

Linking climate to changing discharge at springs in Arches National Park, Utah, USA

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Abstract. Groundwater-fed springs are essential habitat for many dryland species. Climate projections forecast an increasingly arid climate for the southwestern United States. Therefore, an understanding of the relationships between climate and spring discharge is increasingly important. Monthly discharge measurements were recorded from 2001 to 2014 at three jointed bedrock springs in and near Arches National Park, Utah, United States. Discharge was compared with the potential evapotranspiration (PET) and precipitation derived from Daymet gridded climate data. Despite the similarities in location, aquifer type, and climate exposure, all three springs showed different responses to local climate. Two springs emerging from the western aquifer had decreases in discharge during differing portions of their record, while the eastern aquifer spring had stable discharge. At the monthly scale, there was a strong inverse relationship between measured discharge and PET at all three springs, likely due to vegetation accessing the water prior to its surface expression. Annual average winter discharge from both western aquifer springs responded to reductions in 10-year cumulative winter precipitation, while discharge from the eastern aquifer spring did not correspond well to precipitation within the period of record. Uncertainty in climate projections for aquifer recharge remains high, but increasing air temperatures will likely lead to increased PET and reduced spring surface flow. Better characterization of climate and spring discharge relationships will help managers protect contributing areas that may be more susceptible to groundwater withdrawal and better understand the available habitat for groundwater-dependent ecosystems and species.

Key words: Arches National Park; climate; groundwater; monitoring; NPS Inventory & Monitoring; Special Feature: Science for Our National Parks' Second Century; springs.

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INTRODUCTION

In dryland regions, groundwater-dependent ecosystems such as springs and seeps occupy a small fraction of the overall landscape, yet they support disproportionately high levels of productivity (Oberlin et al. 1999, Perla and Stevens 2008), biodiversity, and endemism

(Welsh and Toft 1981, Erman 2002, Hershler and Sada 2002, Sada et al. 2005, Spence 2008). Regionally, springs sustain critical habitat for threatened, endangered, and other rare species (Deacon et al. 2007). At broader spatial scales, springs play key roles in dryland ecosystems by providing refugia for migratory birds (Skagen et al. 1998) and serving as the primary source of

water for more extensive aquatic and riparian habitats (Miller et al. 2016).

In dryland ecosystems, potential evapotranspiration (PET) greatly exceeds precipitation. Groundwater recharge in these ecosystems is often episodic and localized, relying on high-intensity storms and/or preferential flow paths, such as jointed bedrock and alluvial stream channels (de Vries and Simmers 2002). Aquifers underlying vegetated drylands receive little to no recharge (Scanlon et al. 2005), and many dryland springs in basin settings rely on aquifers decoupled from recent changes in climate. In contrast, localized springs with shallow aquifers, such as those found in jointed bedrock systems, allow for modern recharge and may be sensitive to modern changes in climate (Green et al. 2011).

Current climate change models indicate that the dryland western United States faces an imminent increase in aridity due to higher air temperatures, decreasing snowpacks, and overall greater drought frequency (Seager et al. 2007, 2013, Cook et al. 2015). The decade 2001–2010 was the warmest and fourth driest in the southwest of all decades from 1901 to 2010 (Hoerling et al. 2013). Much uncertainty currently exists in how climate change will affect the amount, timing, and frequency of precipitation and thus groundwater recharge (McCallum et al. 2010, Ng et al. 2010, Crosbie et al. 2012). Increasing demand for groundwater is likely to further imperil the extent of groundwater-dependent ecosystems, such as springs and seeps (Green et al. 2011, Klove et al. 2014).

The importance of springs with localized recharge in preserving regional biodiversity in dryland ecosystems is likely to increase as surface water decreases and ancient and regional aquifers are drawn down. Because of their often rugged setting and relatively small aquifer areas, protecting these springs may represent an achievable goal for conservation. Better characterization of climate and spring discharge relationships will help land managers protect contributing areas that may be susceptible to groundwater withdrawal, forecast impacts to endemic species, and better understand spring-flow-dependent wildlife and their distributions.

In the absence of spatially extensive monitoring networks, long-term, site-specific data provide the only current context for evaluating the

links between recent spring discharge and climate. We investigate 14 years of discharge data for three jointed bedrock springs in and adjacent to Arches National Park, Utah, USA. For many springs, the relationships between climate and discharge are confounded by groundwater extraction and other anthropogenic influences. Our study provides a rare example of a long-term data set relatively free from anthropogenic influences whose discharge fluctuations are likely attributable to recent climate patterns. The findings from this work may be relevant to jointed bedrock springs with localized recharge in other dryland regions.

