	59,570	1930	2019	637	-222 (-35)***	1,011,000	-519,600 (-51.4)***

Notes. Each site has a different period of record, but all records span 1960 (or earlier) to at least 2010. All data are from USGS (⁺denotes most-downstream main-stem site for each sub-basin; likelihood levels denoted by: *** for $\hat{\pi} \geq 0.95$ [highly likely]; ** for $0.90 \leq \hat{\pi} < 0.95$ [very likely]; * for $0.67 \leq \hat{\pi} < 0.90$ [likely]). Abbreviations: FN, flow-normalized; UCRB, Upper Colorado River Basin; USGS, US Geological Survey.

a multivariate smoothing approach to interpret the behavior of water quality as it changes with time and in relation to stream discharge. Using discrete water quality and daily mean discharge data, WRTDS defines a concentration-discharge (C-Q) relationship for every day in the period of record, allowing the C-Q relationship to change gradually with season and time (Choquette et al., 2019), using the following equation,

$$\ln(C) = \beta_0 + \beta_1 t + \beta_2 \ln(Q) + \beta_3 \sin(2\pi t) + \beta_4 \cos(2\pi t) + \varepsilon$$

where

\ln is the natural log,

C is daily concentration,

β are fitted coefficients,

t is time,

Q is mean daily discharge, and

ε is unexplained residual.

Water-quality measurements are weighted in the model depending on their distance in time, season, and discharge to a given date when the C-Q relationship is being estimated (half-window widths to determine weights were 7 years for time, 2 for $\ln(Q)$, and 0.5 years for season). WRTDS combines daily C-Q relationships with measured daily discharge to estimate a daily concentration and load, which, in this study, are then aggregated to annual time scales to obtain annual mean estimates. Analyses were conducted using the EGRET R package (Hirsch, De Cicco, Watkins, et al., 2018). See Hirsch et al. (2010) and Hirsch and De Cicco (2015) for a detailed explanation of the WRTDS approach.

Because annual mean concentration and load estimates can be strongly influenced by streamflow conditions, WRTDS integrates the C-Q relationship over the observed probability distribution of stream discharge (pdf) to provide estimates of changes in “flow-normalized” (FN) concentration and load over time (Choquette et al., 2019). This approach removes the effect of year-to-year variations in streamflow, such as high- or low-streamflow conditions in a given year that are largely driven by variable weather, to provide FN values that show gradual change over time (Hirsch et al., 2010). Plots of annual mean and FN annual concentrations and loads (see Figures S4a and S4b in SI) illustrate the contrast between year-to-year variability of annual values and the smoothed FN trend.

In some regions, such as the UCRB, the probability distribution of streamflow may have changed over the study period due to water development, climate, or other factors. To accommodate this, WRTDS allows for nonstationarity in the distribution of streamflow when conducting flow-normalization by using a moving time window to define the streamflow pdf for any given set of years. Referred to as “generalized flow-normalization,” this allows the trend analysis to account for long-term, gradual changes in the streamflow regime that occur over time (Choquette et al., 2019) but not be excessively influenced by the year-to-year variability of streamflow, which can be quite large in this region (Hirsch et al., 2010). The non-stationary discharge window used in this study is 25 years. To evaluate the magnitude and direction of trend over the period of record, annual FN loads (or concentrations) from the starting year are subtracted from annual FN loads (or concentrations) in the ending year ($FN_{\text{end}} - FN_{\text{start}} = \text{water-quality trend}$).

The application of generalized flow-normalization allowed for the parsing of trends into two types of drivers: (1) trends driven by systematic changes in the streamflow regime and (2) trends driven by changes in the C-Q relationship (i.e., the portion of the total trend related to constituent sources, watershed and landscape processes, and management activities). The C-Q relationship component of change, called the “CQ Trend Component” (CQTC) is calculated as the estimated change in flux or concentration between the starting and ending years assuming stationary discharge over the entire period of record ($CQTC_{\text{end}} - CQTC_{\text{start}} = CQTC$). The only source of change in this computation is a change in the C-Q relationship (Choquette et al., 2019; Murphy & Sprague, 2019). The streamflow component of change, the “Q Trend Component” (QTC), is calculated as the difference between the generalized flow-normalized trend (accounting for non-stationary streamflow discharge) and the CQTC ($\text{water-quality trend} - CQTC = QTC$). This

component of the overall trend is attributable to systematic change in the streamflow regime, capturing changes in seasonal patterns as well as changes in the timing and magnitude of streamflow that have occurred. Calculating CQTC and QTC trend components provided a first step toward understanding possible drivers of dissolved-solids trends in the UCRB.

WRTDS (EGRET v3.0) also accommodates abrupt changes in the streamflow regime or C-Q relationship that may result from abrupt changes in the watershed. Abrupt changes in the streamflow regime may be caused by the construction of a dam or water development projects, whereas abrupt changes in the C-Q relationship might be caused by the implementation of a point-source pollution control strategy. For these situations, WRTDS allows for a break in the trend model, defined as the “C-Q wall” for an abrupt change in the C-Q relationship, and as a “flow-break” for an abrupt change in the distribution of streamflow (Hirsch, De Cicco, Watkins, et al., 2018). Each are assigned to a specific day in the record where the abrupt change occurred. For this study, one C-Q wall was used for the Dolores River near Cisco, Utah (Dolores), on January 30, 1980 at the onset of the Paradox Valley Unit, a point-source treatment program that diverts groundwater brine away from the Dolores River.

While almost all selected stream sites were also affected by the construction of large dams during the period of record, no C-Q walls or flow-breaks were used to represent abrupt changes from dams. Dam construction and reservoir filling is a multi-year process resulting in gradual changes to water-quality conditions, as is demonstrated by (1) model residuals that are randomly distributed through time and (2) discrete dissolved-solids concentrations that do not show step changes after dam construction. Generalized flow-normalization and the flexible modeling framework of WRTDS accommodate the gradual changes in streamflow distribution and C-Q relationship that occurred at these sites as a result of multi-year dam construction and reservoir filling activities, providing a more realistic representation of water-quality change than an abrupt shift on a single day.

Uncertainty in estimated trends was evaluated with the WRTDS Bootstrap Test (WBT) using a block bootstrap procedure described in Hirsch et al. (2015), with the exception that generalized flow-normalization was used in this analysis. This approach resamples the data set with replacement, using time blocks of 300 days. Confidence intervals (90%) and the statistical likelihood of the predicted trend direction were determined using 200 bootstrap replicates, where statistical likelihood is estimated as the number of positive (or negative) bootstrap replicates divided by the total number of bootstrap replicates. The EGRETci R package version 2.0 was used to implement the WBT analysis (Hirsch, De Cicco, & Murphy, 2018).

The end result of WRTDS and WBT includes annual mean and FN annual estimates of concentration and load, the change in FN load, the change in FN concentration, and the statistical likelihood ($\hat{\pi}$) that the direction of the estimated trends is correct. Likelihoods can be used to differentiate varying degrees of confidence and, following Hirsch et al. (2015), this study considered trend directions “likely” ($0.67 \leq \hat{\pi} < 0.9$), “very likely” ($0.90 \leq \hat{\pi} < 0.95$), or “highly likely” ($\hat{\pi} \geq 0.95$), based on their respective likelihood. Using likelihood statements provides decision makers with additional information about the degrees of statistical certainty, which they can incorporate when making decisions about future actions.

2.5. Uncertainty of Estimates in Recent Years

Like most regression-based techniques, annual and FN annual load and concentration estimates are subject to change when a new year of water-quality data are added to a record (Lee et al., 2017). This affects the certainty of estimates in years near the end of the period of record. A sensitivity analysis was conducted to recalibrate WRTDS with data that were incrementally extended by 1 year, revealing that annual estimates of load and concentration in years near the end of the period of record had much greater variability than annual estimates not near the end of the record. As such, annual estimates near the end of the calibration period should be considered “provisional.” For dissolved-solids loads and concentrations in the UCRB, annual mean estimates are considered provisional during the last 5 years in the period of record; for FN annual estimates, the last 10 years in the period of record are considered provisional since the incremental addition of new streamflow data affect FN estimates further back in time (sensitivity analysis provided in SI, Section S2.2, Figures S2 and S3).

