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Evaluation of Hanging-Garden Endemic-Plant Monitoring at Southeast Utah Group National Parks, 2013–2020

Natural Resource Report NPS/NCPN/NRR—2022/2473



ON THE COVER

Hanging garden, Arches National Park. NPS photo.

Evaluation of Hanging-Garden Endemic-Plant Monitoring at Southeast Utah Group National Parks, 2013–2020

Natural Resource Report NPS/NCPN/NRR—2022/2473

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Executive Summary

Hanging gardens are the most common type of spring at Arches National Park (NP) and Natural Bridges National Monument (NM). They are also present at Canyonlands National Park, but hanging gardens are rare off the Colorado Plateau. Their cliffside setting provides stable access to water without flood disturbance. This combination provides unique habitat that is rich in endemic plant species. The diffuse, seeping emergence of water makes measuring springflow impossible at most sites.

Park managers have an interest in monitoring hanging gardens—especially as the climate warms and aridity and water demand both increase. The Northern Colorado Plateau Network (NCPN) proposed methods for monitoring seven perennial endemic-plant species at hanging gardens as indicators of spring health and proxies for water availability. Because hanging gardens occur on bedrock outcrops, systematic or random sampling was not possible due to safety concerns and potential resource damage on steep, wet slopes. Examining eight years (2013–2020) of data, this report evaluates the suitability of endemic-plant count data at hanging gardens as a monitoring indicator. It also provides our first evaluation of status and trends at NCPN hanging gardens.

The seven species included in monitoring were Rydberg's thistle (*Cirsium rydbergii*), Kachina daisy (*Erigeron kachinensis*), alcove death camas (*Zigadenus vaginatus*), alcove bog orchid (*Habenaria zothecina*), cave primrose (*Primula specuicola*), alcove columbine (*Aquilegia micrantha*), and Eastwood's monkeyflower (*Mimulus eastwoodiae*). Six of the seven species were found at each park. Up to 500 individuals of each species were counted at 42 hanging gardens in Arches NP, 14 hanging gardens in Natural Bridges NM, and 3 hanging gardens in Canyonlands NP. Larger populations were divided into count classes of 501–1,000, 1,001–10,000, and more than 10,000 individuals.

Counts from two independent observers and from back-to-back years of sampling were compared for repeatability. Repeatability in count classes was less than 50% for Kachina daisy and Eastwood's monkeyflower, which both propagate vegetatively via ramets and/or stolons. Repeatability was greater than 90% for only one species, Rydberg's thistle. The remaining species were categorized in different classes between 15–40% of the time. Independent-observer comparisons were only available for 6.6% of the dataset, but these observations suggested that (1) observer bias was present and (2) the observer with more experience working in hanging gardens generally had higher counts than the observer with less experience in this system.

Although repeatability was variable, it was within the range reported by other studies for most species. The NCPN, in discussion with park staff, has elected to make some modifications to the protocol but will continue using endemic plant counts as an indicator of hanging-garden health to maintain a biological variable as a complement to our physical-response data. This is due to their high value to park biodiversity and the difficulty of developing a more robust approach to monitoring in these sites. Endemic-plant monitoring will continue for the five species with the highest repeatability during pilot monitoring and will focus on detecting changes in smaller populations. Most hanging gardens have more than one endemic species present, so several populations can be tracked at each site.

Our period of record is relatively brief, and the distribution of endemic-plant populations in different count classes at these sites has not yet shown any statistical trends over time. Because of the large count classes, our methods are more sensitive to showing change in smaller populations (fewer than 500 individuals). Small populations are also of greatest concern to park managers because of their vulnerability to declines or extirpation due to drought. Overall, more sites had endemic-plant populations of fewer than 100 individuals at the end of our

record than at the beginning, with Ryberg's thistle and alcove bog orchid having the greatest proportion of small populations at monitoring locations. These two species and alcove columbine also had a statistically significant decline in counts when small-population data (<500 individuals in at least one count) were pooled across parks.

