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LONG-TERM TRENDS IN STREAMFLOW AND PRECIPITATION IN NORTHWEST CALIFORNIA AND SOUTHWEST OREGON, 1953-2012[1]

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ABSTRACT: Using nonparametric Mann-Kendall tests, we assessed long-term (1953-2012) trends in streamflow and precipitation in Northern California and Southern Oregon at 26 sites regulated by dams and 41 “unregulated” sites. Few (9%) sites had significant decreasing trends in annual precipitation, but September precipitation declined at 70% of sites. Site characteristics such as runoff type (groundwater, snow, or rain) and dam regulation influenced streamflow trends. Decreasing streamflow trends outnumbered increasing trends for most months except at regulated sites for May-September. Summer (July-September) streamflow declined at many sites, including 73% of unregulated sites in September. Applying a LOESS regression model of antecedent precipitation *vs*. average monthly streamflow, we evaluated the underlying streamflow trend caused by factors other than precipitation. Decreasing trends in precipitation-adjusted streamflow substantially outnumbered increasing trends for most months. As with streamflow, groundwater-dominated sites had a greater percent of declining trends in precipitation-adjusted streamflow than other runoff types. The most pristine surface-runoff-dominated watersheds within the study area showed no decreases in precipitation-adjusted streamflow during the summer months. These results suggest that streamflow decreases at other sites were likely due to more increased human withdrawals and vegetation changes than to climate factors other than precipitation quantity.

(KEY TERMS: surface water hydrology; runoff; rivers/streams; precipitation; climate variability/change; water supply; time series analysis.)

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INTRODUCTION

Water availability is a growing concern in the western United States (U.S.) for both humans and aquatic ecosystems, particularly during the hot and dry summer months (Moyle *et al.*, 2013; Georgakakos *et al.*, 2014). The climate is warming, shifting precipitation form from snow to rain, reducing snowpack, and causing earlier snowmelt (Regonda *et al.*, 2005; Stewart *et al.*, 2005; Barnett *et al.*, 2008). As a result, in snow-dominated watersheds, the timing of peak streamflow has shifted to earlier in the year (Regonda *et al.*, 2005; Stewart *et al.*, 2005; Hidalgo *et al.*, 2009; Fritze *et al.*, 2011). Summer streamflows are correlated with spring snowpack (Godsey *et al.*, 2014), and summer low flows are likely to decrease as the climate warms (Huntington and Niswonger, 2012; Berghuijs *et al.*, 2014; Vano *et al.*, 2015). Although the hydrologic effects of climate warming are

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expected to be more severe in basins that currently receive substantial snow, rain-dominated basins will also be affected. For example, increased temperatures will increase evapotranspiration of natural vegetation (Vano *et al.*, 2015) and increase water withdrawals for irrigating agricultural crops and landscaping (Katul *et al.*, 2012; Brown *et al.*, 2013) in both rain- and snow-dominated basins.

Aside from the effects of a changing climate, aquatic ecosystems are also already heavily affected by human activities (Katz *et al.*, 2013; Moyle *et al.*, 2013). Large quantities of water are withdrawn from surface and groundwaters for agricultural, industrial, and residential uses (Kenny *et al.*, 2009). Dams built for flood control and water supply have altered the timing and magnitude of peak and low streamflows (Magilligan and Nislow, 2005; Graf, 2006). Other human activities affecting landscape hydrology include urbanization (Booth and Jackson, 1997); wetland destruction through filling, draining (Fretwell *et al.*, 1996), and beaver trapping (Naiman *et al.*, 1988); hydraulic mining of floodplains (James, 1999); and alterations of forests through timber harvest (Bosch and Hewlett, 1982; Moore *et al.*, 2004; Creed *et al.*, 2014) and fire suppression. Most of these activities tend to decrease summer streamflows, with exceptions including dam releases to supplement summer flows (Magilligan and Nislow, 2005) and vegetation removal that can cause transient streamflow increases over multiyear periods (Jones and Post, 2004; Jones *et al.*, 2009). Streamflow has particular ecological and societal importance during summer because human water demands (primarily for irrigation) are greater and streamflow tends to be lower than other seasons.

Long-term trends in streamflow can be caused by basin-scale changes in vegetation and human water withdrawals as well as regional climate variables such as precipitation and air temperature. The Mann-Kendall test for monotonic trend (Helsel and Hirsch, 2002; Yue *et al.*, 2002a) is the statistical test most commonly used to detect long-term hydrologic trends in the western U.S. (Clark, 2010; Mayer and Naman, 2011; Chang *et al.*, 2012) and elsewhere (Pavelsky and Smith, 2006; Huo *et al.*, 2008; Jiang *et al.*, 2011; Patterson *et al.*, 2012; Jung *et al.*, 2013; Ahn and Merwade, 2014). Although many studies use this same statistical test to detect trends, the methods used to ascertain mechanisms causing streamflow changes (i.e., climate change or land use) vary widely. These methods include paired catchment studies (Jones and Post, 2004; Zhao *et al.*, 2010); regression models of precipitation-streamflow relationships (Huo *et al.*, 2008) including comparison of pre-impact and post-impact periods (Bosch and Hewlett, 1982; Zhao *et al.*, 2010); comparing the slopes of long-term precipitation and streamflow trends (Pavelsky and Smith, 2006); energy/water balances and relationships between rainfall, potential evapotranspiration, actual evapotranspiration, and streamflow (Zhang *et al.*, 2001; Zhao *et al.*, 2010; Jiang *et al.*, 2011; Wang *et al.*, 2013; Ahn and Merwade, 2014); and physically based hydrological simulation models (Arnold *et al.*, 1998; Jiang *et al.*, 2011; Zhang *et al.*, 2012; Waibel *et al.*, 2013; Wang *et al.*, 2013; Ahn and Merwade, 2014; Vano *et al.*, 2015).

This article evaluates long-term (1953-2012) trends in monthly and annual streamflow at locations in Northwest California and Southwest Oregon representing a wide range of natural and human-caused factors that affect those trends. This analysis focuses in particular on summer (July-September) streamflows, the depletion of which has contributed to population declines in coldwater anadromous fish species such as coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Oncorhynchus mykiss*) that must spend at least one summer in freshwater (Katz *et al.*, 2013). Our study area includes streams featuring diverse natural variability (e.g., geology, elevation, vegetation, and wildfire activity), as well as varying degrees of human alteration and impact. Previous analyses of streamflow trends within our study area (Van Kirk and Naman, 2008; Kim and Jain, 2010; Madej, 2011; Mayer and Naman, 2011; Sawaske and Freyberg, 2014) focused on climate change detection and therefore only assessed streams that are relatively unimpacted by humans. The only exceptions are Van Kirk and Naman (2008) and Kim and Jain (2010) who each included a single stream with highly impaired summer streamflows (Scott River). The detection of human and landscape alteration effects on streamflow is often obscured by climate variability such as precipitation quantity which is highly variable from year to year. To disaggregate long-term trends in streamflow from climate-driven trends in precipitation quantity, we used a simple statistical model (Locally Estimated Scatterplot Smoothing [LOESS] regression of monthly streamflow *vs.* Antecedent Precipitation Index [API]) to account for the fluctuations in streamflow caused by precipitation variability. Using this model, we assessed the underlying trends in streamflow caused by factors other than precipitation.

STUDY AREA

The study area spans all watersheds draining to the Pacific Ocean from the Mattole River in northwestern California to the Rogue River in southwestern Oregon, including the Eel and Klamath/Trinity River Basins (Figure 1). The study area was chosen to coincide with the range of the Southern Oregon/Northern California Coast (SONCC) Evolutionarily Significant Unit (ESU) of coho salmon (*Oncorhynchus kisutch*) in order to inform the National Marine Fisheries Services' development of an Endangered Species Act recovery plan for coho salmon (NMFS, 2014), which was the purpose of the initial version of this study (Asarian, 2015).

The study area comprises primarily mountainous terrain with some inland valleys and coastal plains. Elevations range from sea level to 4,300 m at Mount Shasta in California. The study area extends from the Coast Ranges along the Pacific Ocean east to the Klamath and Cascade mountains. Most of the study area has steep slopes, impermeable bedrock, and high precipitation (100-400 cm/year). The northeastern portion of the study area has permeable volcanic geology and a semiarid climate with annual precipitation ranging from 30 to 60 cm in the valleys to over 150 cm at the highest elevations. Conifer forests dominate the lower and higher elevations, and hardwood forests and grasslands are prevalent at the mid-elevations. The climate is Mediterranean, with cool wet winters and hot dry summers, except along the coast where summer temperatures are reduced due to marine influence. Most precipitation occurs in the winter and spring. Precipitation occurs as snow and rain at elevations from about 400 to 1,500 m, with snowpack generally accumulating above 1,500 m elevation from mid to late winter.

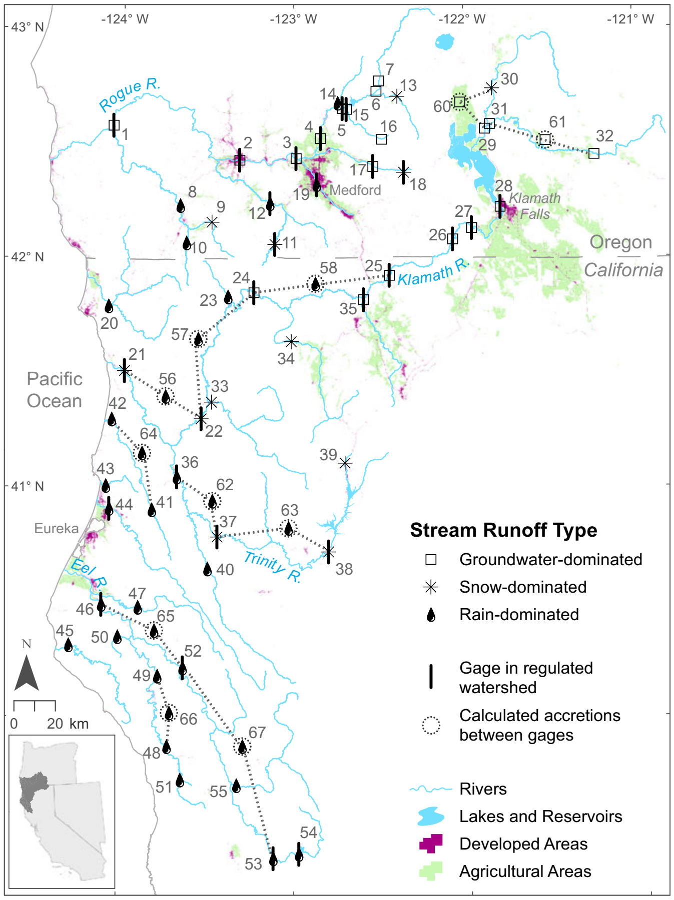

FIGURE 1. Map Showing the Location and Runoff Type for the 55 Streamflow Gages and the 12 Calculated Accretions Used in This Study. See Tables 1 and 2 for a key to site number labels. Developed and agricultural areas (Fry *et al.*, 2011) are shown as indicators of hydrologic alteration.