METHODS

Study sites

Arches National Park, Utah, United States, is located on the semiarid Colorado Plateau. Approximately 8% of the endemic flora in the park is comprised of spring-dependent species (Fertig et al. 2009). The majority of springs at the park emerge from the Moab Member of the Curtis Formation, a well-sorted (relatively uniform throughout) sandstone laid down in the Middle Jurassic period (Doelling 2001). The outcrop ranges from 18 to 36 m in thickness in the park (Graham 2004) at elevations from 1276 to 1665 m and is predominately exposed bedrock with sparse vegetation.

Recharge of the spring aquifers is from precipitation directly onto the highly jointed Moab Member outcrop exposed at the surface, and discharge in most cases emanates from the basal Moab Member at its contact with the underlying Entrada Formation (an impermeable sandstone) and within 8 m above that contact. Sedimentary layers above and below the Moab Member are relatively impervious, limiting the aquifer to this one stratigraphic unit (Hurlow and Bishop 2003). Hurlow and Bishop (2003) estimated that about 10% of winter precipitation enters the aquifer as recharge. The mean total annual precipitation (1981–2010) is 242 mm, and the mean annual temperature is 12.9°C.

The three springs in this study all emerge in the Courthouse Wash–Sevenmile Canyon system on the western side of the park (Fig. 1). The canyons dissect the Moab Member outcrop into distinct aquifer regions. Two springs, Western1