Drought conditions were increasingly common for the latter half of the record and are predicted to increase as the climate warms. Drought effects, such as orchid counts of zero at some sites in Natural Bridges NM, and woody-species stress and mortality, were noted at some sites beginning in 2020. We anticipate that endemic-plant populations at hanging gardens will increasingly be affected by drought. Continued monitoring of hanging gardens can provide park managers with early detection of these changes, even when statistical trend detection lags behind.

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1 Introduction

1.1 Why springs?

In semi-arid regions such as the Colorado Plateau, springs and seeps occupy a tiny 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, Sada et al. 2005, Spence 2008). They are often the only water source for many miles around and are focal points of human and wildlife activity (Shepard 1993, Hubbs 1995).

Current climate change models indicate the US Southwest faces an imminent increase in aridity due to decreased snowpack, increased temperatures, and increased drought frequency (Seager et al. 2007, Seager et al. 2013). Globally, a lack of groundwater observations has hindered understanding of interactions between springs and climate (Taylor et al. 2013). Groundwater-recharge projections vary based on changes in projected rainfall. In the upper Colorado River Basin, recharge is projected to increase slightly (Tillman et al. 2016), while southern portions of the Plateau are expected to see decreases in recharge (Niraula et al. 2017). In previous work, however, ecologists with the Northern Colorado Plateau Network (NCPN) found that regardless of recharge, climate change plays an important role in water availability at three hanging gardens in Arches National Park (Weissinger et al. 2016). For these sites, potential evapotranspiration explained about 80% of the seasonal variability in measurable springflow. With increasing temperatures, spring-adapted species will require more water to maintain their current populations and are likely to experience increasingly frequent drought.

Regionally, springs and other groundwater-dependent ecosystems are of increasing interest, but data on these systems remain scarce. Existing protocols focus primarily on characterizing springs at a point in time (e.g., Stevens et al. 2016, USFS 2012), rather than monitoring over the longer term. There are few to no published data on springs-vegetation monitoring on the Colorado Plateau or the arid southwestern US.

1.2 NCPN springs monitoring

The National Park Service (NPS) Southeast Utah Group (SEUG) manages four units: Arches National Park (ARCH), Canyonlands National Park (CANY), Hovenweep National Monument, and Natural Bridges National Monument (NABR). These parks were established to protect a diverse array of natural and cultural resources, but they are similar in that their semi-arid setting results in a landscape where water is a limited and often isolated resource. The protection of water resources in these parks is a high priority for park managers (Cudlip et al. 1999).

In 2000, SEUG began a program for monitoring water quality at selected springs (Schelz and Moran 2004, 2005a, 2005b). Water-quantity measurements were added in 2004. In 2007, data management and reporting for the project were transferred to the NCPN, under the auspices of regional water-quality and springs monitoring. Monitoring of flow and water quality at selected springs follows standardized methods and has been successful in documenting status and trends; these results are reported elsewhere (e.g., Moran 2011, Weissinger and Moran 2013, Weissinger et al. 2016). The program's purposes can be framed with the following questions:

- What are the ranges in natural conditions for selected springs at NCPN parks?
- What are the current status and long-term trends in selected parameters at springs monitored in NCPN parks?

In 2013, NCPN expanded the springs-monitoring program to include additional sites, with a focus on a spring type unique to the Colorado Plateau, the hanging garden (Weissinger et al. 2017). Hanging gardens are a particular type of spring that develop in highly permeable sandstones along bedding planes or at the contact with a less-permeable layer (Welsh 1989; Springer and Stevens 2009). They can vary from small seasonal patches with little to no vegetation to lush, plant-filled alcoves. Much of the water emerging at hanging gardens is often taken up immediately by plants. This, in combination with their cliffside setting,

makes water-quantity monitoring difficult or, in many cases, impossible.

Hanging-garden vegetation is strongly differentiated from the surrounding landscape, and was selected as the monitoring metric of choice for these systems. Approximately 8–9% of known regional and local endemic species that occur at Arches National Park (NP) (6 of 72) and Natural Bridges National Monument (NM) (6 of 65) are hanging-garden specialists (Fertig 2009, Fertig et al. 2009). As perennial wetland species, the endemic plants targeted for springs monitoring are also useful proxies for water availability at hanging gardens, where water quantity cannot be measured.