Human population density is relatively low, and the majority of land is federally owned. The largest population centers include Medford/Ashland (Rogue River Basin) and Klamath Falls (Klamath River Basin) in Oregon and Eureka/Arcata (Humboldt Bay) and Fortuna (Lower Eel River) in California. In those areas as well as the Shasta and Scott river valleys of California, agricultural irrigation is widely practiced (Table 1). Large dams (>50 Mm^3 of total storage) are present along the upper reaches of the Klamath, Trinity, Eel, Rogue, and Applegate rivers, which substantially regulate streamflows (Table 1, Table S1). Residential and small-scale agricultural water withdrawals on private lands throughout the study area, including for marijuana (*Cannabis sativa*) cultivation, are widely considered to have cumulatively significant impacts to coldwater anadromous fish populations (NMFS, 2014). Data to quantify these withdrawals are relatively scarce, especially since many of the diversions are unregistered, but the amount of land devoted to marijuana cultivation, and the accompanying water diversions, appears to have increased dramatically in recent years (Bauer *et al.*, 2015; Carah *et al.*, 2015). Timber harvest has occurred throughout much of the area although it has been reduced in recent decades on federal lands.

Major floods occurred in the study area in 1955 and 1964 (Lisle, 1982; Madej and Ozaki, 2009), and streamflows could be affected by the resulting aggradation. Channel aggradation increases sediment stored in streambeds, increasing infiltration of surface runoff into streambeds which would become subsurface intergravel flow and not be included in streamflow measurements. Data on changes in geomorphological conditions are not comprehensively summarized/accessible for the study area. Available data indicate that streambeds in many streams degraded back to stable levels within five years of the 1964 flood (Lisle, 1982), with exceptions including the lowest reaches of Redwood Creek where elevations did not peak until the 1990s and are still degrading (Madej and Ozaki, 2009), and Bull Creek which continued to degrade until at least 1982 (Stillwater Sciences, 1999).

TABLE 1. Site Information for Streamflow Gages.

Map No.	Gage Name (Abbreviated)	Gage ID	Basin Area (km^2)	Mean Elev. (m)	Max Elev. (m)	Annual Precip. (cm/yr)	BFI	CT	Run. Type	Flow Reg.	Res. Stor. (%)	ETAW (Mm^3/yr)	Streamflow Period of Record
1	Rogue R Agness	14372300	10,197	928	2,862	104	27	02/24	G	Y	7.8	198.6	1961-2012
2	Rogue R Grants Pass	14361500	6,362	1,006	2,862	97	35	03/09	G	Y	11.5	164.7	1940-2012
3	Rogue R at Raygold	14359000	5,310	1,073	2,862	100	41	03/18	G	Y	13.4	147.6	1906-2012
4	Rogue R Dodge Bridge	14339000	3,155	1,186	2,862	117	44	03/25	G	Y	16.6	8.1	1939-2012
5	Rogue R Near Mcleod	14337600	2,438	1,295	2,862	122	49	04/14	G	Y	20.8	1.4	1966-2012
6	Rogue R Blw Prospect	14330000	986	1,425	2,457	137	52	04/01	G	N	0.1	0.0	1914-1930, 1969-2004, 2006-2012
7[1]	Rogue R Abv Prospect	14328000	811	1,476	2,457	139	45	03/29	G	N	0.0	0.0	1909-1910, 1924-1998
8	Illinois R Kerby	14377100	985	880	2,142	191	3	02/08	R	N	0.0	7.6	1962-2012
9	Sucker Cr	14375100	217	1,210	2,142	151	11	03/06	S	N	0.0	0.0	1966-1991, 1993-2012
10[1]	EF Illinois R	14372500	140	1,116	1,922	230	5	02/14	R	N	0.0	0.1	1928-31, 1942-1996, 2001, 2007-2012
11	Applegate R Copper	14362000	580	1,294	2,248	118	20	03/13	S	Y	15.7	0.0	1939-2012
12	Applegate R Applegate	14366000	1,253	1,121	2,248	97	17	03/08	R	Y	8.9	4.6	1939-2012
13	SF Rogue R Nr Prospect	14332000	217	1,562	2,261	124	12	04/11	S	N	0.0	0.0	1925-1931, 1950-2012
14	Elk Cr Trail	14338000	336	950	1,758	118	1	02/13	R	N	0.0	0.1	1947-2005, 2007-2011
15	Big Butte Cr Near Mcleod	14337500	641	1,076	2,862	99	26	03/01	G	Y	2.4	0.6	1946-1957, 1968-2012
16	SF Big Butte Cr Ab Willow Cr	14335200	184	1,265	2,862	105	44	03/25	G	N	0.0	0.4	1936-2000[3], 2008-2012
17	NF L Butte Cr Nr LakeCr	14343000	100	1,395	2,862	117	39	05/13	G	Y	24.5	0.0	1912, 1923, 1927-1985[3], 2004-2012[3]
18	NF L Butte Cr At F L Nr LakeCr	14342500	43	1,632	2,862	130	16	06/25	S	Y	50.7	0.0	1917-1996, 2009-2012
19	Bear Cr At Medford	14357500	722	1,007	2,281	71	16	03/12	R	Y	12.0	33.1	1921-1981, 1985-2004, 2006-2012
20[1,2]	Smith R	11532500	1,578	772	1,944	259	6	02/06	R	N	0.0	0.0	1932-2012
21	Klamath R Klamath	11530500	40,912	1,317	4,303	94	17	03/03	S	Y	15.0	1003.2	1911-1926, 1951-1994, 1998-2012
22	Klamath R Orleans	11523000	31,496	1,395	4,303	75	24	03/09	S	Y	10.5	996.9	1928-2012
23	Indian C	11521500	310	1,128	2,149	193	10	03/04	R	N	0.0	0.0	1958-2008, 2010-2012
24	Klamath R Seiad Valley	11520500	27,503	1,431	4,303	62	33	03/12	G	Y	14.4	996.7	1913-1925, 1952-2012
25	Klamath R Iron Gate Dam	11516530	21,541	1,489	2,858	58	43	03/06	G	Y	18.9	766.2	1961-2012
26	Klamath R JCB Pwrplnt	11510700	18,500	1,502	2,858	57	36	03/05	G	Y	19.2	685.7	1960-2012
27	Klamath R Keno	11509500	18,081	1,503	2,858	56	31	03/08	G	Y	19.8	683.0	1905-1913, 1930-2012
28	Link R Klamath Falls	11507500	9,787	1,559	2,858	69	23	03/15	G	Y	17.7	167.1	1962-2012
29	Williamson R Blw Sprague R	11502500	7,820	1,578	2,751	63	50	03/30	G	N	0.3	52.6	1918-1922, 1924-2012
30	Williamson R Klamath Agency	11493500	3,475	1,563	2,751	72	1	03/20	S	N	0.0	13.3	1955-1995, 1999-2012
31	Sprague R Chiloquin	11501000	4,121	1,600	2,535	56	33	04/02	G	N	0.6	38.2	1922-2012
32	Sprague R Beatty	11497500	1,362	1,642	2,535	54	33	04/08	G	N	1.4	8.1	1954-2012
33[1,2]	Salmon R	11522500	1,943	1,298	2,664	148	9	03/18	S	N	0.0	0.0	1912-1915, 1928-2012
34	Scott R Ft Jones	11519500	1,714	1,319	2,587	77	6	03/18	S	N	0.1	81.5	1942-2012
35	Shasta R Yreka	11517500	2,047	1,227	4,303	66	12	02/09	G	Y	5.6	139.6	1934-1941, 1946-2012
36	Trinity R Hoopa	11530000	7,391	1,149	2,749	144	11	03/05	R	Y	30.4	1.7	1912-1913, 1917-1918, 1932-2012
37	Trinity R Burnt Ranch	11527000	3,727	1,250	2,749	135	19	03/18	S	Y	62.6	0.0	1932-1940, 1957-2012
38	Trinity R Lewiston	11525500	1,862	1,417	2,749	150	29	04/11	S	Y	115.0	0.0	1912-2012
39[1,2]	Trinity R Coffee Cr	11523200	383	1,630	2,749	130	9	04/07	S	N	0.0	0.0	1958-2012
40[1,2]	SF Trinity Hyampom	11528700	1,980	1,122	2,385	144	4	02/21	R	N	0.0	1.7	1966-2012
41[1]	Redwood Cr Blue Lake	11481500	175	893	1,619	199	2	02/16	R	N	0.0	0.0	1954-1958, 1973-1993, 1998-2012
42[1,2]	Redwood Cr Orick	11482500	718	558	1,619	186	2	02/09	R	N	0.0	0.1	1912-1913, 1954-2012
43[1,2]	Little R	11481200	105	333	1,027	165	4	02/05	R	N	0.0	0.0	1956-2012
44	Mad R Arcata	11481000	1257	800	1,834	168	2	02/10	R	Y	3.2	0.9	1911-1913, 1951-2012
45	Mattole R Petrolia	11469000	623	417	1,221	195	2	02/02	R	N	0.0	0.8	1912-1913, 1951-2012

(continued)

TABLE 1. Continued.