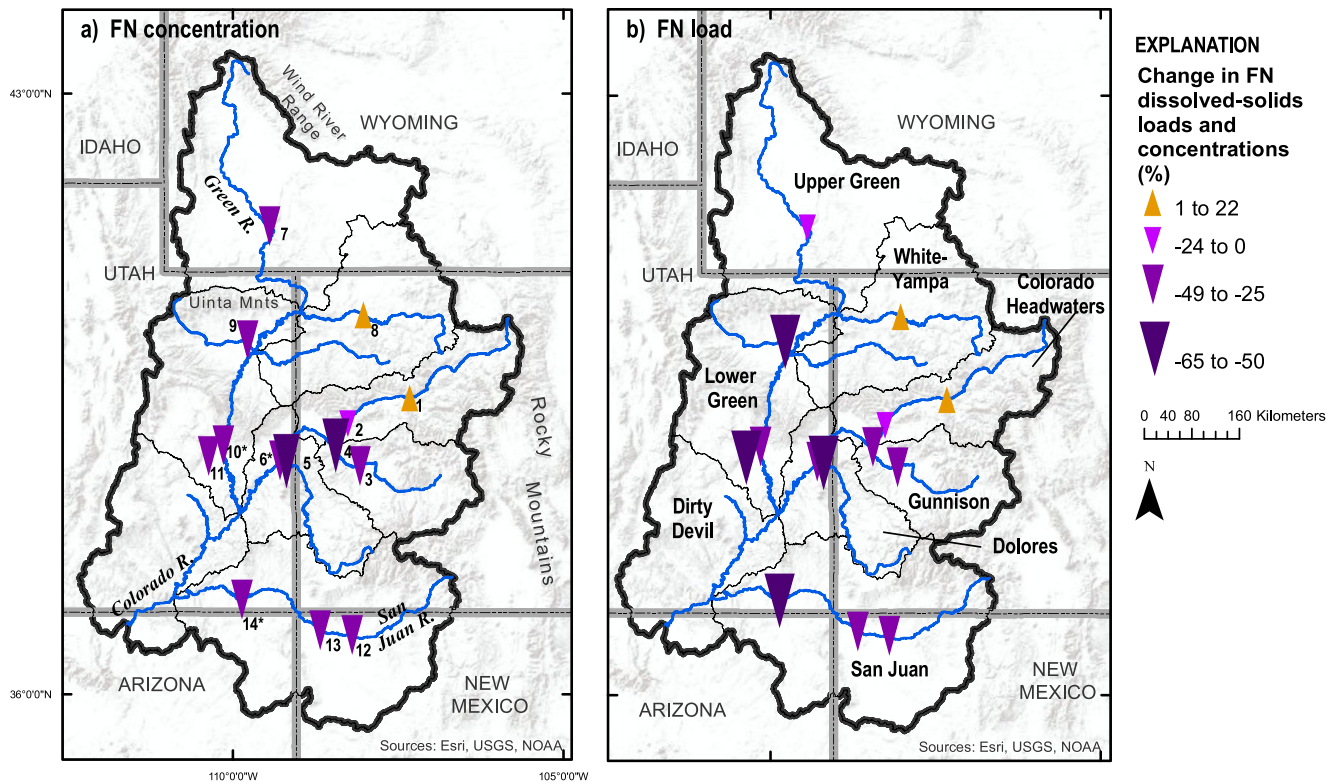


Figure 2. Long-term dissolved-solids trends, presented as percent change, in (a) FN concentration and (b) FN load for 14 sites in the UCRB spanning years 1929–2019. Trends calculated using the starting and ending years of the site-specific period of record; across sites record lengths varied from 54 to 91 years, with a median length of 67.5 years. Twelve of 14 records ended in 2018 or 2019. Records at the Animas and SJ-Shiprock sites ended in 2011 and 2012, respectively. Sites with stars denote the most-downstream main-stem sites. Site information, including starting and ending years of dissolved-solids trends at each site, is included in Table 1. Credits for base map: Map image is the intellectual property of Esri and is used herein under license. Copyright © 2014 Esri and its licensors. All rights reserved. FN, flow-normalized; UCRB, Upper Colorado River Basin.

3. Results and Discussion

Substantial and widespread downward trends in dissolved-solids loads and concentrations occurred in the UCRB over the period spanning 1929 to 2019, with decreases observed as early as 1940 at some sites. Patterns of dissolved-solids trends varied through time and by sub-basin, providing insight about possible drivers of salinity change. The majority of downward trends are estimated to be the result of watershed-related processes, not changes in the streamflow regime. Subsequent sections present the magnitude and extent of dissolved-solids change, explore the spatial and temporal differences in trend patterns, and discuss possible drivers of salinity change.

3.1. Long-Term Dissolved-Solids Trends

Decreasing trends in FN dissolved-solids concentrations were highly likely at 12 of the 14 sites (Figure 2a), with median declines of -38% (range of -14% to -57% ; Table 1). The largest magnitude decreases in FN dissolved-solids concentration occurred at the Dolores and Gunnison sites, where declines were greater than 50% . Likely increases in FN concentration were observed at the Yampa ($+8\%$) and Roaring Fork ($+2\%$) River sites, two of the most-upstream sites draining the Colorado Rocky Mountains. Notably, the magnitudes and prevalence of increasing trends were much less than decreasing trends. Substantial declines in dissolved-solids concentration occurred in all three main-stem river sub-basins, where median percent decreases in FN concentration were -47% , -35% , and -34% for the Colorado ($n = 5$), Green ($n = 4$), and San Juan ($n = 3$) River sub-basins, respectively. Average rates of decrease in FN concentration varied from -1 to -20 mg/L per year, depending on the site.

Similarly, large and widespread downward trends in FN dissolved-solids loads were also observed. Decreases in FN load were highly likely at 12 out of 14 sites (Figure 2b), with median declines of -40% (range of -9 to -65% ; Table 1). The San Rafael, Duchesne, Dolores, and SJ-Bluff sites had the highest percentages of downward change in FN load, all with decreases greater than 50% . Notably, these four sites also had the highest rates of decreasing streamflow over their period of record (Table S3; Figure S6). As with FN concentration, likely or highly likely increases in FN dissolved-solids loads were observed at sites on the Yampa ($+22\%$) and Roaring Fork ($+6\%$) Rivers. Substantial decreases in FN dissolved-solids load were observed in all three main-stem river sub-basins; median percent decreases were -39% , -45% , and -47% , for the Colorado ($n = 5$), Green ($n = 4$), and San Juan ($n = 3$) River sub-basins, respectively. Average rates of decreasing FN dissolved-solids yield (load normalized to drainage area) varied from -0.05 to -0.63 tonnes/km²/year.

Results corroborate many other studies that report significant decreases in dissolved-solids loads and concentrations in the UCRB during the 1900s, when widespread decreasing trends were observed for the Green River sub-basin before 2013 (Thiros, 2017; Vaill & Butler, 1999), for the Colorado River headwaters sub-basin prior to 2003, with many decreasing trends occurring upstream, or prior to the implementation, of salinity-control projects (Bauch & Spahr, 1998; Butler, 1996; Leib & Bauch, 2007; Liebermann et al., 1989; Vaill & Butler, 1999), and for the San Juan River sub-basin pre-1983 (Liebermann et al., 1989; Moody & Mueller, 1984). In a recent assessment of national trends, Oelsner et al. (2017) reported six out of seven UCRB sites had decreasing trends in dissolved solids from 1972 to 2012. Placed in the context of the entire US, the UCRB showed some of the most consistent decreases in dissolved-solids trends out of any region; in contrast, many areas in the United States show increases in dissolved solids over time (Oelsner et al., 2017).

3.2. Drivers of Observed Trends—Watershed Processes Versus Streamflow Regime

Parsing observed dissolved-solids trends into contributions from changes in watershed processes (CQTC) versus changes in the streamflow regime (QTC) provides an initial step toward determining drivers of change. The CQTC component of the trend represents water-quality changes related to point and non-point sources in the watershed, which can be affected by land and water management. In the context of dissolved solids in the UCRB, this might be improved irrigation infrastructure, vegetation recovery, or point-source treatment of saline groundwaters and springs. Alternatively, the QTC component of the trend represents water-quality changes influenced by climate, water management, or other alterations leading to sustained, long term changes in the streamflow regime (Choquette et al., 2019; Murphy & Sprague, 2019). Comparing patterns in CQTC and QTC provides insight into whether watershed processes or streamflow regimes most affect dissolved-solids trends in the UCRB.

Watershed processes (CQTC) are estimated to be the primary cause of decreasing trends in dissolved-solids concentration and loads observed at most sites in the UCRB (Figure 3). The trend in FN load at the SJ-Bluff site represented the only site where CQTC and QTC were roughly equal. In 12 cases (five for concentration and seven for loads), both the QTC and the CQTC contributed to decreases in dissolved solids, indicating that changes in the streamflow regime led to larger decreases in dissolved solids than would have occurred without a change in the variability of streamflow over time. In five cases (two for concentration and three for loads), QTC was opposite the overall trend direction, indicating that changing watershed processes were so effective in lowering dissolved solids as to cancel out potential increases that may have occurred as a result of changing streamflow regimes.