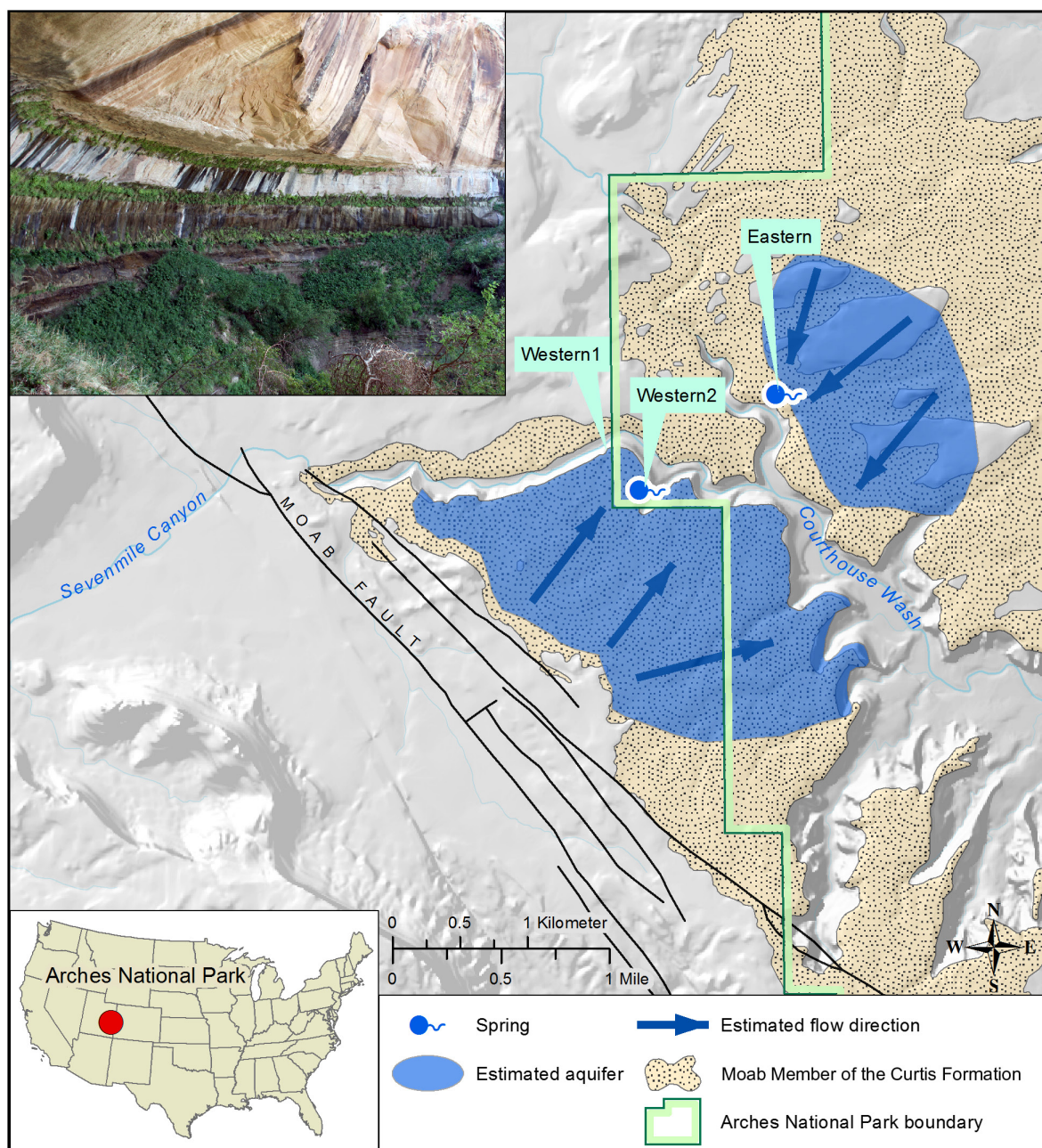


Fig. 1. Location of study springs and estimated aquifers (from Hurlow and Bishop 2003) at and near Arches National Park, Utah, United States. The photograph inset shows the typical architecture of a hanging garden at Arches National Park, with seeping flow emerging from geologic contact lines and through vegetated colluvial slopes. Flow coalesces at the base of the slopes into channels where the discharge measurements can be taken.

and Western2, emerge on the western side of Courthouse Wash in a tributary called Sevenmile Canyon. Their aquifer recharge areas are estimated as 1.7 and 0.9 km², respectively, and the recharge areas extend beyond the national park boundary (Hurlow and Bishop 2003). The Moab Fault acts as a structural barrier to groundwater flow from the west, and no existing groundwater withdrawals are likely to affect recharge to these springs (Weissinger and Moran 2015). The third spring, Eastern, emerges on the eastern side of Courthouse Wash. The aquifer recharge area, estimated as 1.7 km², is entirely protected within the park boundary (Hurlow and Bishop 2003). Groundwater dating at these sites estimated an age of less than 30 years, indicating relatively rapid turnover within the aquifer (M. Masbruch, *unpublished data*).

All three springs are hanging gardens that emerge as diffuse, seeping flow from geologic contacts within bedrock cliffs (*sensu* Springer and Stevens 2008, see photograph inset Fig. 1). These contacts are exposed at the surface and also emerge beneath vegetated colluvial slopes. Diffuse flow coalesces into an outflow channel, where spring discharge can be measured. Diffuse flow is estimated to coalesce into measureable flow between 15 and 30 m in a straight-line distance downstream of the geologic contact zones at our springs. However, diffuse flow traveling through the vegetated colluvial slopes can have variable distances to reach the outflow channel. Thus, spring discharge at these sites can be affected by evapotranspiration prior to measurement, which could be a substantial portion of flow during warmer seasons.

Data collection

Monthly spring discharge measurements from March 2001 to December 2014 at each site were taken by capturing and channelizing flow through a steel plate with an outflow pipe. After flow stabilized upstream of the plate, six timed volumetric measurements were taken by filling a vessel of known volume. The six individual measurements were averaged. Monthly measurements at all three sites were taken on the same day and were scheduled to avoid recent precipitation events that could affect flow via runoff. Prior to the analysis, datapoints that were flagged

as unrepresentative of spring discharge due to runoff events, icy conditions, or leaking plates were removed from the data set (Weissinger and Moran 2015). Suspect data represented only one percent of the measurements. Missing data account for an additional 6–10% of each spring's data set.

Due to the close proximity of springs, we used a single 1-km pixel time series of Daymet climate data (Thornton et al. 2014) centered on the study area to estimate the temperature and precipitation that affects recharge and flow. We chose Daymet specifically because it spans the period of flow measurements and is a daily climate product with all of the parameters needed for computing physically based estimates of PET (Thornton et al. 1997). Daily precipitation was summed to obtain the monthly estimates. We estimated PET via the simple temperature-based Hamon method and the more complex and physically based Penman method (Allen et al. 1998, Dingman 2002). Potential recharge metrics included (1) total annual and seasonal precipitation and (2) total annual and seasonal precipitation adjusted for PET by subtracting monthly PET from monthly precipitation and then summing the positive remainders.

Analytical methods

At each of the three springs, we asked a series of questions: Is discharge changing over time? Can climate variables of PET and precipitation-based recharge metrics explain the monthly patterns? How are long-term trends in discharge related to precipitation and precipitation-derived recharge metrics?

Monthly models.—To explore the nature of the relationships between measured discharge, time, PET, and recharge metrics, we fit a series of linear and general additive models (GAMs; Wood 2006) using R 3.2.3 (R Core Team 2016) and the core stats function `lm()` and `gam()` from package `mgcv` (Wood 2006). GAMs are a nonparametric extension of generalized linear models that have no a priori assumptions about the shape of the response (*i.e.*, linear, quadratic). Given the differences among springs in aquifer size, aspect, pool size, and shape, we treated spring identity as a categorical (factor) fixed effect. All of our models include this main effect for the differences among springs. However, when testing either trends

over time or predictors of discharge, we compare the models with and without interactions involving spring to test for the differences in response between springs.