Map No.	Gage Name (Abbreviated)	Gage ID	Basin Area (km^2)	Mean Elev. (m)	Max Elev. (m)	Annual Precip. (cm/yr)	BFI	CT	Run. Type	Flow Reg.	Res. Stor. (%)	ETAW (Mm^3/yr)	Streamflow Period of Record
46	Eel R Scotia	11477000	8,062	786	2,306	159	1	02/10	R	Y	0.9	14.1	1911-1914, 1917-2012
47^{1,2}	Van Duzen R	11478500	572	923	1,788	166	1	02/05	R	N	0.0	0.0	1951-2012
48^{1}	SF Eel R At Leggett	11475800	642	626	1,289	192	3	02/06	R	N	0.0	0.5	1966-1994, 2000-2004, 2008-2012
49	SF Eel R Nr Miranda	11476500	1,390	526	1,289	185	2	02/05	R	N	0.1	0.5	1940-2012
50^{1,2}	Bull Cr	11476600	72	473	1,023	182	1	02/05	R	N	0.0	0.0	1961-2012
51^{1,2}	Elder Cr	11475560	17	848	1,277	247	3	02/10	R	N	0.0	0.0	1968-2012
52	Eel R Fort Seward	11475000	5,457	922	2,306	154	1	02/10	R	Y	1.3	12.7	1956-2012
53	Eel R Van Arsdale	11471500	904	1,070	2,140	138	3	02/18	R	Y	8.4	0.0	1911-1926, 1928-2012
54	Eel R Scott Dam	11470500	750	1,111	2,140	138	15	02/22	R	Y	10.1	0.0	1923-2012
55^{1,2}	MF Eel R	11473900	1,925	1,122	2,306	154	1	02/17	R	N	0.1	0.0	1966-2012

Notes: Basin elevation, area, and precipitation were computed for the catchment area contributing to site. Map numbers refer to Figure 1. BFI, modified base-flow index; CT, center of timing of streamflow (MM/DD). Key to runoff types: G, groundwater-dominated; S, snow-dominated; R, rain-dominated. Sites were classified as regulated if reservoir storage as % of watershed precipitation [Res. Stor. (%)] is >0.5 if mainstem reservoirs present or >2 if mainstem reservoirs absent. ETAW, evaporation of applied water on agricultural lands.

[1] Site listed as “reference” by GAGES-II (Falcone, 2011).

[2] Site included in USGS HydroClimatic Data Network (HCDN) 2009 (Lins, 2012).

[3] Two to four years missing within the period of record.

METHODS

Streamflow Data and Catchment Boundaries

Long-term U.S. Geological Survey (USGS) streamflow gages were identified through the GAGES-II project, which also provided GIS datasets of catchment boundaries (Falcone *et al.*, 2010; Falcone, 2011). Streamflow data for 55 gages from the USGS National Water Information System (http://nwis.waterdata.usgs.gov/nwis, *accessed* February 2013) were supplemented by additional data at a subset of those sites from the Oregon Water Resources Department (OWRD) (http://apps.wrd.state.or.us/apps/sw/hydro_near_real_time, *accessed* February 2013) (Table 1, Figure 1). The streams in this study span a wide range of human influence from relatively unimpacted to highly impacted due to extensive dams and water diversions (Table 1).

Estimated Accretions between Streamflow Gages

Some river basins have multiple gages allowing for the calculation of accretions between gages. These accretions were calculated as the difference in observed streamflow between the upstream and downstream gages less any flow from additional gaged tributaries between the two gages. Accretions were calculated on a daily basis and then smoothed with a seven-day average to reduce the frequency of negative values. The calculated accretion represents the net contributions of all ungaged tributaries, springs, and groundwater inputs, minus removals from any diversions. The calculated accretions inherently include the combined measurement error of all the component gages and are therefore less accurate than flows measured at individual gages. Despite the increased uncertainty, the 12 calculated accretions (Table 2) provide valuable data to supplement the network of gages within the study area (Table 1) by allowing evaluation of streamflow in unregulated tributaries which provide critically important rearing habitat for juvenile salmonid fish in river systems with regulated mainstem flows (i.e., Klamath, Trinity, and Eel rivers).

Classification of Sites by Runoff Type and Flow Regulation

We classified streamflow sites by two criteria: runoff type (groundwater-dominated, snow-dominated, and rain-dominated) and flow regulation by dams (regulated and unregulated) (Table 1).

TABLE 2. Site Information for Calculated Accretions between Streamflow Gages.

Map No.	Accretion Name	Formula	Basin Area (km^2)	Mean Elev. (m)	Max Elev. (m)	Annual Precip. (cm)	CT	Runoff Type	Flow Reg.	Res. Stor. (%)	ETAW (Mm^3/yr)
56	Klamath R Accretions: Klamath — Orleans — Trinity R	21-22-36	2,025	717	2,106	197	02/19	R	Y	0.0	4.6
57	Klamath R Accretions: Orleans — Seiad — Salmon R	22-24-33	2,050	1,007	2,232	188	02/26	R	Y	0.0	0.2
58	Klamath R Accretions: Seiad — Iron Gate — Shasta R — Scott R	24-25-35	2,201	1,142	2,521	86	03/16	R	Y	0.0	9.4
59	Williamson R: Williamson R — Sprague R	29-31	3,699	1,554	2,751	71	03/24	G	Y	0.0	14.5
60	Williamson R Accretions: Below Sprague — Sprague R — Klamath Agency	29-31-30	224	1,413	1,753	58	03/24	G	Y	0.0	1.1
61	Sprague R Accretions: Chiloquin — Beatty	31-32	2,759	1,579	2,469	57	03/22	G	Y	0.2	30.0
62	Trinity R Accretions: Hoopa — Burnt Ranch — SF Trinity	36-37-40	1,684	958	2,308	164	02/22	R	Y	0.0	0.0
63	Trinity R Accretions: Burnt Ranch — Lewiston	37-38	1,865	1,084	2,724	121	03/04	R	Y	0.1	0.0
64	Redwood Cr Accretions: Orick — Blue Lake	42-41	543	450	1,247	182	02/11	R	Y	0.0	0.1
65	Eel R Accretion: Scotia — Ft Seward — SF Miranda	46-52-49	1,215	470	1,710	155	02/09	R	Y	0.0	0.9
66	SF Eel R Accretions: Miranda — Leggett	49-48	748	440	1,245	179	02/06	R	Y	0.1	0.0
67	Eel R Accretions: Ft Seward — Van Ars — MF	52-53-55	2,628	725	1,882	159	02/06	R	Y	0.1	12.7

Notes: All notes to Table 1 also apply here. The numbers in the Formula column refer to the map numbers (Figure 1, Table 1) of the gages from which the accretion is calculated (downstream minus upstream minus any gaged tributaries). Modified base-flow index (BFI) is not shown because minimum flow seven-day average flow was not calculated due to high uncertainty of calculated accretions at such short time scales.

Geology and elevation affect hydrologic characteristics (Reidy Liermann *et al.*, 2012; Patil *et al.*, 2014) including response to changing climate and land cover/land use (Mayer and Naman, 2011; Waibel *et al.*, 2013). Adapting criteria from Mayer and Naman (2011), groundwater-dominated *vs*. surface-dominated sites were differentiated by modified base-flow index (BFI, long-term average of the ratio of the annual minimum seven-day average flow to the annual mean daily flow), with groundwater-dominated basins having BFI > 0.25. Surface-dominated sites were further differentiated into rain-dominated and snow-dominated, according to elevation and the center of timing of streamflow (CT, the date by which 50% of the runoff in a water year has occurred). Snow basins had mean elevation >1,200 m, and CT occurring during or after mid-March. Rain basins had mean elevation <1,200 m and CT in or before mid-March. Nearly all groundwater-dominated sites occurred at elevations >1,200 m so we did not differentiate these sites by elevation (snow *vs*. rain). Some professional judgment was applied for classification because dams and water diversions affect base-flow index and CT.

Flow regulation was assessed by comparing the combined volume of the water storage reservoirs in a watershed contributing to a streamflow site with total annual watershed precipitation (Table 1). Reservoir volumes were calculated using the NOAA Fisheries Dams 2005 GIS layer (Goslin, 2005) for California and the Oregon Dams 2010 GIS layer from the Oregon Department of Water Resources. Completion dates for the major dams within the study area range from 1910 to 1980 (Table S1). Total annual watershed precipitation was based on the 1981-2010 “normals” from the PRISM precipitation dataset (see Precipitation Data section below). Sites were classified as regulated if reservoir storage was >0.5% of watershed precipitation in watersheds with mainstem reservoirs or >2% in watersheds where no mainstem reservoirs exist.

Calculation of Streamflow Metrics

Key streamflow metrics were selected based on a review of previous analyses (Poff, 1996; Madej, 2011; Mayer and Naman, 2011; Chang *et al.*, 2012) and calculated for each streamflow site and year. These metrics include minimum 7-day average flow, minimum 30-day average flow, minimum 90-day average flow, average flow for each month, annual mean flow, and center of timing of streamflow. Minimum flow seven-day average flow was not calculated for accretions between gages due to the increased uncertainty at shorter time scales.

Estimation of Agricultural Irrigation Consumptive Water Use

We estimated annual consumptive water use by irrigated agriculture in the California portion of the study area using data from the California Department of Water Resources (CDWR). CDWR uses land use surveys and water use models to estimate annual evapotranspiration of applied water (ETAW) at subbasin to basin scales which do not necessarily correspond to streamflow gage catchments. Using ETAW data for 1998-2001 (CDWR Annual Land & Water Use Estimates, http://www.water.ca.gov/landwateruse/anlwuest.cfm, *accessed* November 2012) and 2002-2005 (Gholam Shakouri, CDWR, February 26, 2013, personal communication), we calculated the annual mean ETAW for each subbasin. We then evenly distributed the ETAW across the agricultural lands in the 2006 National Land Cover Database (Fry *et al.*, 2011) within each subbasin and then aggregated the ETAW to the streamflow gage catchments (Table 1).