The small magnitude of QTC at most sites suggests there has been little to no change in the streamflow regime, or that changes in the streamflow regime had a minor effect on dissolved-solids concentration and loads. The latter is more likely since widespread construction of reservoirs and other modifications to streamflow during the period of analysis led to widely and substantially altered distributions of annual streamflow in the UCRB. Many sites on the Green, Colorado, and San Juan Rivers display alterations in the hydrograph indicative of reservoir influence, where minimum to low-flows increased and high to max-flows decreased (see Section S2.3 and Figure S5 in SI). This is consistent with the conceptual understanding of how reservoirs operate, storing large volumes of streamflow from snowmelt runoff during spring months and releasing higher-than-normal volumes of water downstream in low-flow months, ef-

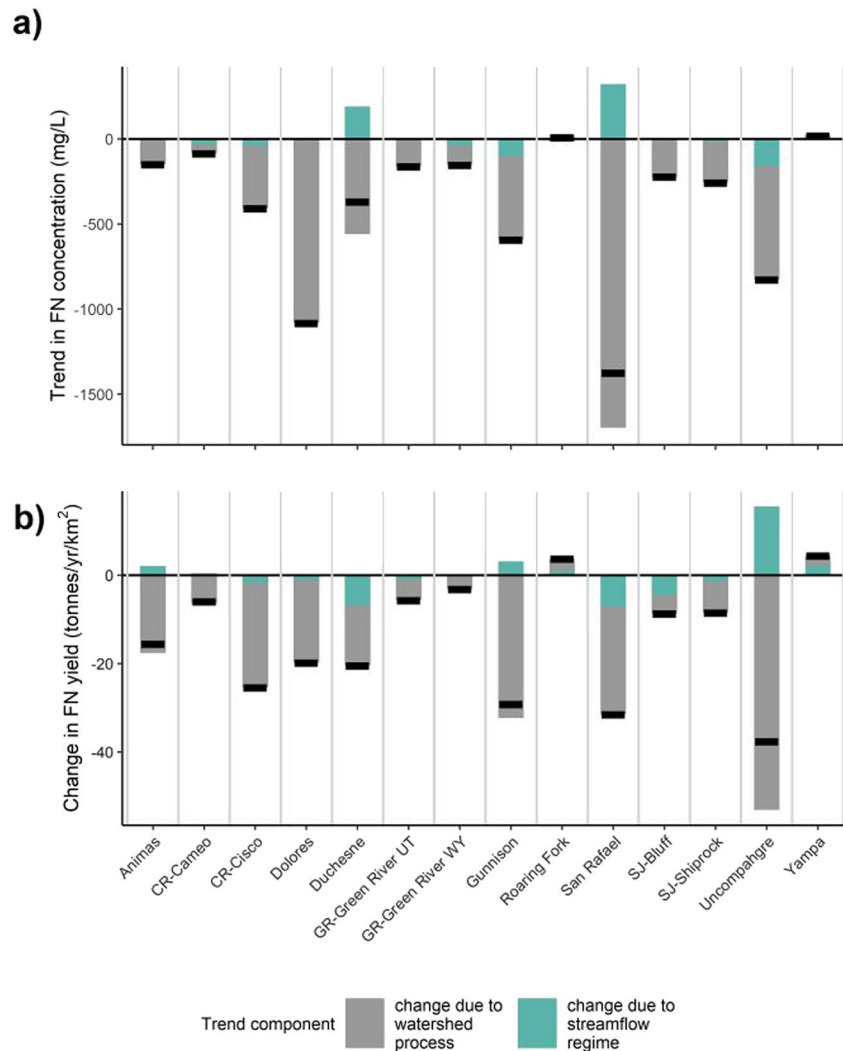


Figure 3. Long-term dissolved-solids trend components in (a) FN concentration and (b) FN yield (load/drainage area). Black rectangles show overall total trend. Trends span 1929–2019, but periods of record vary at each site. See Table 1 for site-specific periods of record. FN, flow-normalized.

fectively lowering high streamflows and raising low streamflows at downstream sites. All sites except the Roaring Fork and Yampa sites clearly reflect this shift (Figure S5 and Section S2.3 in SI) and were affected by reservoir construction during the period of analysis. As observed by Moody and Mueller (1984), reservoir-induced alterations to the streamflow distribution affect the distribution and timing of dissolved-solids transport in the basin, diluting dissolved-solids concentrations in low-flow months and increasing dissolved-solids concentrations in high runoff months, acting to confound drivers of trend by changing the seasonality of dissolved-solids transport. In addition to reservoirs, climatic changes, shifts in water use, water diversions, or other factors may also contribute to observed shifts in streamflow regimes and are important to consider as they may affect when and how dissolved solids are transported through UCRB waterways.

In spite of dramatic changes in the streamflow regime over the period of record, QTC estimates suggest that these changes had a limited effect on long-term trends in FN dissolved-solids concentrations and loads over the last 50–90 years (Figure 3). One potential explanation for this surprising result is that trends in streamflow (Figure S5) shifted the C-Q relationship over time and these effects are captured in the CQTC (Murphy & Sprague, 2019); thus the QTC may not be capturing the full effect of streamflow trends on long-term

changes in water quality. Alternatively, QTC estimates may vary in the positive or negative direction over the period of record and seasonally, causing smaller than expected QTC estimates when aggregated annually and compared between the start and end years of the record. For example, shifts in observed streamflow regimes (Figure S5) may have affected dissolved solids on a seasonal basis, altering the timing of dissolved-solids transport without having a measurable effect on annual streamflow volumes or estimates of FN loads and concentrations. Additionally, decadal estimates of QTC versus CQTC indicate positive QTCs in some decades may be offset by negative QTCs in other decades (Figure S7), effectively canceling out over time when changes are aggregated over the period of record. Further investigation would be required to determine how changing streamflow affects UCRB dissolved solids due to the confounding nature of these relationships.

3.3. Spatial and Temporal Patterns of Dissolved-Solids Change

Comparing monotonic trends in FN concentration and load between starting and ending dates is useful for quantifying the magnitude of change that occurred between the past and present; however, it ignores the nonmonotonic and nonlinear patterns of trend that occurred over the last 50–90 years that can provide clues about potential drivers of change. Temporal patterns in FN annual dissolved-solids concentrations and loads reveal that certain periods of time have undergone higher rates of change than others (Figure 4), with some sites showing decreases in dissolved solids as early as the 1940s. Furthermore, rates of change varied regionally across the UCRB and provide insight into the spatial scale of watershed processes needed to cause dissolved-solids change (Figure 5).

To compare regional patterns of change through time, the period of record at each site was split into roughly 20-year sub-periods: pre-1960, 1960 to 1980, 1980 to 2000, and post-2000. Results for each sub-period were extracted from the full period of record WRTDS model at each site. Sub-periods coarsely represent decades before the construction of large dams (pre-1960), the sub-period mostly affected by the construction of large dams (1960–1980), the sub-period when the salinity-control program was initiated across the basin (1980–2000), and recent decades (post-2000) when ongoing salinity-control efforts coincide with the onset of drought in the early 2000s.

In the Green River sub-basin, decreasing FN dissolved-solids loads and concentrations were observed at four of the five sites over their periods of record (range from 63 to 91 years; Figure 4a; Table 1). Four of the five sites are independent tributaries, with the most-downstream main-stem site, GR-Green River UT, integrating the Duchesne, GR-Green River WY, and Yampa sites. The Duchesne and San Rafael sites followed similar downward patterns in annual FN concentration, FN load, and annual mean streamflow (Figure 4a) over their periods of record, suggesting there may be common watershed and/or streamflow processes occurring along the Wasatch Plateau where these rivers originate. Similar downward trends also occurred for the main-stem sites of GR-Green River WY and GR-Green River UT, even though several tributaries flow into the Green River between these two sites. Unlike most sites, the Yampa site showed gradual increases in FN concentration and load, having distinct patterns of change that appear unrelated to other sites in the Green River sub-basin (Figure 4a). Decreases in dissolved solids did not occur at most Green River sites during pre-1960 or 1960 to 1980 sub-periods, although decreasing FN loads at the Duchesne site and decreasing FN concentrations at the GR-Green River UT site were observed from 1960 to 1980 (Figure 5). Lieberman et al. (1989) also did not observe widespread changes in dissolved solids prior to 1983. Overall, the steepest decreases in FN load in this sub-basin occurred from 1980 to 2000; steepest decreases in FN concentration occurred from 2000 to 2019 (Figures 4a and 5). Decreases in annual mean streamflow occurred at the Duchesne and the San Rafael sites (Figure 4a and Table S3).

For the Colorado River headwaters sub-basin, declines in FN dissolved-solids loads and concentrations were observed in five out of six sites during their periods of record (ranging from 60 to 91 years; Figure 4b; Table 1), with most sites showing similar patterns through time. Similarities in trends may result from shared loads and streamflow occurring at several nested sites in this sub-basin, where the CR-Cameo, Gunnison, and Dolores sites represent flow into the most-downstream site of CR-Cisco, and the Uncompahgre and Roaring Fork sites represent flow into the Gunnison and CR-Cameo sites, respectively. As an example, trends at the CR-Cisco site tended to closely match those at the Gunnison site, which contributes,