We took a sequential approach to analyzing these data, with the form of each test dependent on the results of the previous tests. Instead of comparing all plausible models, we kept the best of the first set of models as the base for the second set, and compared different forms of the second factor added to that base. For example, we first examined the explanatory value of Hamon PET vs. Penman PET to select the most explanatory PET estimate prior to including recharge metrics in the model.

We had no a priori reason to expect linear trends or linear responses to predictors. For sequential tests of hypotheses, forcing linear forms to nonlinear relationships can fail to fully account for earlier predictors, and have additional predictors driven by that nonlinearity. Therefore, at each step, we fit models with and without smoothing via penalized smoothing using the gam function. For example, to test for temporal trends in discharge for each spring, accounting for an interaction between time and the individual spring, we fit both linear (Eq. 1) and smoothed (Eq. 2) models of the forms:

$$\text{Discharge} \sim \text{Spring} + \text{Date} + \text{Spring}:\text{Date} \quad (1)$$

$$\text{Discharge} \sim \text{Spring} + \text{s}(\text{Date}, \text{by} = \text{Spring}) \quad (2)$$

The degree of smoothing in the second model was selected using generalized cross-validation. For penalized smoothing of PET and date, we used thin plate regression splines.

We compared the adequacy of the monthly models with the Akaike information criterion (AIC) and the Bayesian information criterion (BIC); the latter penalizes additional parameters more than AIC does. Where BIC and AIC agree, the model selection is robust to the amount of penalization for model complexity, but where they disagree, the definition of “best” model is problematic. Because smoothed functions do not have simple interpretable parameters to report, when the comparison of GAMs indicated that linear responses were adequate, we estimated the parameters and the confidence intervals of those estimates via conventional linear models.

Finally, we took two different approaches to addressing the possible effects of recharge on measured discharge. First, we added same month recharge metrics to models including spring-specific responses to the variation in PET to test whether recharge metrics improved the fit over PET alone. Second, because the aquifer is likely to both buffer (average out) short-term variation in recharge and possibly delay the response in terms of discharge, we looked at short-term cumulative and lagged recharge metrics. We graphically explored a range of averaging 1–6 months of precipitation and recharge and lagging that average by 0–12 months as predictors of spring-specific variation in discharge for the 20 months with PET equal to 0.

Annual models.—We also looked for an effect of precipitation on discharge at the annual time scale. At each spring, we averaged the discharge measurements from the nongrowing season (November–March for our study area) and summed the precipitation for the same time period each year, hereafter referred to as winter. A majority of recharge is estimated to occur in these months (Hurlow and Bishop 2003), and PET is both small and consistent across years, so by focusing on winter discharge and precipitation, we can omit the annual variation in PET in these annual models. We then used simple linear regression analyses to explore discharge at each spring with cumulative and lagged winter precipitation at that annual time step. Based on the available period of record for Daymet precipitation (1980 to present), winter precipitation was summed for up to 10 years and lagged by up to 10 years. Because all models included a single summed lagged precipitation predictor and the same observations (13 complete winters of discharge 2001–2013 for each spring), we used AIC as a measure of model fit and the ΔAIC for model comparisons for each spring within each model set. We considered models with $\Delta\text{AIC} < 4$ as competitive candidate models (Burnham and Anderson 2002). Given the large number of combinations of summing window and lagging duration ($11 \times 11 = 121$), the logic of this approach was not to dredge the data to find a particular model that “fit,” but rather to examine whether any coherent set of models fit well enough to suggest rough magnitudes of averaging and lagging in the aquifer.