We followed similar steps to estimate agricultural irrigation water demand in Oregon using county-level data from HDR Inc. (2008); however, the HDR Inc. demand estimates included adjustments for conveyance efficiency (constant 80%) and irrigation efficiency (varied by county/crop, range 50-90%), so are higher than ETAW. Therefore, we applied adjustments factors of 80% for conveyance (the same value used by HDR Inc.) and 70% for irrigation efficiency (the middle of the range presented by HDR Inc.) to back-calculate ETAW values that are comparable to the California data.

The ETAW estimates have a relatively high degree of uncertainty due to the assumptions required and inherent complexity; therefore, we present these estimates to inform interpretation of streamflow trends, but do not formally use them to classify streamflow sites or use them in quantitative analyses (i.e., comparisons with streamflow). A significant limitation of the ETAW estimates is that they only encompass traditional legal agricultural crops grown on prime agricultural land in relatively large fields. Small irrigated pastures, gardens, and marijuana cultivation sites are not included. Another limitation is that these estimates do not include specific diversions to areas outside catchment boundaries (i.e., large out-of-basin transfers from the Eel and Trinity rivers).

Agricultural irrigation is the human activity with the largest, but not the only, consumptive use of water in the study area. An early version of this study (Asarian, 2015) estimated domestic indoor/outdoor water use based on U.S. Census data and assumptions of *per-capita* water use, but these estimates are not included in this article due to their high uncertainty. Other uses including industrial, thermoelectric power (i.e., cooling for electronic power generation), livestock, aquaculture, and mining are also not included in this analysis. County-level estimates for 2005 for these uses are available from the USGS (Kenny *et al.*, 2009); however, there is no straightforward way to spatially downscale these estimates to subbasin or watershed scales.

Precipitation Data

Precipitation data were obtained from the PRISM Climate Group (http://www.prism.oregonstate.edu, *accessed* December 2012), which combines measured data from individual weather stations with an expert algorithm to produce a spatially continuous 4-km resolution precipitation grid for each month and year (Daly *et al.*, 2002, 2008). A monthly precipitation time series for the area contributing to each streamflow site was calculated using ArcGIS Python scripts to clip each grid to the study area, convert each grid cell to a point feature, spatially join the points to catchment boundary polygons, and then calculate the mean value within each catchment.

Calculation of Runoff Coefficient

For each streamflow site and year, the runoff coefficient was calculated as total annual streamflow divided by total annual precipitation. Median values are presented in Table S2.

Calculation of "Precipitation-Adjusted Streamflow" to Account for Precipitation Variability

Precipitation is the source of streamflow and is therefore directly correlated with the amount of streamflow. Large yearly fluctuations in precipitation may obscure underlying trends in streamflow caused by changes in other climate factors aside from precipitation quantity (e.g., air temperature, snow *vs.* rain, wind, humidity, and coastal fog), vegetation, or water withdrawals. When the variation in streamflow caused by precipitation is removed, the underlying trends in streamflow can be observed (Helsel and Hirsch, 2002).

To avoid a complex model selection process for each site to predict monthly streamflow based on precipitation from various time periods, a simpler approach was utilized based on the API. The API is computed for each timestep as a weighted sum of current and previous precipitation. Precipitation in the current period is assigned full weight, and each preceding period is assigned a progressively lower weight. The API is a proxy for soil moisture and has been used to predict both storm flow (Fedora and Beschta, 1989) and base flow (Reid and Lewis, 2011). Due to the availability of monthly precipitation data, in this article, the API is calculated on a monthly timestep rather than the conventional daily timestep. At sites dominated by surface runoff, API was calculated for each site at a monthly timestep by combining the precipitation in the given month with a weighted sum of precipitation in the preceding 11 months as follows:

$$\begin{aligned}\mathrm{API}_i = (P_i) + (P_{i-1})(k^1) + (P_{i-2})(k^2)\\ + (P_{i-3})(k^3) + \cdots + (P_{i-11})(k^{11})\end{aligned} \tag{1}$$

where API_i is the API for month i in units of cm, P_i is precipitation for month i in units of cm, and k is a dimensionless recession coefficient ranging from 0 to 1 which is specific to the gage and month. At groundwater-dominated sites, the API formula was identical except that 36-month precipitation was used to account for multiyear memory (Mayer and Naman, 2011) in those systems. A recession coefficient (k) was calibrated separately for each site and month by maximizing Spearman's rank correlation coefficient between monthly average streamflow and API. Details about calibration, recession coefficients, and correlation coefficients are provided in the Supporting Information.

A LOESS regression curve (Helsel and Hirsch, 2002) was then fit to the scatterplot of monthly streamflow *vs.* API, and the error residuals were calculated for each year as observed minus predicted (e.g., Figure 2). These residuals represent the variability in streamflow due to factors other than precipitation and are referred to as precipitation-adjusted streamflow (Helsel and Hirsch, 2002).

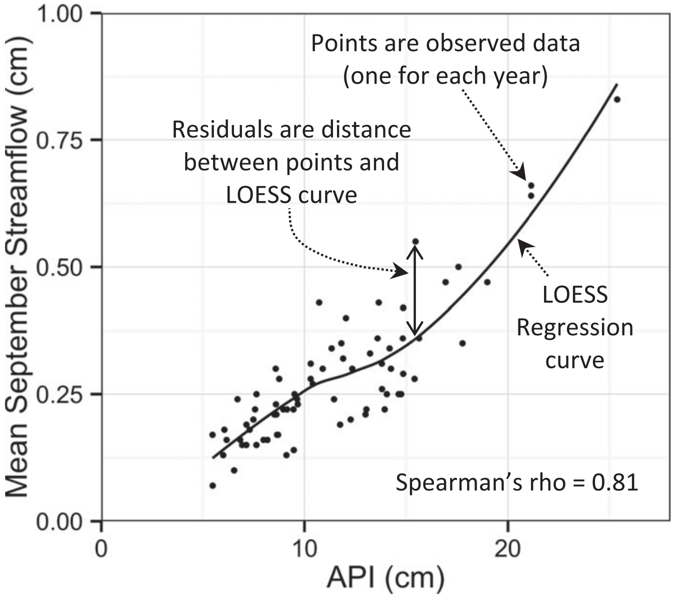

FIGURE 2. Relationship between Streamflow and Antecedent Precipitation Index (API) at an Example Site (South Fork Eel River at Miranda Gage #11476500) for the Month of September.

Long-Term Trends in Precipitation, Streamflow, and Precipitation-Adjusted Streamflow

Long-term trends in precipitation (monthly and annual), streamflow magnitude (monthly average; annual average; and minimum 7-day, 30-day, and 90-day average), streamflow timing (date of water year on which center timing of streamflow [CT] occurs and date of calendar year on which minimum 7-day, 30-day, and 90-day average streamflow occurs), runoff coefficient, and precipitation-adjusted streamflow were analyzed using the nonparametric Mann-Kendall test (Yue *et al.*, 2002a), which is commonly used for assessing hydrologic trends (see Introduction section above). The Mann-Kendall test assumes a lack of serial correlation (Helsel and Hirsch, 2002). Pre-whitening is sometimes used to remove the effect of serial correlation, but this can reduce trend detection power (Yue *et al.*, 2002b; Bayazit and Önöz, 2007; Sonali and Nagesh Kumar, 2013) and sometimes cause incorrect results (Sang *et al.*, 2014). Bayazit and Önöz (2007) found that pre-whitening is not necessary for large sample sizes (≥50) and trend high slopes (≥0.01). Given our relatively high sample sizes, we did not pre-whiten. We acknowledge that serial correlation could affect the statistical significance values we report.

A p-value of 0.10 was used as the statistical significance threshold for determining whether a trend existed for a given parameter and site, following the convention used in similar previous analyses (Clark, 2010; Madej, 2011; Chang *et al.*, 2012). Many figures also differentiate which results yielded p-values of

<0.05. Given the 0.10 threshold and the 67 hypothesis tests (one per site) performed on each parameter, the family-wise error rate (i.e., the chance of at least one Type I error [false detection of a nonexistent trend]) is very high across the entire study area. Spatial autocorrelation between sites also likely present and should be considered when making inferences about region-wide trends. However, our purpose was not to make formal statistical inferences about unmonitored sites within the study area, but rather to focus on the existence of trends in the gaged watersheds only, with a secondary purpose of understanding the factors that contribute to those trends (e.g., geology, elevation, precipitation quantity, other climate variables, regulation by dams, and other human influences). The results should thus be interpreted as being descriptive rather than inferential when considered in aggregate across the study area.

Tests were performed in R 2.15.2 (R Core Team, 2012) using the WQ package (Jassby and Cloern, 2012). To facilitate comparison of trends between sites, trend tests were run on the 60-year period 1953-2012, with some gaps allowed. Following guidance from Helsel and Hirsch (2002), sites that did not have at least 20% coverage (four years) in each third (1953-1972, 1973-1992, and 1993-2012) of the 60-year period were excluded. Trend slopes were calculated using the nonparametric Sen slope estimator method (Helsel and Hirsch, 2002).

A statistically significant trend in precipitation-adjusted streamflow indicates a shift over time in the relationship between streamflow and precipitation (e.g., that monthly streamflows in recent years are lower or higher than those in previous years with similar precipitation). When significant trends were present for streamflow and precipitation-adjusted streamflow, the Sen slope of the precipitation-adjusted streamflow trend was divided by the Sen slope of the streamflow trend to yield the percent of the streamflow trend not due to precipitation. In a few cases, the slope of the precipitation-adjusted streamflow trend was greater than the slope of the streamflow trend, resulting in values exceeding 100% which should be interpreted to mean that the streamflow decline was due entirely to factors other than precipitation.