But developing high-quality long-term monitoring metrics for hanging gardens has proved challenging. Attempts to develop a repeatable method for monitoring vegetation composition and structure at hanging gardens have encountered many difficulties. A systematic, quadrat-based method was attempted during pilot sampling but was discontinued due to concerns about safety and resource damage, particularly in the steep bedrock and colluvial-slope settings at hanging gardens (Weissinger et al. 2017).

Instead, geomorphic, surface-based methods were developed for descriptive use by Spence (as described in Spence 2008) and used for springs inventories at hanging gardens throughout the Colorado Plateau during protocol development (Springer et al. 2006). While not ideal because of the lack of a standardized sampling area and the lack of subsamples within a site, they were selected as the basis for pilot monitoring, with the caveat that their usefulness as indicators of hanging-garden health would need to be evaluated after testing.

1.3 Purpose and scope

The purpose of this report is to evaluate endemic-plant count data collected at long-term monitoring sites of hanging gardens in Arches NP, Natural Bridges NM, and Canyonlands NP from 2013 to 2020 for repeatability and sensitivity to change over time. (Hanging gardens are not present at Hovenweep NM.) It also presents the baseline status and initial trend analysis for the program. This period represents a full monitoring panel of all selected sites at the parks. Seven endemic species were quantified in count classes on backwalls, alcoves, and colluvial slopes of hanging gardens, as well as for the total spring area. A random sample of accessible hanging gardens was included at two parks, with 42 sites monitored at ARCH and 14 sites monitored at NABR. Three sentinel sites were monitored at CANY.

1.4 Study area description

The four parks of the NPS Southeast Utah Group (Figure 1-1) are well-known for their dramatic canyons and structures from precolonial Puebloan cultures. The geology of the region is primarily sedimentary, and sandstones and sand-dominated formations serve as both regional and local aquifers. Water travels through these porous bedrock aquifers until it encounters a less-permeable layer. In the canyon-bound systems of southeast Utah, these seeplines often emerge along bedding planes within more-expansive sandstone cliffs or at the contact between a sandstone aquifer and an underlying geologic layer. These cliff-side seeps form unique habitats known as hanging gardens (Figure 1-2). At Arches NP, hanging gardens emerge in outcroppings of the Moab Member of the Curtis Formation, while at Natural Bridges NM they are associated with the Cedar Mesa sandstone. Geology is more variable at Canyonlands NP, with hanging gardens emerging from both the Wingate and Navajo sandstones.

The presence of perennial, low-flow seeps and absence of flooding at hanging gardens creates a stable, mesic environment. In eolian sandstones, groundwater seepage creates protective alcoves that reduce solar radiation, temperature, and evaporation potential when compared to surrounding areas (May et al. 1995). These isolated oases are strongly differentiated from the surrounding landscape and have become focal points for vascular-plant endemism on the Colorado Plateau.