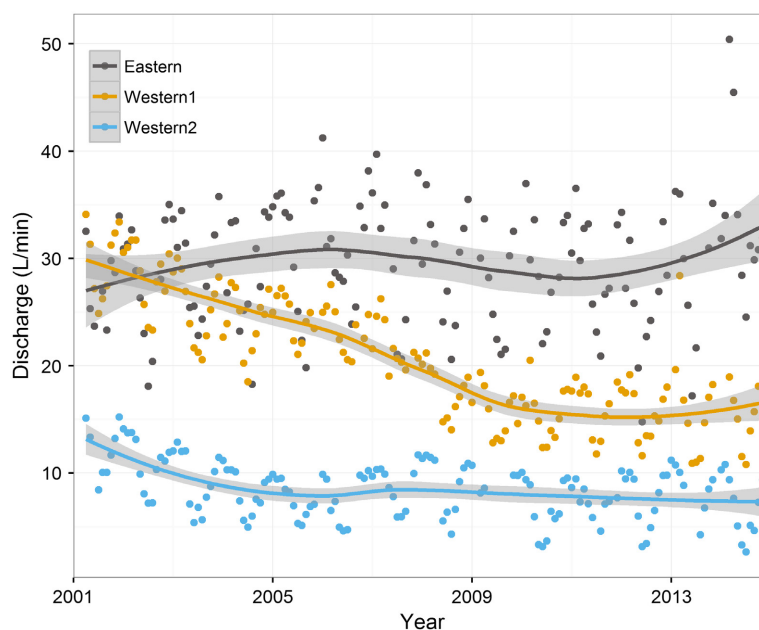


Fig. 2. Trends in spring discharge over the 14-year time series for each spring with a loess smooth of 0.75 and 95% confidence interval. Both western aquifer springs had decreasing discharge early in their record, while the eastern aquifer spring has remained relatively stable over time.

RESULTS

Temporal trends in discharge

Despite being located in close proximity and in similar topographic and aquifer settings, our three springs showed different trends in discharge over time (Fig. 2). The two springs emerging from the western aquifer both had declining discharge over time, while the Eastern spring remained relatively stable over the 14-year period. Whether these raw trends are linear or curved is ambiguous, with BIC favoring linear and AIC favoring modest amounts of smoothing (Table 1: model 2 vs. model 4). In either case, the models with spring-specific means and temporal trends were much better than the models without those terms (model 2 beats 1 and model 4 beats 3). Repeating the analyses with only June–September dry season data yields the same results, with a smoothed model preferred (Table 2: models 5–8).

Beyond these longer-term trends, substantial within-year variation in discharge was also apparent for each spring (Fig. 2). Western1 and Western2 had median annual ranges of 7.8 and 6.9 L/min compared with the median annual flows of 19.7 and 9.0 L/min, respectively. Median

Table 1. Model comparisons testing linear vs. non-linear smoothed trends over time, and pooled vs. spring-specific trends, in measured discharge at the monthly scale.

No.	Model	df	BIC	AIC
1	Discharge ~ Spring + Date	5	2652.5	2632.0
2	Discharge ~ Spring + Date + Date:Spring	7	2549.3	2520.6
3	Discharge ~ Spring + s(Date)	8.4	2653.3	2618.9
4	Discharge ~ Spring + s(Date, by = Spring)	17	2557.4	2487.8

Notes: s() indicates a term that was fit with a smooth, with the optimal amount of smoothing determined via generalized cross-validation. The lowest BIC and AIC are bolded and underlined.

declines in flow over the period of record corresponded to approximately 14.4 L/min at Western1 and 7.6 L/min at Western2 (Weissinger and Moran 2015). Eastern's median annual range was much higher at 16.5 L/min, with a median annual flow of 30.0 L/min.

Explanatory value of PET and recharge metrics for monthly discharge

Monthly increases in PET corresponded to lower discharge at all sites. For any given model

Table 2. Model comparisons testing measured discharge for low-flow months only (June–September).

No.	Model	df	BIC	AIC
5	Low Flow ~ Spring + Date	5	814.0	798.8
6	Low Flow ~ Spring + Date + Date:Spring	7	750.3	729.1
7	Low Flow ~ Spring + s(Date)	10.7	821.4	788.9
8	Low Flow ~ Spring + s(Date, by = Spring)	20	743.8	683.4

form, Hamon PET produced a better fit than Penman PET (Table 3: models 9–12), and the slopes of the lines differed between springs (Fig. 3). A linear model was favored using BIC, while AIC favored smoothing (Table 3: model 10 vs. model 12). Both models explain nearly 84%

of the variation in monthly discharge for the overall model (linear model-adjusted $r^2 = 0.837$; smoothed model-adjusted $r^2 = 0.841$). As expected based on the differences in aspect, evaporative area, and vegetated cover between sites, individual springs had varying intercepts and slopes (models 10 and 12 with separate slopes per spring were much better than their corresponding models with a single slope for all springs, 9 and 11, respectively). Adding monthly precipitation or recharge did not improve the fit for models based on linear Hamon PET for seasonality (Table 4: models 13–20). Further, models with PET and precipitation fit much worse than the corresponding models with smoothed terms for date added (Table 3: models 21–22). When graphed, none of the smoothed and lagged recharge

Table 3. Comparison of monthly Hamon PET and Penman PET as a predictor of measured monthly discharge.