RESULTS AND DISCUSSION

API Model to Calculate Precipitation-Adjusted Streamflow

Optimal recession coefficients followed expected patterns reflecting physical processes according to runoff type classification (i.e., highest at groundwater-dominated basins and lowest at surface-dominated rain basins) and month (i.e., higher in summer than winter and spring at surface-dominated rain basins; Figure S4). Although the streamflow *vs.* API model was originally designed for rain-dominated basins, it also performed well at snow- and groundwater-dominated basins. Spearman's correlation coefficients were highest at rain-dominated basins and lowest at groundwater-dominated sites, reflecting more complex hydrology in the latter category (Figure S5). As expected, Spearman's correlation coefficients were lower at regulated sites than unregulated sites (Figure S5).

Long-Term Trends in Precipitation

Of the 67 sites evaluated in this study, very few (9%) had significant decreases in annual precipitation (Figure 3a). However, all sites had at least one month with a precipitation trend (Figure S6). The most geographically widespread trend was a decrease in September precipitation, which occurred at 70% of sites (Figures 3d and 4), confirming results previously reported by Madej (2011) for the western portion of the study area. Other precipitation trends were more geographically limited and included: decreased August precipitation primarily in the Eel and Trinity Basins in the southeastern portion of the study area, increased April precipitation in the Upper Rogue Basins and the Upper Klamath Basin, increased May through July precipitation in parts of the Eel River Basin and nearby coastal areas (the absolute amount of precipitation in these months is still very low relative to the rest of the year), and decreased January precipitation in the Middle Klamath Basin as well as parts of the Eel River Basin and the upper Illinois River (Figure S6). September was the only month with a geographically widespread decreasing trend in API (data not shown), apparently from decreased September precipitation rather than prior months.

Long-Term Trends in Annual Streamflow

Annual streamflows declined at 24% of sites, primarily in groundwater-dominated sites in the Upper Klamath Basin (Figure 3b) where Mayer and Naman (2011) had previously documented declining streamflow, exceeding the 9% of sites that had declining annual precipitation (Figure 3a). Only one site, the regulated Trinity River at Lewiston, showed significant increases in annual flow due to reduced water diversions as part of a river restoration program (USFWS and HVT, 1999; Beechie *et al.*, 2014).

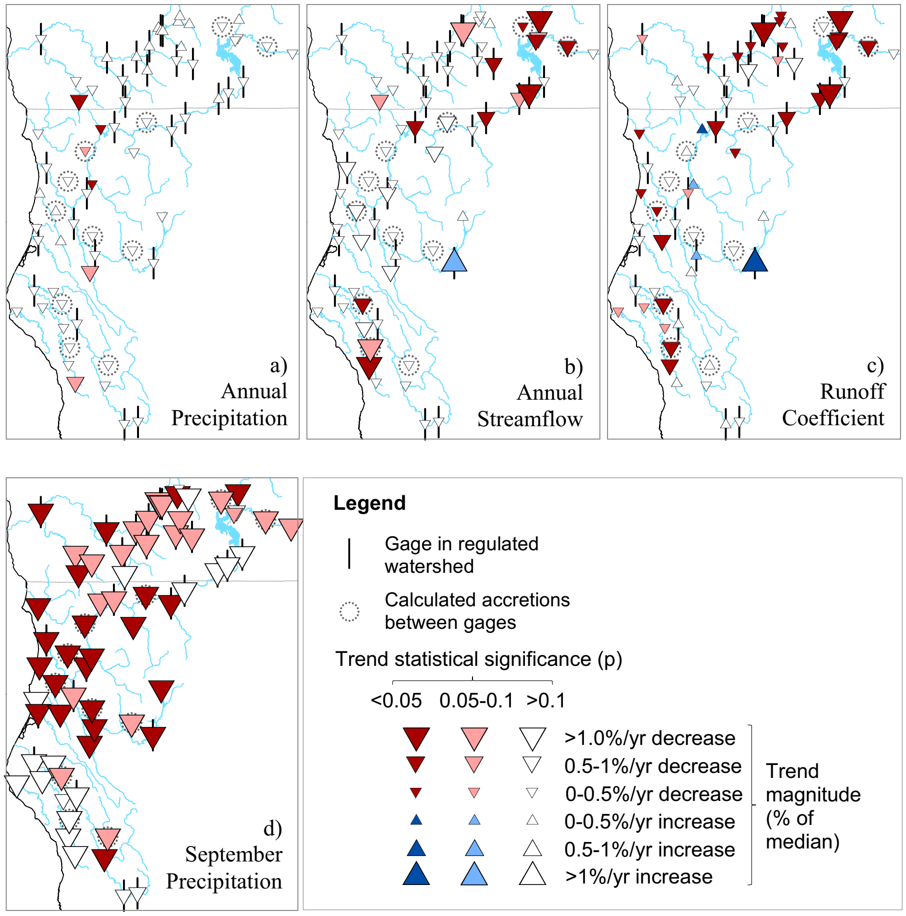

FIGURE 3. Map Showing Trends in (a) Annual Precipitation, (b) Annual Streamflow, (c) Runoff Coefficient, and (d) September Precipitation for Catchments Contributing to Streamflow Sites, 1953-2012.

This was also one of only four sites (6%) with an increasing runoff coefficient. In contrast, 46% of sites had declining runoff coefficients (Figure 3c). The cause of the declining trends in runoff coefficients is unclear. Potential explanations include some combination of increased vegetation/forest evapotranspiration (from climate change and/or change in forest stand structure/composition) and/or increased water diversions.

Long-Term Trends in Monthly Streamflow

Seasonal streamflow trends varied by month and appeared to be affected by hydrologic regulation as well as runoff type (i.e., geology and elevation; Figure 5). Overall, the percent of site months with significant flow decreases substantially outnumbered those with significant increases (Figures 4b, 5, and 6). At unregulated and regulated sites, declining flow trends vastly outnumbered increasing flow trends for October through April (Figures 5 and 6). For the remainder of year (May through September), regulated and unregulated sites showed opposing patterns with increasing trends outnumbering decreasing trends at regulated sites and decreasing trends outnumbering increasing trends at unregulated sites (Figures 5 and 6). At some regulated sites (Rogue and Applegate rivers), increased May through October flows resulted from dam construction partway through the 1953-2012 trend period, while in others (Eel and Trinity rivers), instream releases from reservoirs were increased to benefit coldwater anadromous fisheries in recent decades (USFWS and HVT, 1999; NMFS 2002). At regulated sites, the month with the largest percentage of declining flows was February (69%) (Figures 5 and 6). September flows declined at 73% of the unregulated sites, more than in any other month (Figures 5 and 6), likely due in part to decreased precipitation in that month (Figures 3d and 4a), although the relative magnitude of the declines were greater in November than September (Figure 7). Groundwater-dominated sites had more months with declining streamflow than other runoff types (Figure 5). No unregulated rain-dominated site had a significant increase in streamflow in any month

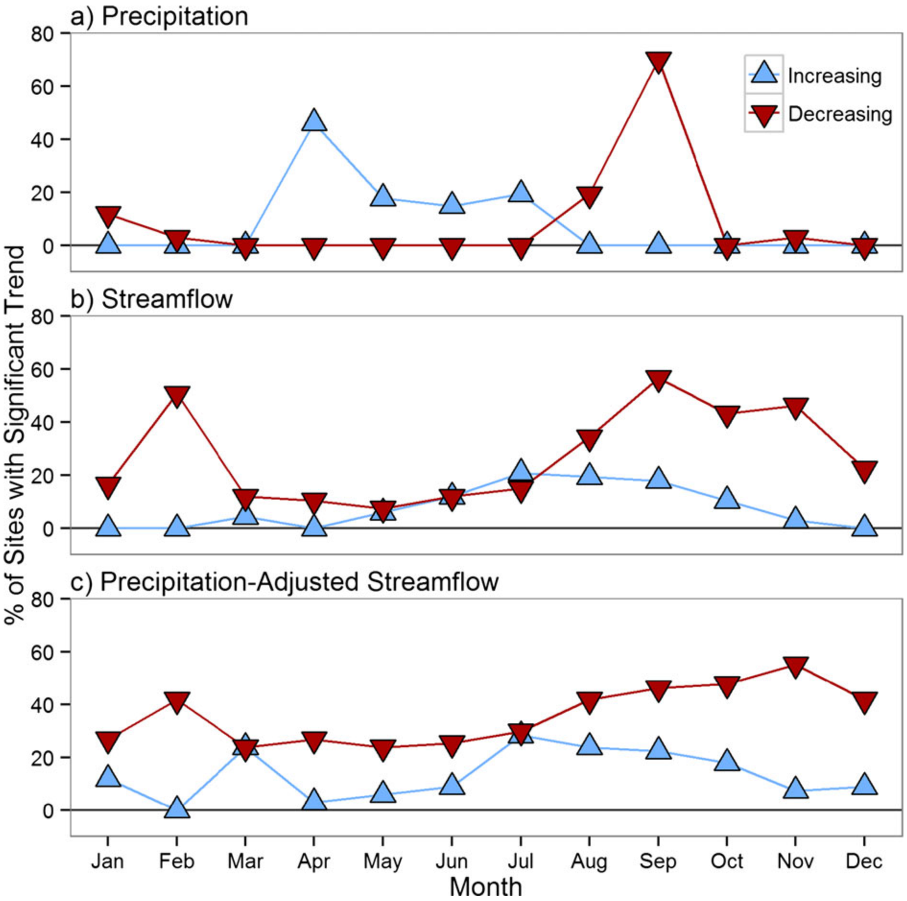

FIGURE 4. Percent of Streamflow Sites with Significant Increasing or Decreasing Trends in (a) Precipitation, (b) Streamflow, and (c) Precipitation-Adjusted Streamflow.

(Figure 5). The monthly patterns in the relative magnitude of increases/decreases (Figure 7) largely matched those of the percent of increasing/decreasing trends. The absolute magnitude of increases/decreases were greatest in November through April (Figure S8), the months when streamflows are higher. For the 14 gages analyzed both here and by Sawaske and Freyberg (2014), the presence/absence and direction of trends in streamflow during the summer months match closely.