Of the 20 endemic-plant species that occur in Northern Colorado Plateau springs,

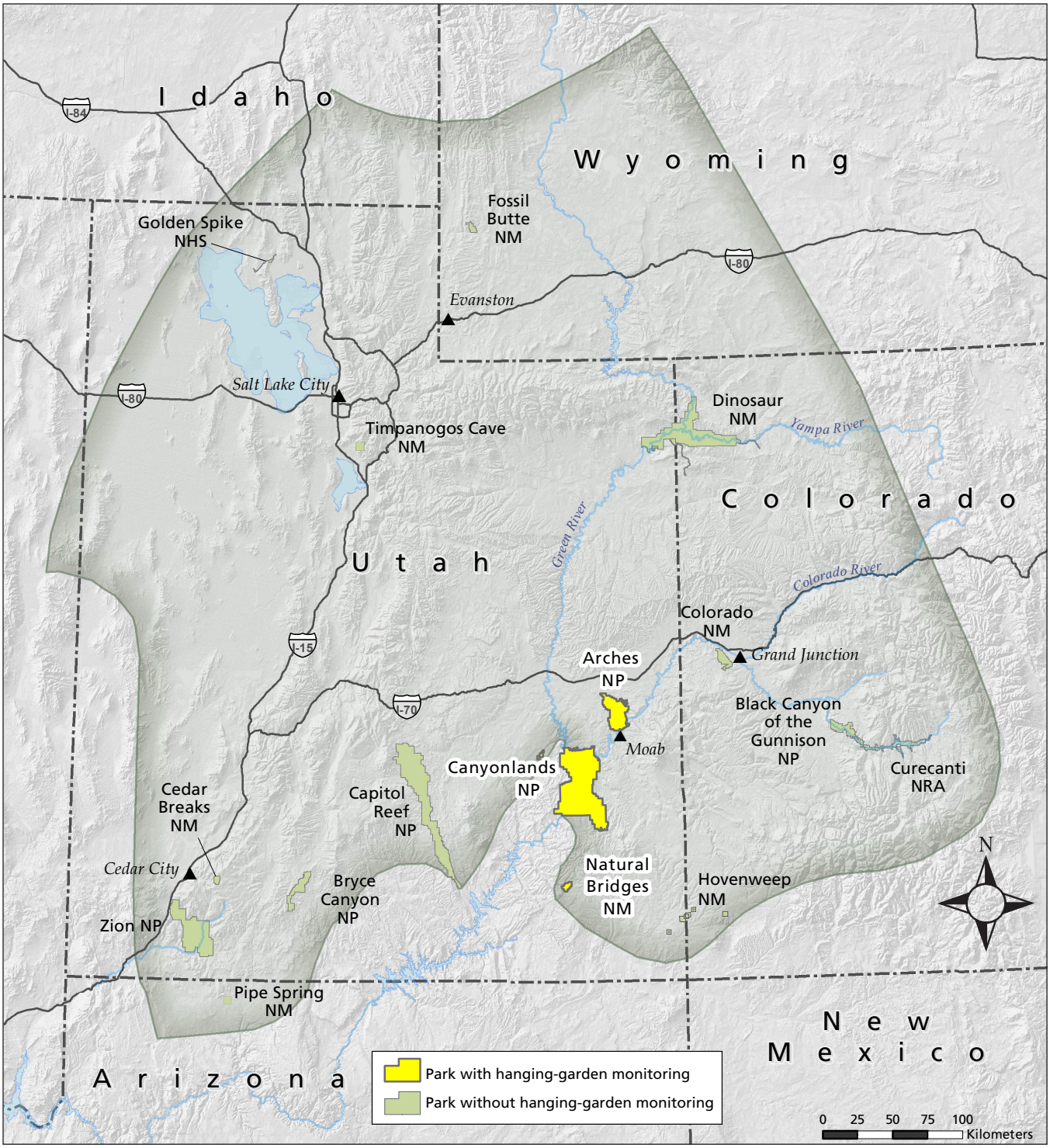


Figure 1-1. Parks of the Northern Colorado Plateau Network. Parks where hanging gardens are monitored are highlighted in yellow.



Figure 1-2. Example of a hanging-garden backwall and colluvial slope at Arches National Park.

most are strongly associated with hanging gardens, rather than adjacent riparian or wetland habitats (Spence 2008). Fowler (1995) found that approximately 5% (12 of 201) of vascular-plant species observed at 84 hanging gardens in seven national parks on the Colorado Plateau were endemic to the region, and about 3.5% (7 of 201) were endemic specifically to hanging gardens on the Plateau.

The hanging-garden flora is a combination of species found in similar moist regional habitats and relict species from northern and eastern biomes (Fowler 1995). Hanging-garden endemic plants are characterized by being perennial, showing late maturity, and producing relatively few seeds (Keate 1996), indicating adaptation to a stable habitat. Seven species occurring in hanging gardens at ARCH, CANY, and NABR were selected for pilot monitoring (Figure 1-3). These species are described below. Because the primary audience for this report is park managers, we provide the current scientific names as determined by the International Code of Botanical Nomenclature (ICBN, www.itis.gov accessed 9/8/2022), but preferentially use nomenclature from Welsh et al. (2008), which is the most-accessible taxonomic key for these parks.