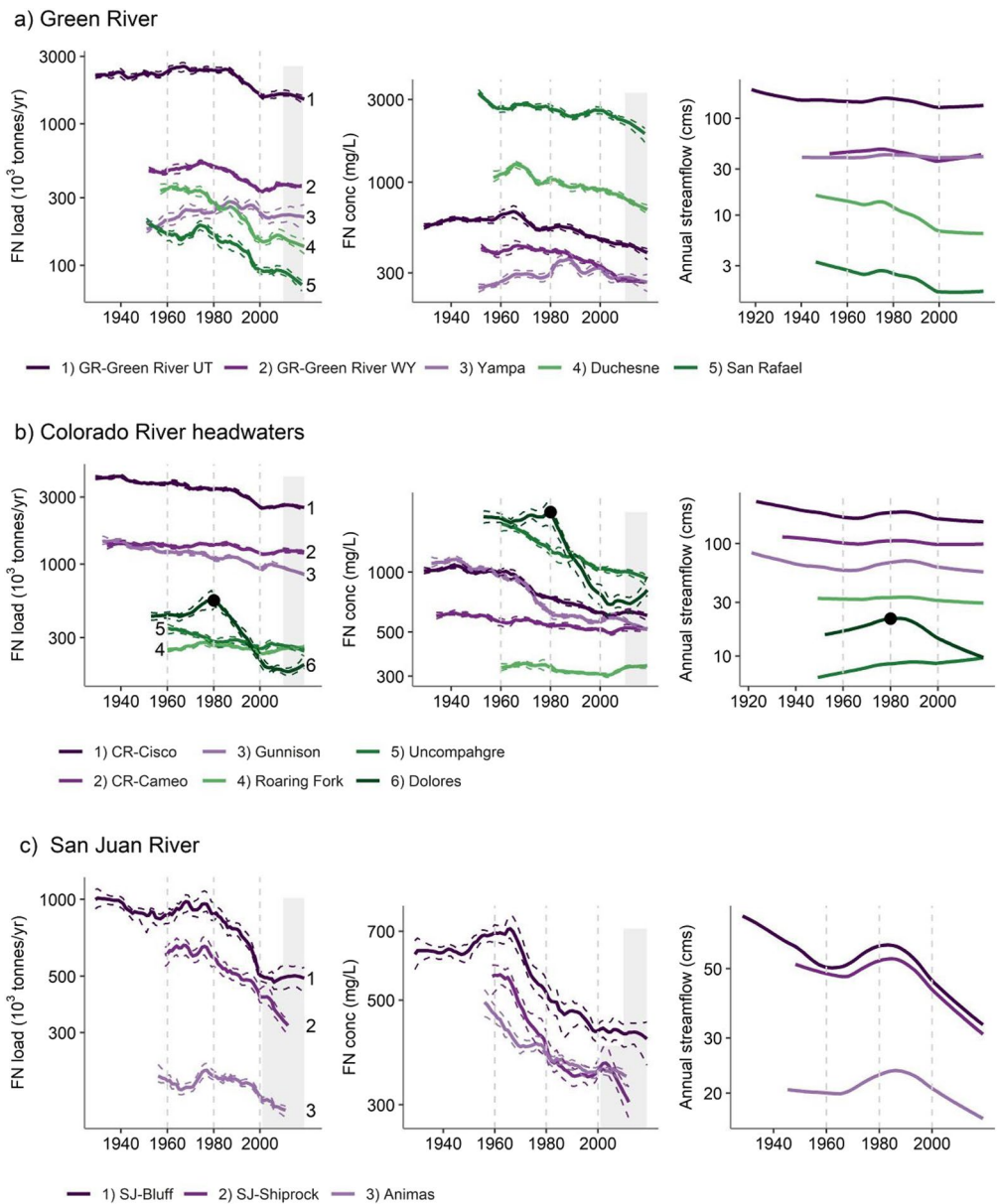


Figure 4. FN load, FN concentration, and the loess smooth of annual mean streamflow for stream sites in the (a) Green River, (b) Colorado River headwaters, and (c) San Juan River sub-basins from as early as 1929 to 2019. Y-axis is log-scale. Solid lines are FN annual values or loess smooth; dashed lines are the 90% confidence interval of FN annual values; black dots indicate when a “C-Q wall” was used in WRTDS. The shaded gray areas indicate where the estimates are considered provisional. Vertical dashed lines denote sub-periods. CR, Colorado River; FN, flow-normalized; GR, Green River; SJ, San Juan River.

on average, 35% of streamflow to the Colorado River above Cisco (CR-Cisco). Notably, downward trends in FN dissolved-solids loads and concentrations at these two sites began as early as the 1940s. As in the Green River sub-basin, some of the steepest declines in FN loads occurred from 1980 to 2000 (Figure 5). The steepest decreases in FN concentration were observed during 1960–1980 for the Gunnison, CR-Cisco, and Uncompahgre sites, while for Roaring Fork, CR-Cameo, and Dolores sites, the steepest decreases in FN concentrations were observed from 1980 to 2000. For the Dolores site, sharp decreases in FN loads, FN concentrations, and annual mean streamflow were observed following the implementation of the Paradox Valley Unit (black dot in Figure 4b), a salinity-control project that prevents groundwater brine from flowing

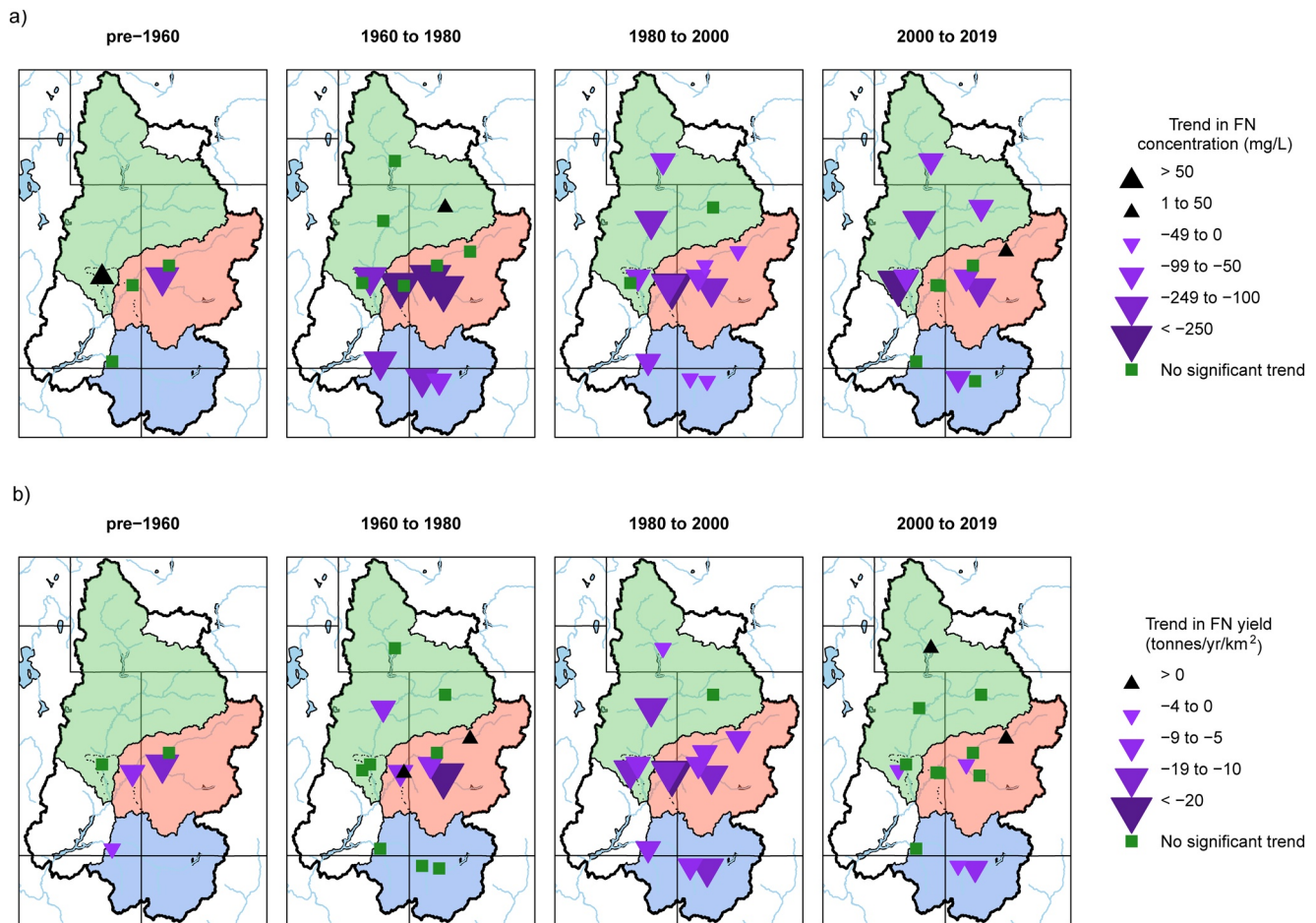


Figure 5. Map of 20-year-sub-period trends in (a) FN concentration and (b) FN yield (load normalized to drainage area) for 14 sites in the UCRB. Green shaded area = Green River sub-basin; red shaded area = Colorado River headwaters sub-basin; blue shaded area = San Juan River sub-basin. FN, flow-normalized; UCRB, Upper Colorado River Basin.

into the river. Increases in annual mean streamflow were observed for the Uncompaghre site (Figure 4b, Table S3). Annual mean streamflow did not change significantly at the four remaining sites.

For the three sites in the San Juan River sub-basin, decreasing trends in FN concentration and load tended to follow similar patterns through time (Figure 4c). Sites shared dissolved-solids loads and streamflows, as water at the Animas site flows into the SJ-Shiprock site, and the SJ-Shiprock site flows into the most-downstream site at SJ-Bluff. Similar to the other sub-basins, the steepest decreases in FN load occurred from 1980 to 2000 (Figures 4c and 5), again highlighting these decades as a period of drastic change in dissolved-solids transport. The steepest decreases in FN concentration occurred from 1960 to 1980 at all sites, similar to patterns observed at the Gunnison and CR-Cisco sites in the Colorado River headwaters sub-basin during that time (Figure 4b). Likely decreases in annual mean streamflow occurred for the SJ-Bluff and SJ-Shiprock sites (Figure 4c; Table S3) but were not significant for the Animas site.