Long-Term Trends in 7-day, 30-day, and 90-day Streamflow

Trends in the magnitude of minimum 7-day, 30-day, and 90-day average low flows were similar to each other and were highly affected by hydrologic regulation (Figure S7). Approximately 48-54% of unregulated sites showed significant declines, while only 2-4% of these sites showed increases. In contrast, 44-48% of regulated sites increased, while 7-15% decreased. Significant trends in the timing of the minimum 7-day, 30-day, and 90-day average flows were largely confined to regulated sites, with those flows occurring later in the calendar year at 48-56% of regulated sites, but only 4-10% of unregulated sites. For regulated sites where low flows occurred significantly later, the median delay normalized across the entire 60-year trend period was 41 days for the 7-day average low flow, 30 days for the 30-day average low flow, and 38 days for the 90-day average low flow (data not shown).

Long-Term Trends in Center of Timing of Streamflow

The center of timing of streamflow (CT, the date by which 50% of the runoff in a hydrologic year has occurred) occurred significantly later at 35% of unregulated sites and 74% of regulated sites, compared to only one site occurring earlier (Figure S7). This shift toward later runoff, which occurred at sites dominated by surface runoff (not groundwater), matches regional trends of later runoff in rain-dominated basins of the

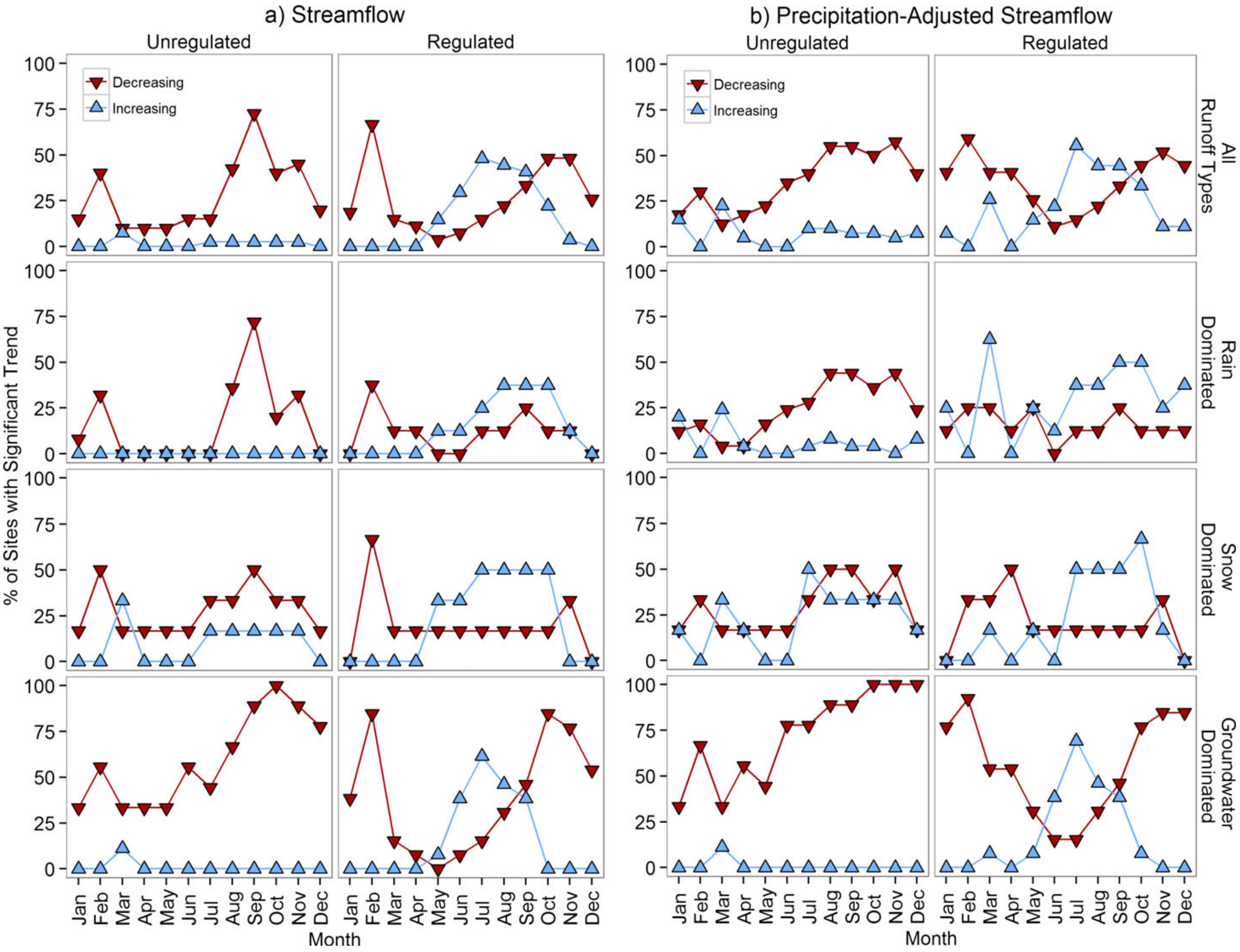

FIGURE 5. Percent of Streamflow Sites with Significant Increasing or Decreasing Trends in (a) Monthly Streamflow and (b) Precipitation-Adjusted Streamflow, Grouped by Runoff Type.

Pacific Coast of the U.S. (Stewart *et al.*, 2005; Fritze *et al.*, 2011). Two of six snowmelt-dominated sites (Scott River and Williamson River near Klamath Agency) also had later runoff, contrary to trends detected in some previous analyses (Regonda *et al.*, 2005; Hidalgo *et al.*, 2009) that found earlier runoff in other areas of the western U.S. (outside our study area) in response to climate warming causing earlier snowmelt and precipitation form shifting from snow to rain. Chang *et al.* (2012) detected very few significant trends in CT in unregulated streams in Oregon, Washington, Idaho, and western Montana for the years 1958-2008. Our results suggest that increased precipitation during the spring months (Figure 4a) has partially offset the effects of climate warming on spring runoff timing; however, it is uncertain whether increased spring precipitation will continue to occur.

Long-Term Trends in Precipitation-Adjusted Streamflow

Trends in precipitation-adjusted streamflow varied by month and degree of hydrologic regulation (Figures 4c, 5, and 8). Precipitation-adjusted streamflows declined significantly in at least one of the summer (July-September) months at 35 of 67 sites. Decreasing trends substantially outnumbered increasing trends for most months except June through September at regulated sites and January and March at unregulated sites (Figure 5). The months with highest percentage of unregulated sites with declining trends were July through November (40-58%). There were a greater percentage of site months with significant trends for precipitation-adjusted streamflow than for streamflow (Figures 4 and 5), likely because accounting

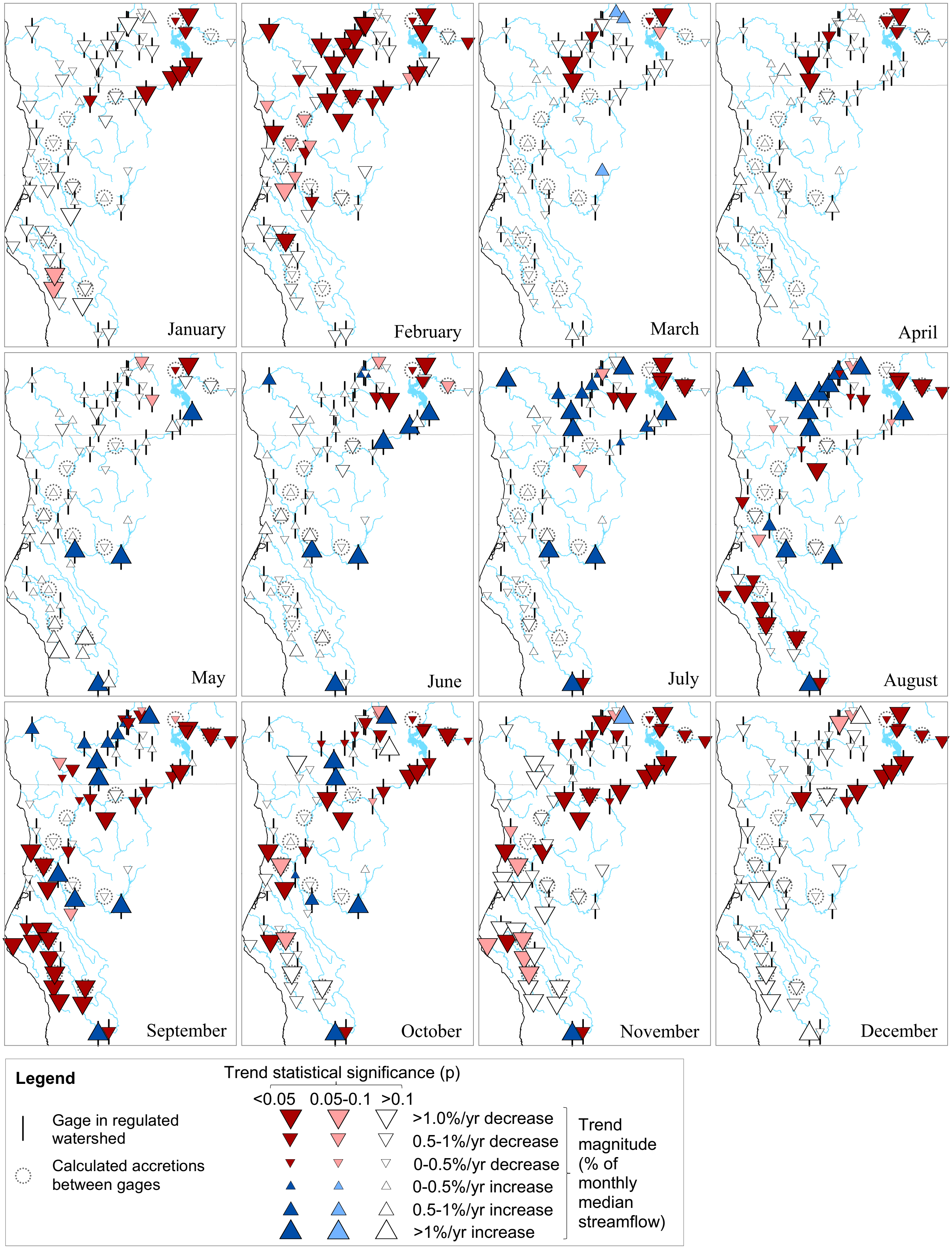

FIGURE 6. Trends in Mean Monthly Streamflow at Streamflow Sites, 1953-2012.