1.5 Species descriptions

1.5.1 Cirsium rydbergii

Welsh: *Cirsium rydbergii* – Compositae
ICBN: *Cirsium rydbergii* Petr. (Asteraceae)

Rydberg's thistle is a large, perennial herb (Figure 1-3) that is easy to identify when visiting a site due to its "huge basal rosettes, tall slender flowering stems, and small heads" (Welsh et al. 2008). It is confined to hanging gardens, or rarely in canyons below them, at 1,125–1,525 meters in southeastern Utah and northeastern Arizona (Welsh et al. 2008). This species is non-clonal with separate genetic individuals. This species occurs in all three parks.



Figure 1-3. Endemic species counted during hanging-garden monitoring at Arches National Park and Natural Bridges National Monument, 2013–2020. Clockwise from top left: *Cirsium rydbergii* (Rydberg's thistle), *Erigeron kachinensis* (Kachina daisy), *Zigadenus vaginatus* (alcove death camas), *Mimulus eastwoodiae* (Eastwood's monkeyflower), *Aquilegia micrantha* (alcove columbine), *Primula specuicola* (cave primrose), and *Habenaria zothecina* (alcove bog orchid).

1.5.2 Erigeron kachinensis

Welsh: *Erigeron kachinensis* – Compositae
ICBN: *Erigeron kachinensis* S.L. Welsh & Glen Moore (Asteraceae)

The Kachina daisy (Figure 1-3) is a perennial herb with a substoloniferous habit that distinguishes it from most other *Erigeron* species. It is endemic to the Colorado Plateau in Garfield and San Juan counties in Utah and Montrose County in Colorado. It occurs from 1,680 to 1,890 meters in hanging gardens at lower elevations and on higher-elevation slopes in aspen and Ponderosa pine (Welsh et al. 2008). This species occurs only at Natural Bridges NM.

1.5.3 Zigadenus vaginatus

Welsh: *Zigadenus vaginatus* – Liliaceae
ICBN: *Anticlea vaginata* Rydb. (Melanthiaceae)

The alcove death camas (Figure 1-3) occurs in hanging-garden communities in seeps and alcoves from 1,200 to 1,900 meters in several counties in Utah and Colorado. It is a perennial herb with white, paniculate flowers that has a genetic affinity to *Zigadenus elegans* (Palmquist et al. 2015) but is found at lower elevations. Although considered non-clonal, the species sprouts repeatedly from the base, forming dense clumps (Palmquist et al. 2015). This species is known to occur at all three parks.

1.5.4 Habenaria zothecina

Welsh: *Habenaria zothecina* – Orchidaceae
ICBN: *Platanthera zothecina* (L.C. Higgins & S.L. Welsh) Kartesz & Gandhi (Orchidaceae)

The alcove bog orchid is a perennial herb found in moist stream banks, seeps, and hanging gardens in mixed desert shrub, pinyon-juniper, and oakbrush communities at 1,220 to 1,895 meters in several counties in Utah, Arizona, and Colorado (Welsh et al. 2008). It can be easily distinguished from other *Habenaria* species by the spur length when in flower (Figure 1-3). When small, it is possible to confuse *H. zothecina* with *Epipactis gigantea*, a common hanging-garden orchid, but the two species' vegetative characteristics diverge as the plants mature and are distinguishable once three or more leaves are present. Due to its small size, it is possible to overlook this species when it grows in thick vegetation. This species is non-clonal with separate genetic individuals. It is known to occur at all three parks.

1.5.5 Primula specuicola

Welsh: *Primula specuicola* – Primulaceae
ICBN: *Primula specuicola* Rydb. (Primulaceae)

Cave primrose is a perennial herb found in hanging gardens in southern Utah and northern Arizona from 1,125 to 1,680 meters (Welsh et al. 2008). Its distinctive, silvery rosette is easy to distinguish (Figure 1-3). It blooms in early spring, and the dried umbel can remain attached to the rosette after the showy pink flowers have faded. This species is known to occur at Arches and Canyonlands NPs. While it has been found in the surrounding county, it has not yet been recorded at hanging gardens in Natural Bridges NM.