No.	Model	Hamon PET			Penman PET		
		df	BIC	AIC	df	BIC	AIC
9	Discharge ~ Spring + PET	5	2561.4	2540.8	5	2587.8	2567.3
10	Discharge ~ Spring + PET + PET:Spring	7	2542.4	2513.6	7	2573.4	2544.7
11	Discharge ~ Spring + s(PET)	7.6	2563.7	2532.7	7.8	2598.7	2550.5
12	Discharge ~ Spring + s(PET, by = Spring)	11.3	2553.4	2507	15.3	2594.7	2533.7

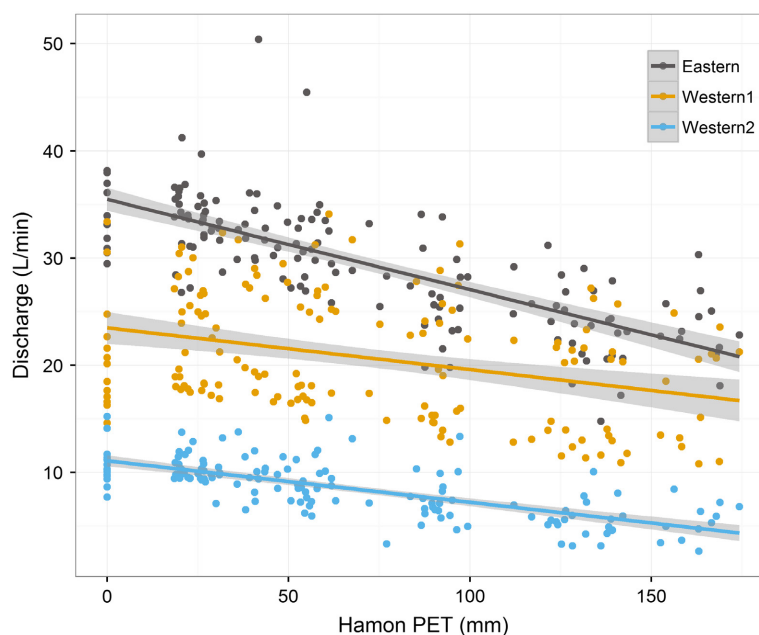


Fig. 3. Higher potential evapotranspiration (PET) is correlated with lower measured spring discharge at the monthly time step. Lines are linear models with 95% confidence interval.

Table 4. Comparison of precipitation, recharge (precipitation adjusted for PET), and date as predictors of measured monthly discharge.

No.	Model	df	BIC	AIC
13	Discharge ~ Spring + PET + PET:Spring + Precip	8	2550.2	2517.4
14	Discharge ~ Spring + PET + PET:Spring + Precip + Precip:Spring	10	2559.6	2518.6
15	Discharge ~ Spring + PET + PET:Spring + s(Precip)	8	2550.2	2517.4
16	Discharge ~ Spring + PET + PET:Spring + s(Precip, by = Spring)	12.7	2572.0	2520.0
17	Discharge ~ Spring + PET + PET:Spring + Recharge	8	2550.2	2517.3
18	Discharge ~ Spring + PET + PET:Spring + Recharge + Recharge:Spring	10	2556.2	2515.1
19	Discharge ~ Spring + PET + PET:Spring + s(Recharge)	11.5	2564.0	2517.0
20	Discharge ~ Spring + PET + PET:Spring + s(Recharge, by = Spring)	14.7	2568.5	2508.0
21	Discharge ~ Spring + PET + PET:Spring + Date + Date:Spring	10	2234.4	2193.4
22	Discharge ~ Spring + PET + PET:Spring + s(Date, by = Spring)	27	<u>2177.5</u>	<u>2066.7</u>

metrics showed strong relationships to measured discharge across the months of PET = 0. Variation in precipitation or recharge from month to month does not have a simple correlation with the monthly variation in discharge. Therefore, even after accounting for PET and monthly precipitation, spring-specific trends in discharge remain.

Winter precipitation as a driver of annual average winter discharge

Annual winter precipitation varied considerably over the Daymet time series, while the 10-year cumulative winter precipitation shows an abrupt decline near the time when spring discharge was beginning to be measured (Fig. 4). For the western springs, models with the lowest AIC involved summing precipitation over 8–10 years and 6–7 years of lagging, aligning the general decline in winter precipitation in 1983–1989 into a smooth decline from 1994 to 2004 (Fig. 5), and shifting it to align with the spring-specific decline in discharge visible in Fig. 2. These models are alignments of a single decline in each time series and thus have little predictive