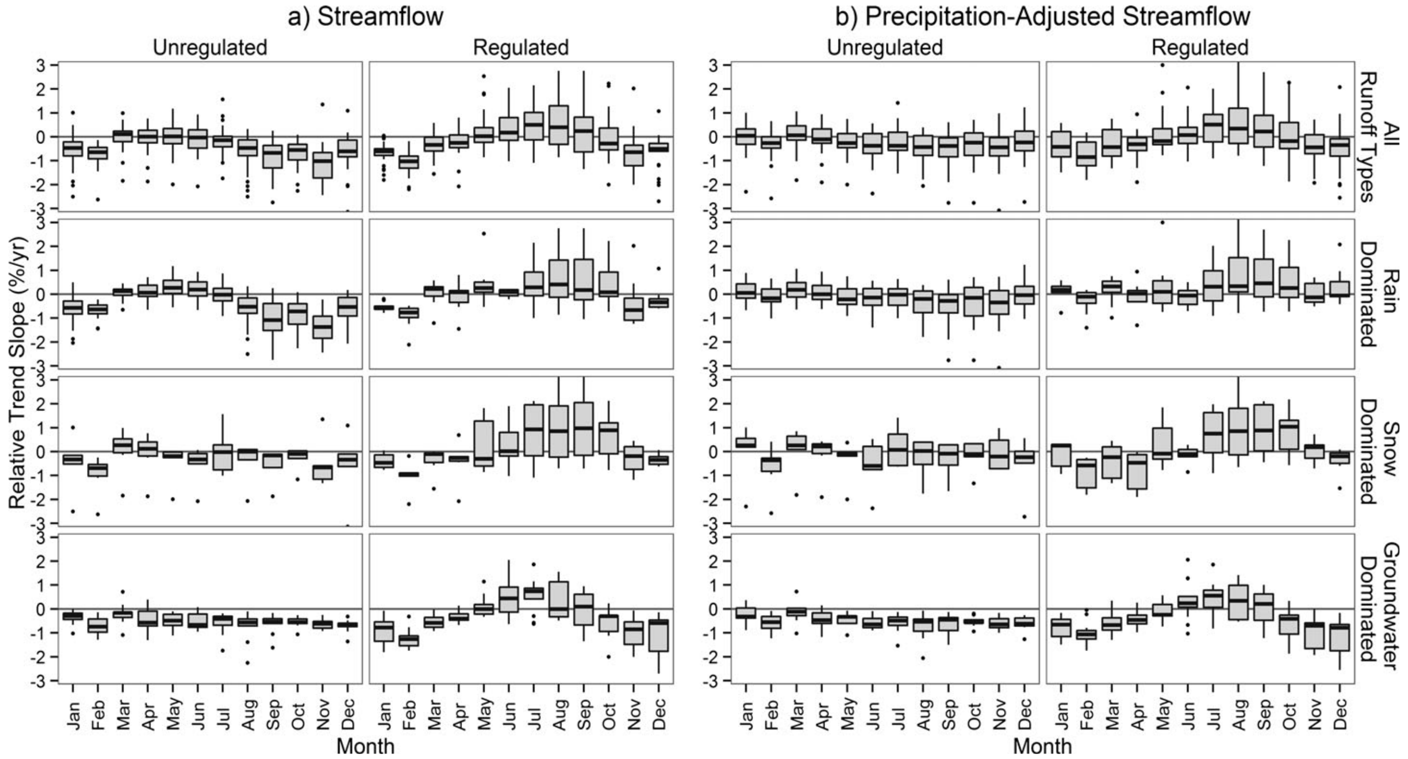

FIGURE 7. Relative Magnitude of Trends in Monthly (a) Streamflow and (b) Precipitation-Adjusted Streamflow, Grouped by Regulated/Unregulated Streams and Runoff Type. *Y*-axis is cropped for clarity, eliminating some outliers.

for precipitation reduces interannual variation that can obscure trends. As with streamflow, the percent of sites with declining precipitation-adjusted streamflow was greater for groundwater-dominated sites than other runoff types (Figure 5). In September at unregulated rainfall-dominated sites, the percent of sites with a declining trend (Figure 5), and the median trend magnitude (Figure 7), was smaller for precipitation-adjusted streamflow than for streamflow, coincident with declining September precipitation (Figure 4). The presence/absence and direction of trends in precipitation-adjusted streamflow matches the trends in base-flow recession reported by Sawaske and Freyberg (2014) for 12 of 14 gages included in both analyses.

Comparing the Sen slope of the streamflow trend with the Sen slope of precipitation-adjusted streamflow trend allows quantification of the relative contribution of precipitation to the observed trend in streamflow. A spatial pattern is apparent for unregulated sites in the month of September, which had the most widespread streamflow declines, with factors other than precipitation accounting for over >75% of the streamflow decline at many sites in the Upper Klamath Basin and Upper Rogue Basin as well as the Scott River, with lesser but still substantial amounts (30-75%) at many sites in the southwest portion of the study area (Redwood Creek, Mattole River, and Eel River Basin; Figure 9).

Potential Explanations for Trends in Precipitation-Adjusted Streamflow

The data and methods we used do not allow for quantification of the relative impacts of the various factors contributing to the declines in precipitation-adjusted streamflow, which include some combination of increased water withdrawals and/or increased vegetation/forest evapotranspiration. Increased vegetation/forest evapotranspiration could be due to changes in climate (i.e., air temperature, wind, humidity, or precipitation shifting from snow to rain) and/or forest structure/composition. By carefully examining the trends that have occurred over the study period in watersheds with contrasting conditions and histories, we can develop hypotheses about causal mechanisms that could be tested with additional analyses.

The most pristine surface-runoff dominated watersheds within the study area (i.e., those with very few water diversions, relatively little history of timber harvest, and few roads), such as Elder Creek, Smith River, Salmon River, and tributaries to the Klamath River between Seiad Valley and Orleans, showed no decreases in summer precipitation-adjusted streamflow (Figure 8). This indicates that streamflow decreases at other sites were likely due more to increased human withdrawals and vegetation changes than to climate factors other than precipitation quantity; however, as

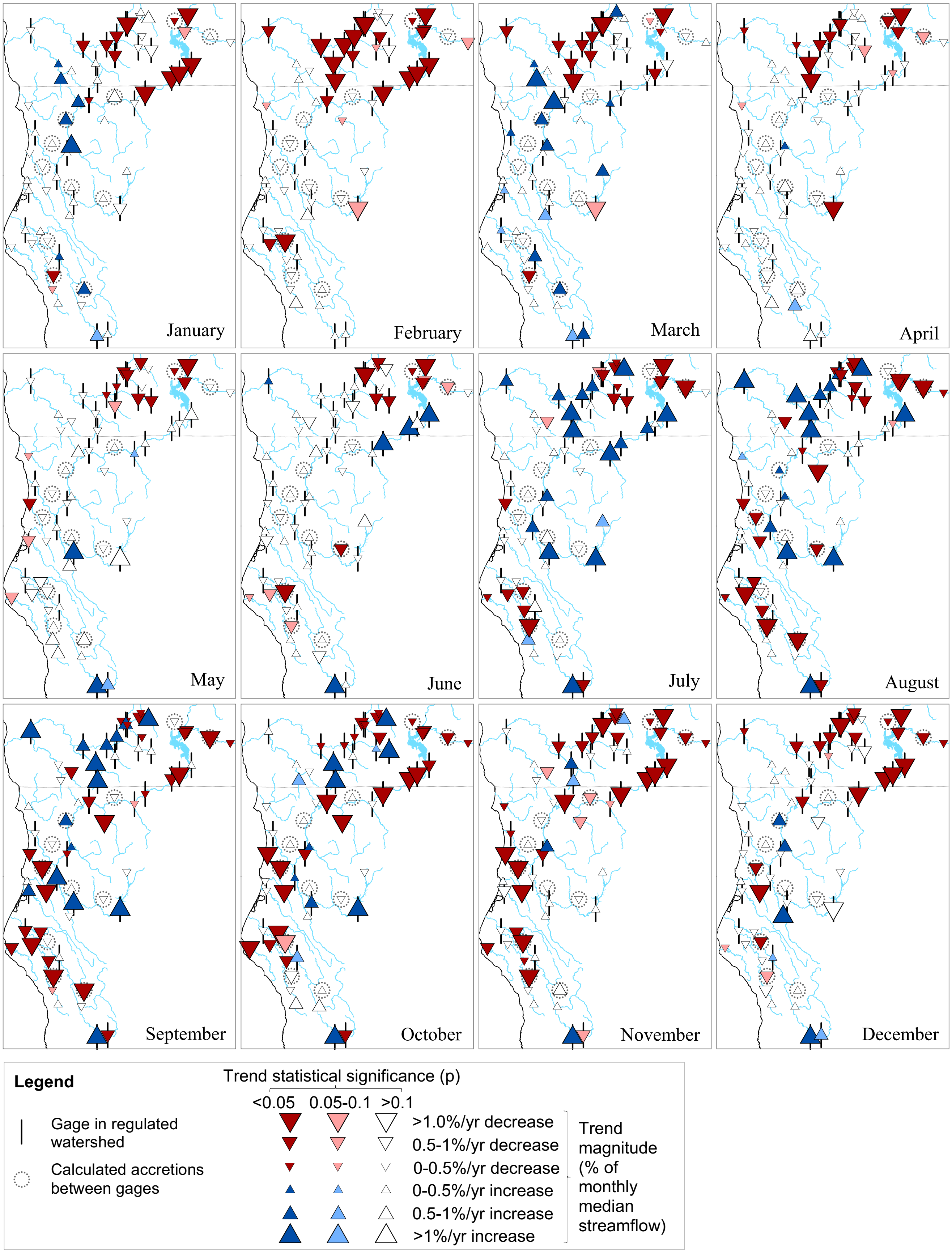

FIGURE 8. Trends in Precipitation-Adjusted Mean Monthly Streamflow at Streamflow Sites, 1953-2012.