1.5.6 Aquilegia micrantha

Welsh: *Aquilegia micrantha* – Ranunculaceae
ICBN: Aquilegia micrantha Eastw. (Ranunuculaceae)

Alcove columbine is the only columbine endemic to hanging gardens in southeastern Utah, northeast Arizona, and southwest Colorado. It grows at between 1,125 and 1,895 meters. With distinctive flowers and foliage, it is one of the most-common species at hanging gardens in SEUG parks (Figure 1-3). Glandular foliage and small, pale flowers distinguish it from other columbines found in the region (Welsh et al. 2008). This species is non-clonal with separate genetic individuals. *Aquilegia micrantha* occurs at all three parks.

1.5.7 Mimulus eastwoodiae

Welsh: *Mimulus eastwoodiae* – Scrophulariaceae
ICBN: Mimulus eastwoodiae Rydb. (Phrymaceae)

Eastwood's monkeyflower (Figure 1-3) is a perennial herb with leafy stolons and bright-red blossoms. It is found in hanging gardens in southeast Utah, northeast Arizona, and southwest Colorado, from 1,125 to 1,375 meters (Welsh et al. 2008). This species grows almost exclusively on backwall habitats and often forms patchy mats of hundreds to thousands of stolons in suitable areas. *Mimulus eastwoodiae* occurs at all three parks.

2 Methods

Monitoring methods are detailed in Weissinger and others (2017). Here we provide a review of hanging garden-specific methods evaluated in this report.

2.1 Spatial and temporal sampling design

Hanging gardens are the most common spring type at Arches NP and Natural Bridges NM. Populations of hanging gardens at these parks were documented during a rare-plant inventory in 2009 and 2010 (Weissinger et al. 2011). Sites were located using a combination of documentation from previous rare-plant surveys and opportunistic surveys for accessible sites during the inventory. At ARCH, 72 hanging gardens were identified, with populations further split into three geographic subcategories based on canyon systems: Courthouse Wash (n = 29), Salt Wash (n = 20), and Freshwater Canyon/Delicate Arch (n = 23). At NABR, 19 accessible sites were identified. An attempt to inventory hanging gardens at Island in the Sky in Canyonlands NP determined that the accessible population was too small to support a random draw. Instead, three specific sites were selected to be monitored as sentinel sites, ranging in size from an estimated 328 to 2,772 m^2 in size (Figure 2-1).

Within each population at ARCH and NABR, individual hanging gardens were assigned a number in GIS. A random number generator (accessed at www.random.org) was then used to create a random order without replacement within each population. Each hanging garden was sampled in the randomly generated order until the affordable sampling effort was reached based on a programmatic commitment of 80 person hours per year for fieldwork. At ARCH, 42 sites are monitored (58% of the documented accessible population), 14 in each subpopulation (Figure 2-2). These sites range from an estimated 46 to 4,260 m^2 in size. At NABR, 14 sites are monitored, 74% of the documented accessible population (Figure 2-3). These sites range from an estimated 14 to 656 m^2 in size. It should be noted that the amount of hanging-garden habitat within each park is greater than the accessible portion documented, and relatively continuous seeplines often continue beyond the extent of the monitored seeps. The true extent of hanging-garden habitat at these parks is unknown.

The revisit design for hanging-garden populations attempts to balance the need to observe interannual variability with the need to maximize replication (Woodward 2004). A rotating-panel design is used to increase the overall sample size and provide a better estimate of population status. Randomly sampled hanging gardens are sampled once during the growing season per sampling year, two years in a row, then rested for three years. One site per subpopulation was subjectively selected for its perceived resistance to trampling; these sites are monitored yearly to track responses to unpredictable or cyclical climate events.

2.2 Field methods

Hanging gardens are particularly difficult to sample because of safety and resource-disturbance concerns. Any type of systematic or random sampling would result in unacceptable risks to the field crew or damage to the site. Our field methods focus on accessible portions of hanging gardens, where traversing the hanging garden will not cause displacement of plant roots or expose the field crew to serious falls.