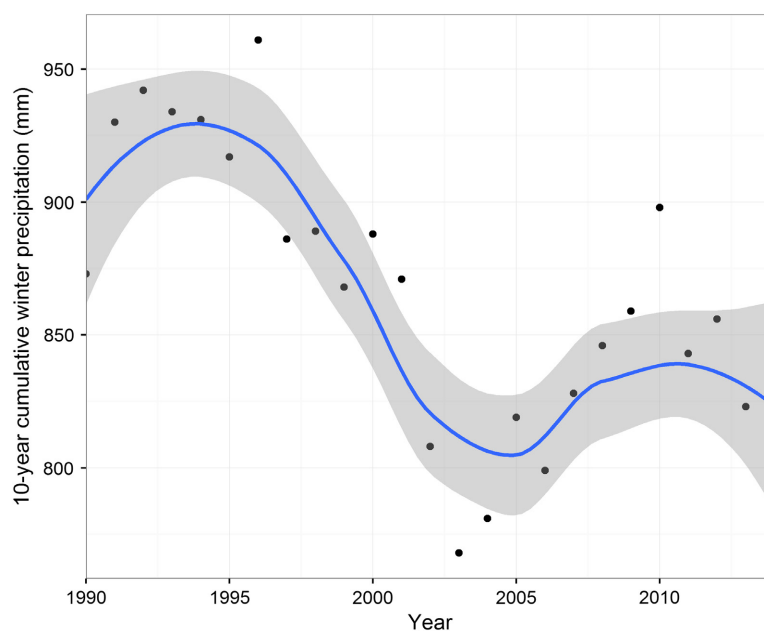


Fig. 4. Ten-year cumulative winter precipitation derived from Daymet climate data from a 1-km pixel centered on the study area with a loess smooth of 0.75 and 95% confidence interval.

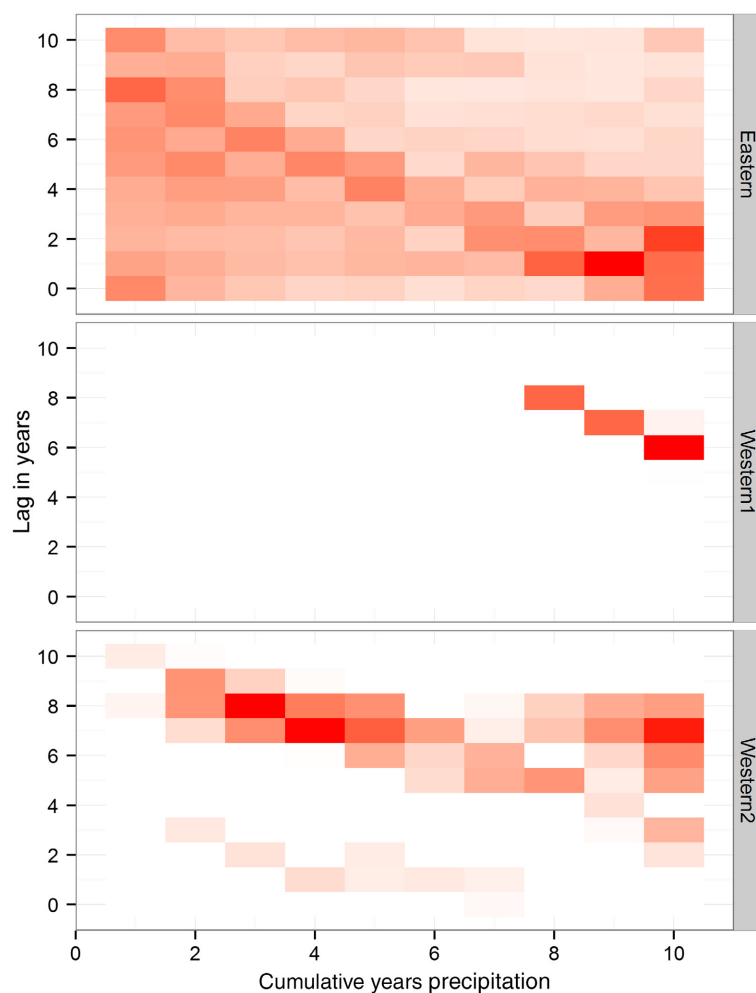


Fig. 5. Heat maps of ΔAIC for simple linear regression analysis testing different combinations of cumulative and lagged winter precipitation to predict the average winter discharge for each spring. Values of $\Delta AIC < 4$, represented by darker colors, were considered competitive models. Western aquifer springs had competitive models with 10 years of cumulative precipitation and 6–7 years of lag, while the eastern aquifer spring had little differentiation between models.

or explanatory power. For the Eastern spring, almost all models (93/110; 85%) had $\Delta AIC < 4$ (Fig. 5). The best fit model of one-year cumulative and no lagged precipitation explained almost no variation in annual discharge ($r^2 = 0.03$).