Trend patterns from these three sub-basins reveal interesting similarities and differences that provide clues about potential drivers of change. For example, decreasing dissolved-solids trends occurred prior to 1980 in both the Colorado River headwaters and San Juan River sub-basins, but the Green River sub-basin did not show widespread decreases until decades after 1980 (Figure 5). The most-downstream main-stem sites at CR-Cisco and the SJ-Bluff integrate changes that occurred in the Colorado River headwaters and San Juan River sub-basins, respectively, and showed similar patterns of decrease in FN load for every sub-period except 1960 to 1980 (Table 2), suggesting similar watershed factors may have affected these two sub-basins over time. Both sites originate in the Colorado Rocky Mountains and potentially experienced related

Table 2
Trends in FN Dissolved-Solids Loads for Three Most-Downstream Main-Stem Sites in the UCRB for Four Sub-Periods of Interest

Site ID	Total change in FN load over the period of record, 1930–2019, in tonnes/yr	Change in FN load for four sub-periods of interest, in tonnes/yr (percent of 1930–2019 change)			
		1930–1960	1960–1980	1980–2000	2000–2019
CR-Cisco (Colorado River headwaters sub-basin)	–1,584,000	–369,000 (23%)	–338,000 (21%)	–856,000 (54%)	–21,000 (1%)
GR-Green River UT (Green River sub-basin)	–681,000	+167,000 (–25%)	+52,000 (–8%)	–787,000 (116%)	–113,000 (17%)
SJ-Bluff (San Juan River sub-basin)	–520,000	–149,000 (29%)	+14,000 (–3%)	–372,000 (72%)	–13,000 (3%)

Note. The percent of the total change occurring in each sub-period is given in parentheses. Abbreviations: FN, flow-normalized; UCRB, Upper Colorado River Basin.

changes in land cover, land use, or climatic factors that would have led to similar patterns in observed water quality. About a quarter of the total decrease in their dissolved-solids loads occurred prior to 1960 (Table 2), a time not yet affected by the construction of large dams or the salinity-control program. For the CR-Cisco site, nearly 45% of the total decrease in dissolved-solids load occurred prior to 1980 and the start of salinity-control efforts, demonstrating watershed factors independent of salinity control substantially affected salinity loading. In contrast, dissolved solids changed at few sites prior to 1980 in the Green River sub-basin (Figure 5). In fact, at the GR-Green River UT site, which integrates most changes from the Green River sub-basin, load increased (+32%) prior to 1980, indicating distinct drivers of dissolved-solids change occurred in this sub-basin pre-1980. This pattern disappears during 1980–2000, when widespread decreases in dissolved solids occurred in all three sub-basins and at all three main-stem sites (Figure 5 and Table 2), suggesting watershed processes acted on a basin-wide scale to reduce salinity during these decades. After 2000, patterns of salinity change are again distinct between sub-basins, where steep decreases in FN concentration continued in the Green River sub-basin, but for the Colorado River headwaters and San Juan River sub-basins the magnitude and extent of decreases in FN concentration were reduced (Figure 5). Flat or slower rates of decline in FN load were observed in all three sub-basins and at the three most-downstream main-stem sites after 2000 (Figure 5 and Table 2), indicating a shift in watershed process occurred during recent decades to halt the downward trajectory of dissolved solids that occurred from 1980 to 2000.

3.4. Dissolved-Solids Trends in the Context of UCRB Change

The three most-downstream main-stem sites in the study area—the CR-Cisco, the GR-Green River UT, and the SJ-Bluff—drain 80% of the UCRB and provide a good approximation for how dissolved-solids mass has changed in the basin over time. From 1930 to 2019, the cumulative change from these three sites amounted to a 91.2 million tonnes decrease in dissolved solids over the past 90 years. The Salinity Control Forum estimates that from 1980 to 2019, salinity mitigation efforts had reduced dissolved-solids loads by 23.8 million tonnes in the UCRB, less than half of the observed change that occurred at these sites over the same period (52.9 million tonnes). This finding, along with substantial decreases in dissolved solids preceding the implementation of salinity-control efforts, indicates there are additional land-cover, land-use, or climatic processes in the UCRB that significantly affect dissolved-solids transport in the basin (Bauch & Spahr, 1998; Butler, 1996; Leib & Bauch, 2007; Rumsey et al., 2017). Factors thought to affect salinity in the UCRB include erosion, channel evolution, hydrologic variation, shifts in land cover, changes in land use, and the construction of large reservoirs.

Decreases in dissolved solids were observed as early as the 1940s (Figures 4 and 5) and cannot be explained by influences from large reservoirs or salinity-control efforts. Instead, one theory is that century-scale arroyo evolutionary processes, along with climatic fluctuations, may have affected salinity and sediment loading. During the late 1800s and early 1900s, widespread arroyo incision in the Colorado Plateau region of the UCRB delivered massive quantities of sediment and salt to the Colorado River (Gellis et al., 1991). This was

followed in the early 1940s by widespread sediment aggradation and floodplain formation as channels stabilized throughout the basin (Hereford, 1987), likely influenced by low peak flows and below-normal average precipitation that occurred in the 1940s and 1950s (Gellis et al., 1991; Hereford, 1984). Such widespread sediment storage in the basin was coincident with decreases in suspended sediment observed at the CR-Cisco, the GR-Green River UT, and the SJ-Bluff sites (Gellis et al., 1991), as well as decreased salt loading observed throughout the UCRB (Kircher et al., 1984; Moody & Mueller, 1984; Thomas et al., 1963). Strong correlation between sediment and dissolved-solids concentrations has been observed in surface runoff in certain areas of the UCRB, demonstrating that soil erosion, sediment yield, and salinity transport processes can be highly related (Cadaret et al., 2016). Prior to large dam construction, Gellis et al. (1991) observed that salinity and sediment loading were highly correlated, concluding that decreased salt loads in the Colorado River prior to the 1960s were, at least partially, related to decreasing sediment loads resulting from reduced sediment production and increased sediment storage as the system recovered from widespread arroyo cutting. Conceptually, we theorize that severe erosion during the turn of the 20th century increased contact between soil particles and water, enhancing the release of dissolved solids from the surfaces of soil particles that were not previously in contact with moving water. When the period of erosion ended in the 1940s, the amount of new sediment coming in contact with water was reduced, causing decreased dissolved-solids loading to waterways.

In addition to hydrologic and geomorphic forces acting in the basin, land-use and land-cover change may have also affected stream water quality and arroyo evolution during the late 19th and early 20th centuries. From the mid-19th century to 1934, lands in the western US were subjected to overgrazing and unrestrained use by livestock without consideration of grazing capacity or adverse consequences to plant cover. As a result, conditions of rangelands deteriorated until public lands began to be regulated after the passage of the Taylor Grazing Act in 1934 (Cole et al., 1997; Hadley et al., 1977). In response to grazing regulation, there was a significant reduction in the number of grazing animals along with reported improvement in forage condition and soil erosion condition (Hadley et al., 1977), which coincided with reductions in sediment yield that were observed in the UCRB between 1926 and 1962 (Hadley, 1974). Ungrazed basins produced 45%–50% less sediment yield than grazed basins (Hadley et al., 1977; Lusby, 1970), and land-treatment efforts to reduce erosion and restore shrublands to grasslands resulted in dramatic decreases in sediment yield (Hadley et al., 1977; Lusby, 1970). Additionally, biotic soil crusts, which decrease sediment transport by reducing the amount of exposed bare ground (Belnap et al., 2013) and are adversely affected by livestock grazing (Belnap et al., 2009; Schwinning et al., 2008), may have recovered after grazing declined. Since salinity and sediment transport are less in areas with greater vegetation and biotic soil crusts (Belnap et al., 2009; Cadaret et al., 2016), their recovery in response to improved grazing practices may have contributed to declines in salinity loading. In summary, the reduction in sediment and salinity yield after 1940 coincides with a period when grazing regulations began on public lands and when land-use treatments to improve forage and soil erosion condition were initiated (Hadley et al., 1977), suggesting that reductions in stream salinity may be partly due to these management changes in addition to observed geomorphic evolution that occurred in the UCRB during this time.