Rough dimensions of hanging gardens are collected during an initial site visit using a laser rangefinder. Separate geomorphic surfaces are mapped onto graphing paper. Hanging garden-specific surfaces of the backwall, alcove floor, and colluvial slope are noted, although not all three surfaces are present at all hanging gardens. The backwall is a near-vertical bedrock surface below the seepline bedding plane, while the colluvial slope is an often-steep build-up of soil at the base of the backwall (see Figure 1-2). Alcove floors are near-flat surfaces at the base of a backwall or colluvial slope that are sometimes present due to disturbances, such as historic livestock trampling or human use. The ease of thoroughly censusing these surfaces is generally associated with the presence of an alcove floor, the presence

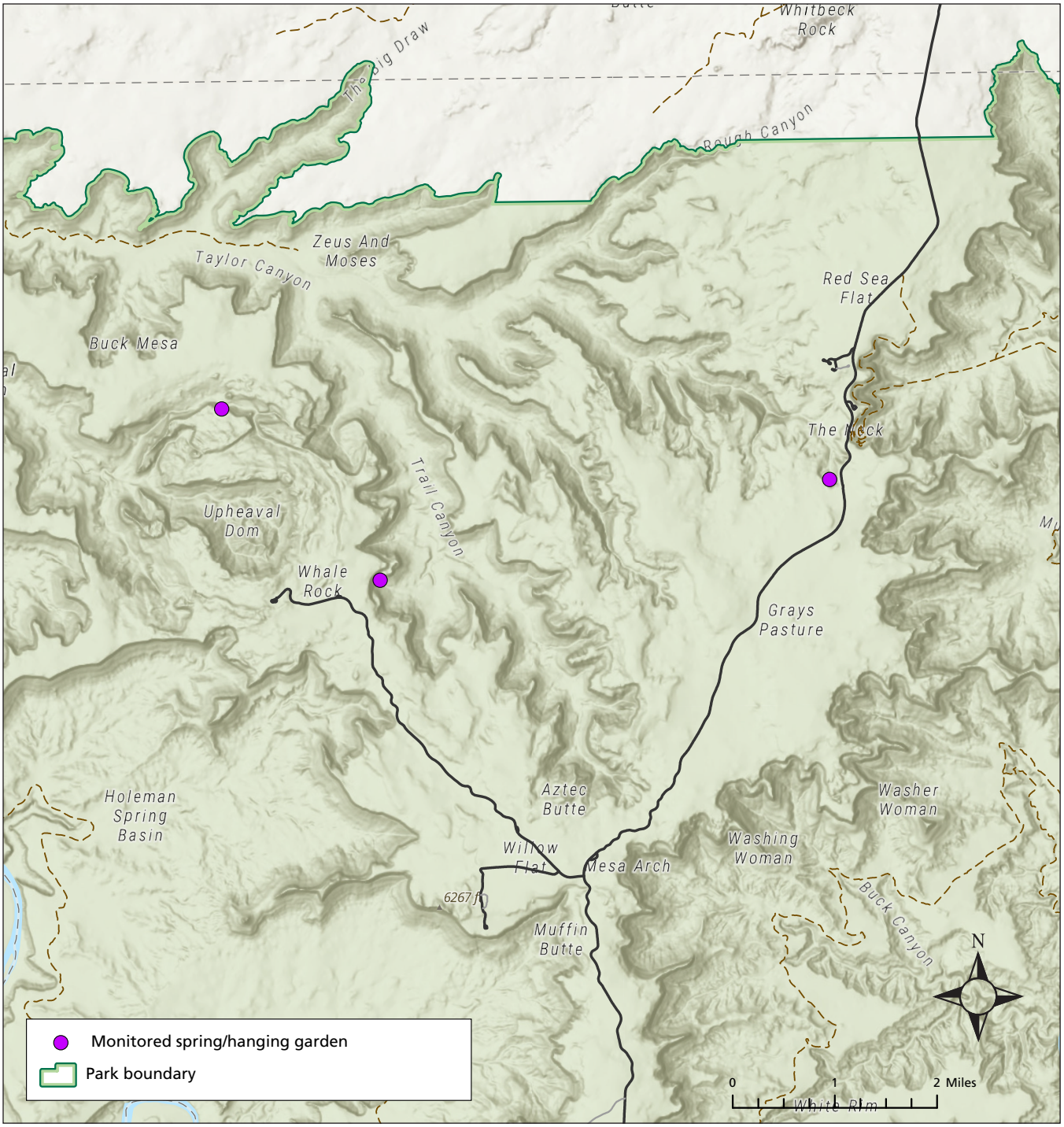


Figure 2-1. Three hanging-garden sentinel sites were selected for monitoring at Canyonlands National Park.