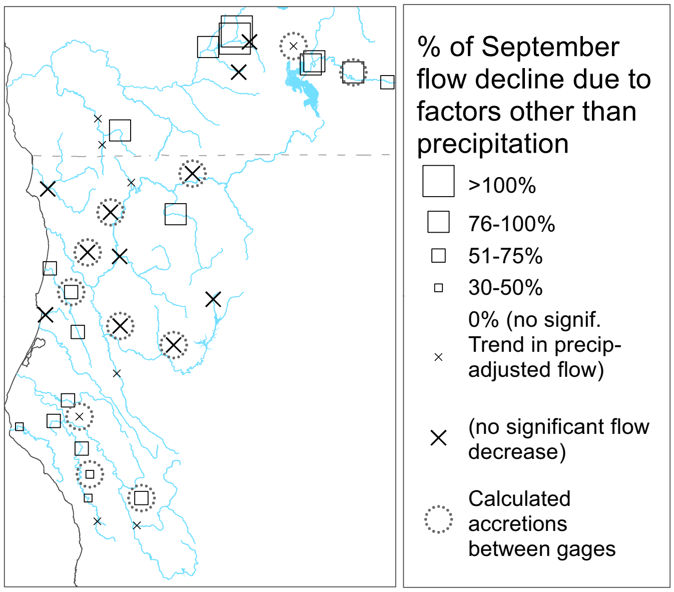

FIGURE 9. Percent of Magnitude of Declining 1952-2012 September Streamflow Trends Explained by Factors Other than Precipitation. Only unregulated sites are shown due to a stronger linkage between streamflow and precipitation. Values exceeding 100% indicate that the streamflow decline was due entirely to factors other than precipitation.

climate warming continues in future years, even the most pristine watersheds will likely experience summer streamflow declines. For example, in five Pacific Northwest basins outside our study area, the average predicted decrease in streamflow per 1°C of annual warming was 31, 21, and 7% for July, August, and September, respectively (Vano *et al.*, 2015).

Our results appear to support the hypothesis that water withdrawals are an important factor, but not the only one, contributing to the declining trends in precipitation-adjusted streamflows. There were few declines (though not none, e.g., Bull Creek and Rogue River Above/Below Prospect) in those watersheds with the least amount of diversions (e.g., those cited in the previous paragraph as well as Little River, South Fork Trinity, upper Trinity River, and accretions to the lower Trinity River). In the Scott River, where precipitation-adjusted summer streamflow declined (Figure 8), reductions in base flows since the 1970s have been attributed to increased groundwater pumping and decreased snow accumulation (Van Kirk and Naman, 2008). There is a general lack of data regarding small-scale domestic and agricultural withdrawals within the study area; however, Bauer *et al.* (2015) estimated water use for marijuana cultivation in four watersheds, including Redwood Creek near Blue Lake (gage 11481500) where our results show daily precipitation-adjusted streamflows for the month of September are declining at a rate of 166 m^3/day/yr (1.1% of the 15,178 m^3/day median daily September flow; Figure 8) yielding a total reduction of 9,957 m^3/day over the 60-year study period. Estimated daily water use of marijuana plants in the watershed was 523 m^3/day (Bauer *et al.*, 2015), equivalent to only about 5% of the total reduction in streamflow, which suggests other factors are also contributing to declining precipitation-adjusted streamflow.

Several lines of evidence suggest that changes to watershed vegetation affected the trends in precipitation-adjusted summer streamflow. First, evapotranspiration typically accounts for more than 50% of annual precipitation in forested watersheds (Zhang *et al.*, 2001), so relatively small changes could have large effects on low summer streamflows. Second, most forests within the study area have been harvested (NMFS, 2014), converting older forests first to clear-cuts which increase streamflow for a multiyear period immediately after harvest (Jones and Post, 2004; Jones *et al.*, 2009) but then result in young regenerating stands with high evapotranspiration rates in the following decades (Moore *et al.*, 2004; Jassal *et al.*, 2009; Creed *et al.*, 2014). For example, Bull Creek's gage was installed in 1961 soon after most of the watershed had been clear-cut (Stillwater Sciences, 1999) and as the forest has regenerated due to protection within a state park, summer/fall precipitation-adjusted streamflows have declined despite having almost no diversions (Figure 8). Bull Creek is still degrading through massive aggradation that occurred during the 1955 and 1964 floods (Stillwater Sciences, 1999), making the streamflow declines even more remarkable because recovery from aggradation would be expected to increase summer streamflow due to less infiltration into subsurface sediments. An alternative explanation for Bull Creek's trends is declining coastal fog (see below). A contrasting example is provided by Little River, which also has nearly no diversions but where timber has been actively harvested throughout the gaged record and precipitation-adjusted summer streamflow did not decline in any month (Figure 8). Third, fire suppression has allowed Douglas fir (*Pseudotsuga menziesii*) trees to encroach into prairies and oak woodlands (Engber *et al.*, 2011). Encroachment is likely occurring across large portions of our study area, including the Mattole, South Fork Eel, Van Duzen River, and Redwood Creek watersheds where summer precipitation-adjusted streamflow is declining (Figure 8); however, encroachment has not been well quantified except in the Bald Hills at the eastern edge of Redwood Creek where prairies were reduced by up to 44% between 1875 and 1998 (Fritschle, 2008) and the Little Bald Hills in the Smith River watershed where grass-dominated areas decreased by approximately 80% from 1942 to 2009 (Sahara *et al.*, 2015). Conversely, the Salmon River is the site with the greatest percent of area burned in wildfires in recent decades and is also the only unregulated stream with increasing precipitation-adjusted streamflow for all three months July-September (Figure 8) as well as the only gage for which Sawaske and Freyberg (2014) reported a decreasing trend in the rate of base-flow recession.

Another factor that could explain declining precipitation-adjusted streamflow in Bull Creek is that summer fog along the California coast declined during the 20th Century (Johnstone and Dawson, 2010). Annual wood production in old-growth redwood (*Sequoia sempervirens*) trees on Bull Creek's alluvial flats (downstream of the gaging station) was higher from 1970 to present than any time since at least 1750, likely due in part to reduced fog/cloud cover and increased light availability (Sillett *et al.*, 2013; Carroll *et al.*, 2014). However, precipitation-adjusted streamflow in Little River, which also has redwoods and coastal fog influence, did not decline (Figure 8).

CONCLUSIONS

Not surprisingly, regulation by dams appeared to exert a strong influence on trends in streamflow and precipitation-adjusted streamflow. Reservoirs store winter and spring runoff, increasing summer water supplies and providing a source to supplement withdrawal of summer streamflow. Whether increasing summer streamflow trends occurred at regulated sites depended in part on the timing of dam construction relative to the trend period evaluated (increasing trend in Rogue and Applegate rivers) and instream flow requirements (increasing trends in Eel and Trinity rivers). In basins without surface water storage reservoirs, the only sources available for water withdrawals in summer are diversion of streamflow and extraction of groundwater (which is often connected to streamflow). As a result, summer streamflow declines were much more common at unregulated than at regulated sites.

September precipitation decreased across almost the entire study area, but our application of a model of the relationship between antecedent precipitation and streamflows indicated that precipitation explained only a small portion of the observed declines in streamflow in most months. The most pristine surface-runoff dominated watersheds within the study area showed no decreases in precipitation-adjusted streamflow during the summer months, indicating that streamflow decreases at other sites were likely due to more increased human withdrawals and vegetation changes than to climate factors other than precipitation quantity. This is likely to change in the future as the increasing temperatures will increase evapotranspiration and decrease streamflow (Vano *et al.*, 2015).

Declining streamflows, which occurred primarily at unregulated sites in the summer and fall and regulated sites in the fall and winter, is a troubling indicator for the future of anadromous salmonid fisheries within the study area. Decreasing summer streamflow reduces the quality and quantity of pools available where juvenile fish can survive during the dry summer months (May and Lee, 2004). Declining fall flows could affect migration and spawning of adult salmonids, which use flow increases as migratory cues and a means by which to enter small streams (Shapovalov and Taft, 1954). The conventional approach to increasing summer water supply is construction of new dams and reservoirs. Dams have profound effects on river ecosystems, including impeding species migration and altering sediment dynamics (Ligon *et al.*, 1995; Graf, 2006), hydrology (Magilligan and Nislow, 2005), and food webs (Power *et al.*, 1996). Due to these effects, dams have been identified as a primary cause of declining salmon populations within the study area (Katz *et al.*, 2013; NMFS, 2014); thus, construction of new dams is unlikely to be a successful strategy for increasing summer streamflow without causing other detrimental effects to aquatic ecosystems. As an alternative to dam-based water storage, a program to equip rural residences with tanks to store spring and winter runoff for summer use has reduced summer water withdrawals and resulted in measureable increases in summer low flows in the Mattole River at the south end of the study area (Schremmer, 2014). Another potential method for increasing summer flows is to reduce forest evapotranspiration by harvesting trees or burning (Bosch and Hewlett, 1982); however, the hydrologic effects of single treatments are transient, repeated treatments can cause sedimentation and flooding (Jones *et al.*, 2009), and there are substantial obstacles to widespread implementation (Ziemer, 1987). A third approach for increasing summer flows is to increase the capacity of the landscape to store water by reconnecting floodplains and raising groundwater tables, including utilizing beavers (*Castor canadensis*, a mammal native to our study area; Lanman *et al.*, 2013) and beaver dam analogs (Beechie *et al.*, 2012; Pollock *et al.*, 2014). Finally, another essential step toward increasing streamflow is to reduce consumption of water for human uses.

SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article: Table of major dams in the study area; details about the API model used to calculate precipitation-adjusted streamflow including the calibration process, recession coefficients, and correlation coefficients; maps showing trend results for additional parameters (monthly precipitation, magnitude/timing of low flows, and center of timing); and charts showing absolute magnitude of trends in monthly streamflow and precipitation-adjusted streamflow.

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