Following the 1940s and 1950s, the sub-period from 1960 to 1980 had many of the steepest decreases in FN concentrations, particularly for sites in the Colorado River headwaters and San Juan River sub-basins (Figures 4 and 5). These time periods precede the implementation of large-scale salinity-control activities in the basin, but correspond to a period of widespread and substantial dam construction in the UCRB (see changes in dam storage over time in Figure S8), likely resulting in the dramatic shifts in observed streamflow regimes (Figure S5), where increasing low flows and decreasing high flows reflect water management practices that occur with dam operation. During the 1960s, water storage capacity upstream of Lake Powell increased by over 6.5 million acre-feet, mainly due to the construction of several large dams: Flaming Gorge Dam on the Green River, Blue Mesa Dam on the Gunnison River, and Navajo Dam on the San Juan River (Lieberman et al., 1989). During initial reservoir filling, increased salinity was observed at some sites as bank materials were inundated and mineral salts dissolved (e.g., GR-Green River UT site; Lieberman et al., 1989; Vaill & Butler, 1999); however, over longer time scales, decreases in dissolved-solids concentrations downstream of reservoirs (Moody & Mueller, 1984; Vaill & Butler, 1999) indicated reservoir processes may have removed dissolved solids from streams. In a recent investigation of Lake Powell, Deemer et al. (2020) estimated that roughly 10% of the inflow of dissolved solids was retained in the reservoir. Dissolved-solids retention in

Lake Powell, along with substantial decreases in FN dissolved-solids concentrations observed during the 1960s and 1970s (Figures 4 and 5), suggest reservoirs may be an important sink for dissolved solids throughout the UCRB.

The most widespread decreases in FN dissolved-solids loads and concentrations occurred between 1980 and 2000 (Figure 5); all but one site had significant ($p \leq 0.10$) declines in FN loads and all but two sites had significant declines in FN concentrations. The greatest magnitude decreases in FN dissolved-solids loads also occurred during this time, when all but three sites experienced their highest rates of decline. Such broad and steep patterns of declining dissolved solids from 1980 to 2000 indicate that regional-scale watershed or streamflow processes were acting across the basin to reduce dissolved-solids transport in the UCRB. Observed trends coincide with the start of the salinity-control program in the UCRB, which began widespread implementation of improved irrigation infrastructure, canal lining, and point-source treatments of saline groundwater in the 1980s and 1990s (Anning et al., 2010). Many of these mitigation strategies aim to reduce the amount of water transported through the subsurface, where water picks up soluble minerals as it travels through soil and rocks before discharging to streams (Anning et al., 2010; Mueller & Osen, 1988). As evidence of the potential effectiveness of these strategies, dissolved-solids loads in the groundwater-discharged fraction of UCRB streamflow decreased sharply during the 1990s (Rumsey et al., 2017). This finding, along with steep decreases in stream dissolved solids (Figures 4 and 5), supports the conclusion that salinity mitigation efforts reduced dissolved-solids transport in the basin during these decades (Anning et al., 2010; Butler, 1996; Schaffrath, 2012; Thiros, 2017).

However, there was a consistent slowing or reversing of downward trends after 2000 even though salinity-control efforts continued (Figures 4 and 5), suggesting another shift in dissolved-solids sources, transport, or loading in the basin. While FN load and concentration estimates are provisional for the most recent decade (post-2010; Figure 4), the changes in trend slopes in the early 2000s are considered to be reliable representations of salinity change. After 2000, only 4 out of 14 sites had significantly decreasing FN loads (Figure 5) and the magnitudes of decrease were, on average, 80% less than those observed from 1980 to 2000. Decreasing trends in FN concentration were more common after 2000, with 8 out of 14 sites showing decreases (Figure 5); the average magnitude of change was 8% less than from 1980 to 2000. Notably, significant decreases in FN concentration continued past 2000 at all five sites in the Green River sub-basin (Figure 5). Slower rates of dissolved-solids load and concentration reduction in recent decades may result from salinity-control efforts achieving easy initial success in the 1980s and 1990s, but having difficulty attaining additional progress. Alternatively, it could indicate that salinity-control strategies continued to work well past 2000, but that other watershed process, such as climate, vegetation changes, or other unidentified processes counteracted the effectiveness of salinity-control efforts in recent decades.

Land disturbances, degradation of biotic soil crusts, and reduced plant cover may have contributed to greater erosion and salinity transport after 2000 (Belnap et al., 2009; Cadaret et al., 2016; Gellis et al., 1991; Tillman & Anning, 2014). Substantial land disturbance from oil and gas development has occurred in the UCRB since 2000, and although it was not shown to affect dissolved solids in streams, these activities may increase erosion and affect the quality of soil stability and vegetation cover (Buto et al., 2010). Degraded and reduced extents of biotic soil crusts from land disturbance and climatic fluctuations can also weaken soil stability, reduce plant cover, and lead to increases in barren land that encourage water infiltration and soil erosion potential (Belnap, 2006; Ferrenberg et al., 2015), two mechanisms that increase salinity loading to surface waters in some areas of the UCRB.

Another reason for flattening trends in the Colorado River headwaters and San Juan River sub-basins may be the onset of a prolonged drought that occurred from 2000 to 2014, when decreases in streamflow occurred as a result of increased temperatures and decreased precipitation in the eastern sub-basins of the UCRB (Xiao et al., 2018). In the Green River sub-basin, precipitation and runoff were less affected by the post-2000 drought than the eastern sub-basins (Xiao et al., 2018); notably, there was no flattening of dissolved-solids concentration trends there after 2000. Salinity concentrations typically increase in streams during drought as a result of evapoconcentration and, perhaps more importantly for the UCRB, less dilution of groundwater-discharge containing high amounts of dissolved solids (Mosley, 2015). In the UCRB, the majority of dissolved solids (>80%) are estimated to be delivered to streams via groundwater discharge (Rumsey et al., 2017), and since the fraction of groundwater discharge to streams increases during years

with low runoff efficiency and drought (Rumsey et al., 2020), it is possible that groundwater discharge dominated total streamflow during the post-2000 drought and led to greater dissolved-solids concentrations during that time. These lower rates of decrease in dissolved-solids concentrations may also explain the flattening trends in FN dissolved-solids loads post-2000.

Given the potential for climatic fluctuations to alter salinity loading in the basin, it is important to consider that the climatic conditions experienced in recent decades are part of an overall warming trend in the region that is likely to continue lowering streamflows and altering hydrological processes into the future (Milly & Dunne, 2020; Williams et al., 2020). Williams et al. (2020) found that anthropogenic warming substantially amplified the 2000 to 2018 drought in the southwestern US, making it one of the most severe megadroughts in the last 1200 years. Additionally, the variability in climate that is predicted to increase with anthropogenic warming could affect salinity transport. For example, the flattening salinity trends in the early 2000s occurred during a drastic wet-to-dry transition period in the southwestern US, where the wettest period in 1,200 years (1980–1998) was followed by the 2000 to 2018 megadrought (Williams et al., 2020). These shifts in climate affect runoff and infiltration processes, altering the pathways by which salinity travels to streams. A thorough understanding of how changing climatic conditions affect salinity transport processes will support resource decisions in the successful application of future salinity mitigation strategies.

Multiple drivers of changing stream salinity are active in the UCRB. Although certain processes may have dominated salinity loading to streams during various periods of the last 50–90 years, it is likely that all of the abovementioned processes, along with others not addressed here, are continually affecting and interacting to change salinity loads and concentrations in UCRB streams. By putting trends in the context of watershed changes that occurred in the UCRB, this study highlights subsequent investigations needed to tease apart specific drivers of salinity change, including: quantifying dissolved-solids retention in reservoirs, understanding the effects of changing climate and changing streamflow regimes on salinity transport, evaluating changes in C-Q relationships on a seasonal basis, and quantifying the effects of watershed and land management activities, such as irrigation and grazing practices, on dissolved-solids loading.

4. Conclusions

At present, annual dissolved-solids loads and concentrations in the UCRB are substantially less than those observed 50–90 years ago. Long-term water quality and streamflow records from 14 stream sites across the UCRB show that all but two sites experienced significant and considerable decreases in dissolved-solids loads and concentrations from 1929 to 2019, where dissolved solids decreased by as much as 50% at some locations. Regional trends indicate similar watershed processes may have led to salinity changes in the Colorado River headwaters and San Juan River sub-basins, whereas trends in the Green River sub-basin were distinct and possibly influenced by different drivers of salinity change. Overall, we estimated that flow-normalized trends were largely caused by changes in watershed processes and less affected by changes in the streamflow regime.

Decreases in dissolved solids were observed as early as the 1940s, indicating there are watershed factors independent of reservoirs or salinity-control efforts that influence dissolved-solids transport in the UCRB. Possible mechanisms affecting salinity during this time relate to reductions in soil erosion and sediment loading driven by natural channel evolutionary processes and improved land management practices. Widespread and unparalleled rates of declining dissolved solids were observed from 1980 to 2000 and coincide with the launch of salinity-control efforts in the 1980s and 1990s. Notably, even with continued investment in salinity-control projects, the pace and extent of decreases in dissolved-solids concentrations and loads began to decline in the early 2000s around the time of a mega-drought from 2000 to 2014.

Because streams integrate and respond to changes on the landscape, the timing and extent of observed dissolved-solids trends suggest that broad, regional changes in stream morphology, land cover, land use, water regulation, climate, and/or other unidentified processes affected dissolved-solids transport in the UCRB during the 20th century. By analyzing long-term trends in dissolved solids across the UCRB, this work enhances our understanding of the evolving nature of salinity and lays the foundation for identifying specific drivers of change so that water managers in the basin can anticipate future changes in salinity, develop

more efficient salinity-control practices, and capitalize on natural processes that attenuate or reduce dissolved-solids transport.

Data Availability Statement

All data are publicly available from the USGS National Water Information System (<http://waterdata.usgs.gov/nwis>) and are cited in the references.