DISCUSSION

Dryland springs as unique entities

Springs are highly complex ecosystems that naturally vary in water quantity (spring discharge), water quality, biological communities, topographic setting, and disturbance regimes. Previous

studies in the dryland western United States have shown that even springs in close physical proximity can have distinct habitat characteristics and biotic components (Malanson and Kay 1980, Spence and Henderson 1993, Sada et al. 2005, Meretsky 2008). Our study sites in and near Arches National Park support the individualistic nature of dryland springs and extend these findings to spring discharge dynamics that form the basis for these ecosystems. Despite similarities in topographic setting, aquifer type, and climate exposure, each spring in our study responds to local precipitation and recharge in a unique

manner. Monthly variations in flow are strongly tied to the PET of diffuse near-surface flow through vegetated slopes, which is in turn affected by varying degrees of sun exposure resulting from variation in slope, aspect, and shading.

For our study sites, short-term changes in precipitation, such as a wet year or a dry year, are not sufficient to change the trajectory of long-term discharge trends. The stability in flow seen at Eastern spring is likely due to geologic controls, including a larger storage volume for the eastern aquifer, a larger bedrock outcrop to capture recharge, and greater connectivity with surrounding formations that can exert an increased hydrostatic pressure on the aquifer. In contrast, the western aquifer is bounded on its western edge by a significant fault, which limits its storage volume, recharge area, and landscape connectivity (Hurlow and Bishop 2003). Western1 spring seems to respond most directly to the recent patterns in local climate, while Western2 spring has a more complex response. If the estimated time frame of 10 years of precipitation with a lag of 6–7 years holds true, discharges are likely to increase from the western aquifer sites in the next few years in response to the recent increases in 10-year cumulative precipitation. In our record, recent measurements at Western1 and Western2 springs appeared to be stabilizing at a new, lower discharge than earlier in the record.

Predicting future discharge at our springs is difficult. Uncertainties in future precipitation due to human-caused climate change lead to widely differing future recharge scenarios in dryland systems, including both increasing and decreasing recharge (McCallum et al. 2010, Ng et al. 2010). High-intensity rainfall events have been linked to increasing air temperatures (Allan and Soden 2008) and diffuse groundwater recharge in dryland systems (Scanlon et al. 2005, Crosbie et al. 2012). However, in jointed bedrock systems, much high-intensity rainfall is lost to runoff and higher-intensity storms may not lead to increased recharge.

Increased PET decreases surface water availability at springs

Despite the uncertainties in precipitation relationships, our data show a clear relationship between seasonally high PET and decreasing spring discharge. The temperature-based Hamon

model of PET outperformed a more complex PET model, as was found in another study that estimated the observed output from a water balance model (Oudin et al. 2005). In our case, PET is acting at the spring discharge location, where diffuse discharge is subject to transpiration and evaporation prior to measurement. Vegetation accesses near-surface water flowing through colluvial slopes prior to its surface expression, which affects the availability of surface water downstream during the growing season.

Using 10-, 20-, and 30-year moving windows from 1901 to 2012, Monahan and Fisichelli (2014) showed that maximum air temperatures in the warmest month and mean temperatures of the warmest quarter have increased at Arches National Park to the extremes of their historic range. Annual precipitation has remained highly variable, and precipitation within each quarter remains within its historic range. The driest month has moved to the extreme wettest of its historic range. Similar temperature patterns have occurred at national park units throughout the western United States, while precipitation patterns are mixed (Monahan and Fisichelli 2014).

Further increases in air temperature are expected with human-caused climate change, and regional estimates project a 23% increase in evapotranspiration over the next 70 years (Ficklin et al. 2013). In the late Quaternary, rapid climate change caused extensive drying of wetland systems in the dryland western United States (Springer et al. 2015). The decrease in spring discharge with an increase in PET at our springs indicates that reductions in surface water availability can be expected with increasing temperatures if increases in precipitation do not counter the observed trends. For springs in dryland regions, this increases the risk of drying, with profound ecological consequences for species dependent on the surface expression of groundwater.

Implications for management and conservation of groundwater-dependent ecosystems

Groundwater stress has already been noted in the western United States (Famiglietti and Rodell 2013) and is expected to increase under human-caused climate change. Our study suggests that even springs with relatively little direct anthropogenic influence will be affected by increases in

air temperature and correspondingly higher rates of evaporation and transpiration by wetland vegetation. Given the ecological importance of water resources in dryland landscapes, tracking spring discharge on a site-by-site basis is essential for land managers to understand what habitat is available for preserving groundwater-dependent ecosystems and species. To maximize conservation potential, managers could consider prioritizing deeply shaded, north-facing, and/or higher-elevation springs for protection and restoration.

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