Acknowledgments

This project was funded by the Colorado River Basin Salinity Control Forum, the Bureau of Reclamation, and the Bureau of Land Management. This work is possible because of the availability of long-term discharge and water-quality data from the US Geological Survey. The authors would like to express appreciation and gratitude for the technicians and hydrologists, past and present, who collected these data. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the US Government.

References

- Anning, D. W., Bauch, N. J., Gerner, S. J., Flynn, M. E., Hamlin, S. N., Moore, S. J., et al. (2010). Dissolved solids in basin-fill aquifers and streams in the Southwestern United States (U.S. Geological Survey Scientific Investigations Report 2006-5315, 1.1, p. 168).
- Bauch, N. J., & Spahr, N. E. (1998). Salinity trends in surface waters of the Upper Colorado River Basin, Colorado. *Journal of Environmental Quality*, 27(3), 640–655. <https://doi.org/10.2134/jeq1998.00472425002700030023x>
- Belnap, J. (2006). The potential roles of biological soil crusts in dryland hydrologic cycles. *Hydrological Processes*, 20, 3159–3178. <https://doi.org/10.1002/hyp.6325>
- Belnap, J., Reynolds, R. L., Reheis, M. C., Phillips, S. L., Urban, F. E., & Goldstein, H. L. (2009). Sediment losses and gains across a gradient of livestock grazing and plan invasion in a cool, semi-arid grassland, Colorado Plateau, USA. *Aeolian Research*, 1(1–2), 27–43. <https://doi.org/10.1016/j.aeolia.2009.03.001>
- Belnap, J., Wilcox, B. P., Van Scoyoc, M. W., & Phillips, S. L. (2013). Successional stage of biological soil crusts: an accurate indicator of ecohydrological condition. *Ecohydrology*, 6(3), 474–482. <https://doi.org/10.1002/eco.1281>
- Butler, D. L. (1996). *Trend analysis of selected water-quality data associated with salinity-control projects in the Grand Valley, in the Lower Gunnison River Basin, and at Meeker Dome, Western Colorado* (U.S. Geological Survey Water-Resources Investigations Report 95-4274, p. 38).
- Buto, S. G., Kenney, T. A., & Gerner, S. J. (2010). *Land disturbance associated with oil and gas development and effects of development-related land disturbance on dissolved-solids loads in streams in the Upper Colorado River Basin, 1991, 2007, and 2025* (U.S. Geological Survey Scientific Investigations Report 2010-5064, p. 56). Retrieved from <http://pubs.usgs.gov/sir/2010/5064>
- Cadaret, E. M., Nouwakpo, S. K., McGwire, K. C., Weltz, M. A., & Blank, R. R. (2016). Experimental investigation of the effect of vegetation on soil, sediment erosion, and salt transport processes in the Upper Colorado River Basin Mancos Shale formation, Price, Utah, USA. *Catena*, 147, 650–662. <https://doi.org/10.1016/j.catena.2016.08.024>
- Chafin, D. T. (2003). *Effect of the Paradox Valley Unit on the dissolved-solids load of the Dolores River near Bedrock, Colorado, 1988-2001* (U.S. Geological Survey Water-Resources Investigation Report 02-4275, p. 6). <https://doi.org/10.3133/wri024275>
- Choquette, A. F., Hirsch, R. M., Murphy, J. C., Johnson, L. T., & Confesor, R. B., Jr. (2019). Tracking changes in nutrient delivery to western Lake Erie: Approaches to compensate for variability and trends in streamflow. *Journal of Great Lakes Research*, 45(1), 21–39. <https://doi.org/10.1016/j.jglr.2018.11.012>
- Christensen, N. S., Wood, A. W., Voisin, N., Lettenmaier, D. P., & Palmer, R. N. (2004). The effects of climate change on the hydrology and water resources of the Colorado River Basin. *Climatic Change*, 62, 337–363. <https://doi.org/10.1023/b:clim.0000013684.13621.1f>
- Cole, K. L., Henderson, N., & Shafer, D. S. (1997). Holocene vegetation and historic grazing impacts at Capitol Reef National Park reconstructed using packrat middens. *Great Basin Naturalist*, 57(4), 315–326. Retrieved from www.jstor.org/stable/41713018
- Colorado River Basin Salinity Control Forum (CRBSCF). (2014). *2014 Review: Water quality standards for salinity, Colorado River System* (p. 25). Retrieved from <http://coloradoriversalinity.org/docs/2014%20Final%20REVIEW%20-%20complete.pdf>
- Copeland, S. M., Bradford, J. B., Duniway, M. C., & Schuster, R. M. (2017). Potential impacts of overlapping land-use and climate in a sensitive dryland: A case study of the Colorado Plateau, USA. *Ecosphere*, 8(5), e01823. <https://doi.org/10.1002/ecs2.1823>
- Deemer, B. R., Stets, E. G., & Yackulic, C. B. (2020). Calcite precipitation in Lake Powell reduces alkalinity and total salt loading to the Lower Colorado River Basin. *Limnology & Oceanography*, 65(7), 1–17. <https://doi.org/10.1002/lno.11399>
- Dennison, P. E., Brewer, S. C., Arnold, J. D., & Moritz, M. A. (2014). Large wildfire trends in the western United States, 1984–2011. *Geophysical Research Letters*, 41(8), 2928–2933. <https://doi.org/10.1002/2014gl059576>
- Farr, T. G., & Kobrick, M. (2000). Shuttle radar topography mission produces a wealth of data. *Eos, Transactions, American Geophysical Union*, 81(48), 583–585. <https://doi.org/10.1029/eo081i048p00583>
- Ferrenberg, S., Reed, S. C., & Belnap, J. (2015). Climate change and physical disturbance cause similar community shifts in biological soil crusts. *Proceedings of the National Academy of Sciences of the United States of America*, 112(39), 12116–12121. <https://doi.org/10.1073/pnas.1509150112>
- Gellis, A., Hereford, R., Schumm, S. A., & Hayes, B. R. (1991). Channel evolution and hydrologic variations in the Colorado River Basin: Factors influencing sediment and salt loads. *Journal of Hydrology*, 124(3–4), 317–344. [https://doi.org/10.1016/0022-1694\(91\)90022-a](https://doi.org/10.1016/0022-1694(91)90022-a)
- Hadley, R. F. (1974). Sediment yield and land use in southwest United States. *International Association of Hydrological Sciences*, 113, 96–98.
- Hadley, R. F., Dasgupta, B., Reynolds, N., Sandford, S., & Worthington, E. (1977). Evaluation of land-use and land-treatment practices in semi-arid western United States. *Philosophical Transactions of the Royal Society of London, Series B, Biological Sciences*, 278(962), 543–554. Retrieved from <https://www.jstor.org/stable/2417807>
- Hamlet, A. F., & Lettenmaier, D. P. (2005). Production of temporally consistent gridded precipitation and temperature fields for the continental United States. *Journal of Hydrometeorology*, 6, 330–336. <https://doi.org/10.1175/jhm420.1>
- Hereford, R. (1984). Climate and ephemeral-stream processes: Twentieth-century geomorphology and alluvial stratigraphy of the Little Colorado River, Arizona. *The Geological Society of America Bulletin*, 95, 654–668. [https://doi.org/10.1130/0016-7606\(1984\)95<654:caeptg>2.0.co;2](https://doi.org/10.1130/0016-7606(1984)95<654:caeptg>2.0.co;2)
- Hereford, R. (1987). The short term: Fluvial processes since 1940. In W. L. Graf (Eds.), *Geomorphic systems of North America* (Geological Society of America Centennial Special, 2, pp. 276–288). Geological Society of America. <https://doi.org/10.1130/DNAG-CENT-v2>
- Hirsch, R. M., Archfield, S. A., & De Cicco, L. A. (2015). A bootstrap method for estimating uncertainty of water quality trends. *Environmental Modelling & Software*, 73, 148–166. <https://doi.org/10.1016/j.envsoft.2015.07.017>
- Hirsch, R. M., & De Cicco, L. A. (2015). User guide to Exploration and Graphics for River Trends (EGRET) and dataRetrieval: R packages for hydrologic data. In R. M. Hirsch & L. A. De Cicco (Eds.), *U.S. Geological Survey techniques and methods* (4, chap. A10, p. 93). <http://dx.doi.org/10.3133/tm4A10>

- Hirsch, R. M., De Cicco, L. A., & Murphy, J. (2018). *EGRETCi: Exploration and Graphics for RivEr Trends (EGRET) confidence intervals, version 2.0*. Retrieved from <https://CRAN.R-project.org/package=EGRETCi>
- Hirsch, R. M., De Cicco, L. A., Watkins, D., Carr, L., & Murphy, J. (2018). *EGRET: Exploration and Graphics for RivEr Trends, version 3.0*. Retrieved from <https://CRAN.R-project.org/package=EGRET>
- Hirsch, R. M., Moyer, D. L., & Archfield, S. A. (2010). Weighted Regressions on Time, Discharge, and Season (WRTDS), with an application to Chesapeake Bay River inputs 1. *Journal of the American Water Resources Association*, 46(5), 857–880. <https://doi.org/10.1111/j.1752-1688.2010.00482.x>
- Iorns, W. V., Hembree, C. H., & Oakland, G. L. (1965). *Water resources of the Upper Colorado River Basin* (Technical Report, U.S. Geological Survey Professional Paper 441, p. 370).
- James, T., Evans, A., Madly, E., & Kelly, C. (2014). *The economic importance of the Colorado River to the basin region* (p. 39). Arizona State University. Retrieved from <https://businessforwater.org/wp-content/uploads/2016/12/PTF-Final-121814.pdf>
- Kenney, T. A., Gerner, S. J., Buto, S. G., & Spangler, L. E. (2009). *Spatially referenced statistical assessment of dissolved-solids load sources and transport in streams of the Upper Colorado River Basin* (U.S. Geological Survey Scientific Investigations Report 2009-5007, p. 50). Retrieved from <http://pubs.usgs.gov/sir/2009/5007>
- Keum, J., & Kaluarachchi, J. J. (2015). Calibration and uncertainty analysis using the SPARROW model for dissolved-solids transport in the Upper Colorado River Basin. *Journal of the American Water Resources Association*, 51(5), 1192–1210. <https://doi.org/10.1111/1752-1688.12302>
- Kircher, J. E., Dinicola, R. S., & Middelburg, R. F. (1984). *Trend analysis of salt load and evaluation of the frequency of water-quality measurements for the Gunnison, the Colorado, and the Dolores Rivers in Colorado and Utah* (U.S. Geological Survey Water-Resources Investigations Report 84-4048, p. 69).
- Lee, C. J., Murphy, J. C., Crawford, C. G., & Deacon, J. R. (2017). *Methods for computing water-quality loads at sites in the U.S. Geological Survey National Water Quality Network* (U.S. Geological Survey Open-File Report 1120, p 20). <https://doi.org/10.3133/ofr20171120>
- Leib, K. J., & Bauch, N. J. (2007). *Salinity trends in the Upper Colorado River Basin upstream from the Grand Valley Salinity Control Unit, Colorado, 1986–2003* (U.S. Geological Survey Scientific Investigations Report 2007–5288, p. 21).
- Liebermann, T. D., Mueller, D. K., Kircher, J. E., & Choquette, A. F. (1989). *Characteristics and trends of streamflow and dissolved solids in the Upper Colorado River Basin, Arizona, Colorado, New Mexico, Utah, and Wyoming* (U.S. Geological Survey Water-Supply Paper 2358, p. 64).
- Lusby, G. C. (1970). *Hydrologic and biotic effects of grazing versus nongrazing near Grand Junction, Colorado* (U.S. Geological Survey Professional Paper 700-B, pp. B232–B236).
- Miller, M. P., Buto, S. G., Lambert, P. M., & Rumsey, C. A. (2017). *Enhanced and updated spatially referenced statistical assessment of dissolved-solids load sources and transport in streams of the Upper Colorado River Basin* (U.S. Geological Survey Scientific Investigations Report 2017-5009, p. 23). <https://doi.org/10.3133/sir20175009>
- Milly, P. C. D., & Dunne, K. A. (2020). Colorado River flow dwindles as warming-driven loss of reflective snow energizes evaporation. *Science*, 367(6483), 1252–1255. <https://doi.org/10.1126/science.aay9187>
- Moody, C. D., & Mueller, D. K. (1984). *Water quality of the Colorado River system: Historical trends in concentration, load, and mass fraction of inorganic solutes* (U.S. Bureau of Reclamation Report REC-ERC-84-9, p. 48).
- Mosley, L. M. (2015). Drought impacts on the water quality of freshwater systems; review and integration. *Earth-Science Reviews*, 140, 203–214. <https://doi.org/10.1016/j.earscirev.2014.11.010>
- Mueller, D. K., & Osen, L. L. (1988). *Estimation of natural dissolved-solids discharge in the Upper Colorado River Basin, Western United States* (U.S. Geological Survey Water-Resources Investigations Report 87–4069, p. 46).
- Murphy, J., & Sprague, L. (2019). Water-quality trends in US rivers: Exploring effects from streamflow trends and changes in watershed management. *Science of the Total Environment*, 656, 645–658. <https://doi.org/10.1016/j.scitotenv.2018.11.255>
- National Research Council. (2007). *Colorado River Basin water management: Evaluating and adjusting to hydroclimatic variability*. The National Academies Press. <http://doi.org/10.17226/11857>
- Oelsner, G. P., Sprague, L. A., Murphy, J. C., Zuellig, R. E., Johnson, H. M., Ryberg, K. R., et al. (2017). *Water-quality trends in the nation's rivers and streams, 1972–2012—Data preparation, statistical methods, and trend results* (U.S. Geological Survey Scientific Investigations Report 2017-5006, p. 136). <https://doi.org/10.3133/sir20175006>
- Radtko, D. B., Davis, J. V., & Wilde, F. D. (2005). *Specific electrical conductance, version 1.2* (Book 9, chap. A6.3). U.S. Geological Survey Techniques of Water-Resources Investigations. <https://doi.org/10.3133/twri09A6.3>
- Rumsey, C. A., Miller, M. P., Schwarz, G. E., Hirsch, R. M., & Susong, D. D. (2017). The role of baseflow in dissolved solids delivery to streams in the Upper Colorado River Basin. *Hydrological Processes*, 31(26), 4705–4718. <https://doi.org/10.1002/hyp.11390>
- Rumsey, C. A., Miller, M. P., & Sextstone, G. A. (2020). Relating hydroclimatic change to streamflow, baseflow, and hydrologic partitioning in the Upper Rio Grande Basin, 1980 to 2015. *Journal of Hydrology*, 584, 124715. <https://doi.org/10.1016/j.jhydrol.2020.124715>
- Schaffrath, K. R. (2012). *Surface-water salinity in the Gunnison River Basin, Colorado, water years 1989 through 2007* (U.S. Geological Survey Scientific Investigations Report 2012–5128, p. 47). <https://doi.org/10.3133/sir20125128>
- Schwinning, S., Belnap, J., Bowling, D. R., & Ehleringer, J. R. (2008). Sensitivity of the Colorado Plateau to change: Climate, ecosystems, and society. *Ecology and Society*, 13(2), 28. <https://doi.org/10.5751/es-02412-130228> Retrieved from <http://www.ecologyandsociety.org/vol13/iss2/art28/>
- Thiros, S. A. (2017). *Estimated dissolved-solids loads and trends for selected surface-water sites in and near the Uinta Basin, Utah, water years 1989–2013* (U.S. Geological Survey Scientific Investigations Report 2017–5004, p. 68). <http://doi.org/10.3133/sir20175004>
- Thomas, H. E., & others (1963). *Effects of drought in the Colorado River Basin—Drought in the southwest, 1942–1956* (U.S. Geological Survey Professional Paper 372-F, p. 50).
- Tillman, F. D., & Anning, D. W. (2014). A data reconnaissance on the effect of suspended-sediment concentrations on dissolved-solids concentrations in rivers and tributaries in the Upper Colorado River Basin. *Journal of Hydrology*, 519(A), 1020–1030. <https://doi.org/10.1016/j.jhydrol.2014.08.020>
- U.S. Bureau of Reclamation. (2012). *Colorado River Basin water supply and demand study, executive summary* (p. 28). U.S. Department of the Interior. Retrieved from https://www.usbr.gov/lc/region/programs/crbstudy/finalreport/Executive%20Summary/CRBS_Executive_Summary_FINAL.pdf
- U.S. Bureau of Reclamation. (2017). *Quality of water, Colorado River Basin, progress report No. 25* (p. 76). U.S. Department of the Interior. Retrieved from <https://www.usbr.gov/uc/progact/salinity/pdfs/ProgressReports/PR25final.pdf>

- U.S. Environmental Protection Agency. (1971). *The mineral quality problem in the Colorado River Basin, summary report* (p. 65). U.S. Environmental Protection Agency Regions VIII and IX. Retrieved from <https://azmemory.azlibrary.gov/digital/collection/p17220coll7/id/2492/>
- U.S. Geological Survey. (2021). *National water information system: U.S. Geological Survey web interface*. <http://dx.doi.org/10.5066/F7P55KJN>
- Vaill, J. E., & Butler, D. L. (1999). *Streamflow and dissolved-solids trends, through 1996 in the Colorado River Basin upstream from Lake Powell-Colorado, Utah, and Wyoming* (U.S. Geological Survey Water-Resources Investigation Report 99-4097, p. 47).
- Warner, J. W., Heimes, F. J., & Middelburg, R. F. (1985). *Ground-water contribution to the salinity of the Upper Colorado River Basin* (U.S. Geological Survey Water-Resources Investigations Report 84-4198, p. 92).
- Williams, A. P., Cook, E. R., Smerdon, J. E., Cook, B. L., Abatzoglou, J. T., Bolles, K., et al. (2020). Large contribution from anthropogenic warming to an emerging North American megadrought. *Science*, 368, 314–318. <https://doi.org/10.1126/science.aaz9600>
- Xiao, M., Udall, B., & Lettenmaier, D. P. (2018). On the causes of declining Colorado River streamflows. *Water Resources Research*, 54, 6739–6756. <https://doi.org/10.1029/2018wr023153>