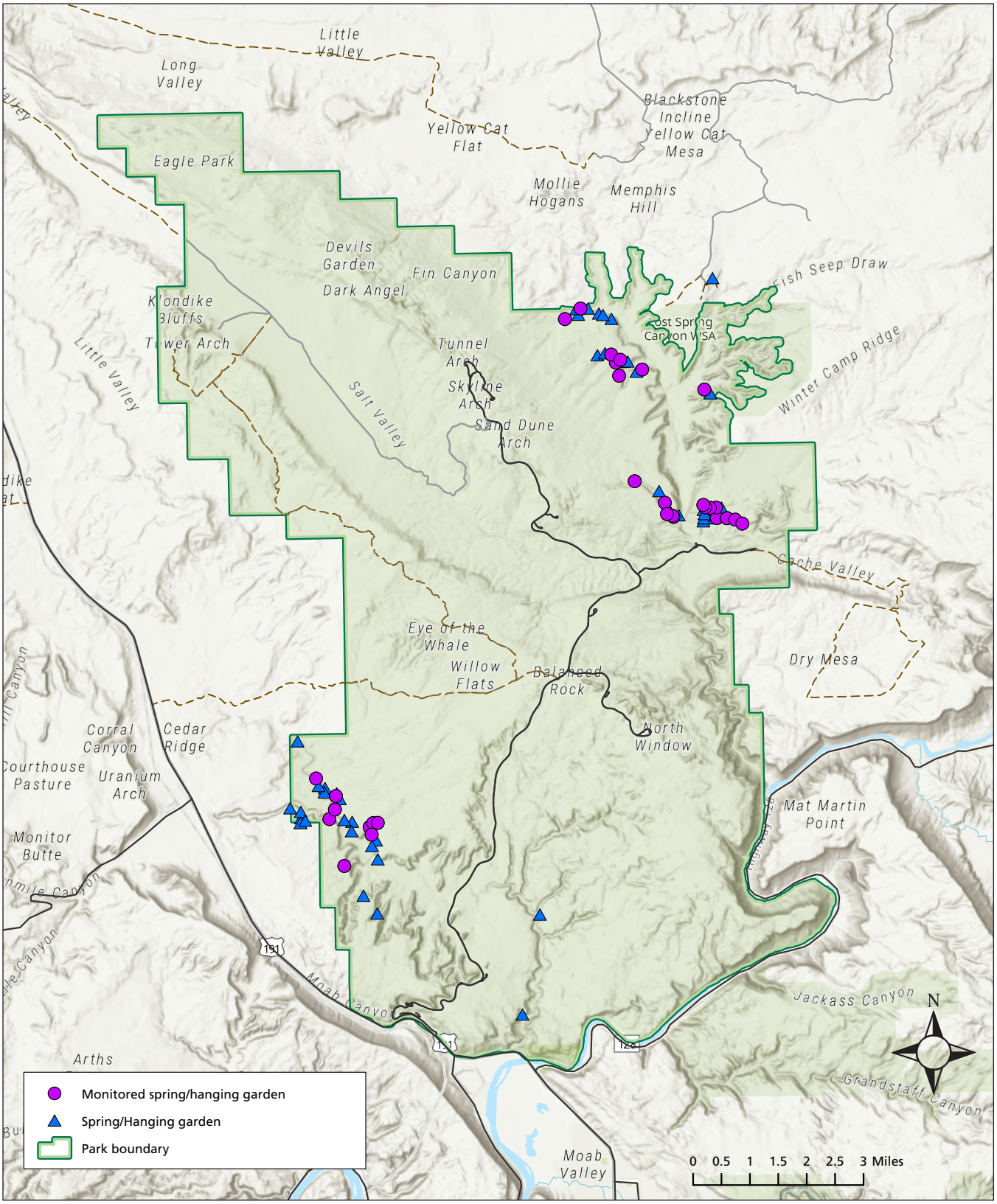


Figure 2-2. Hanging gardens at Arches National Park, with 42 of 72 sites selected for monitoring.

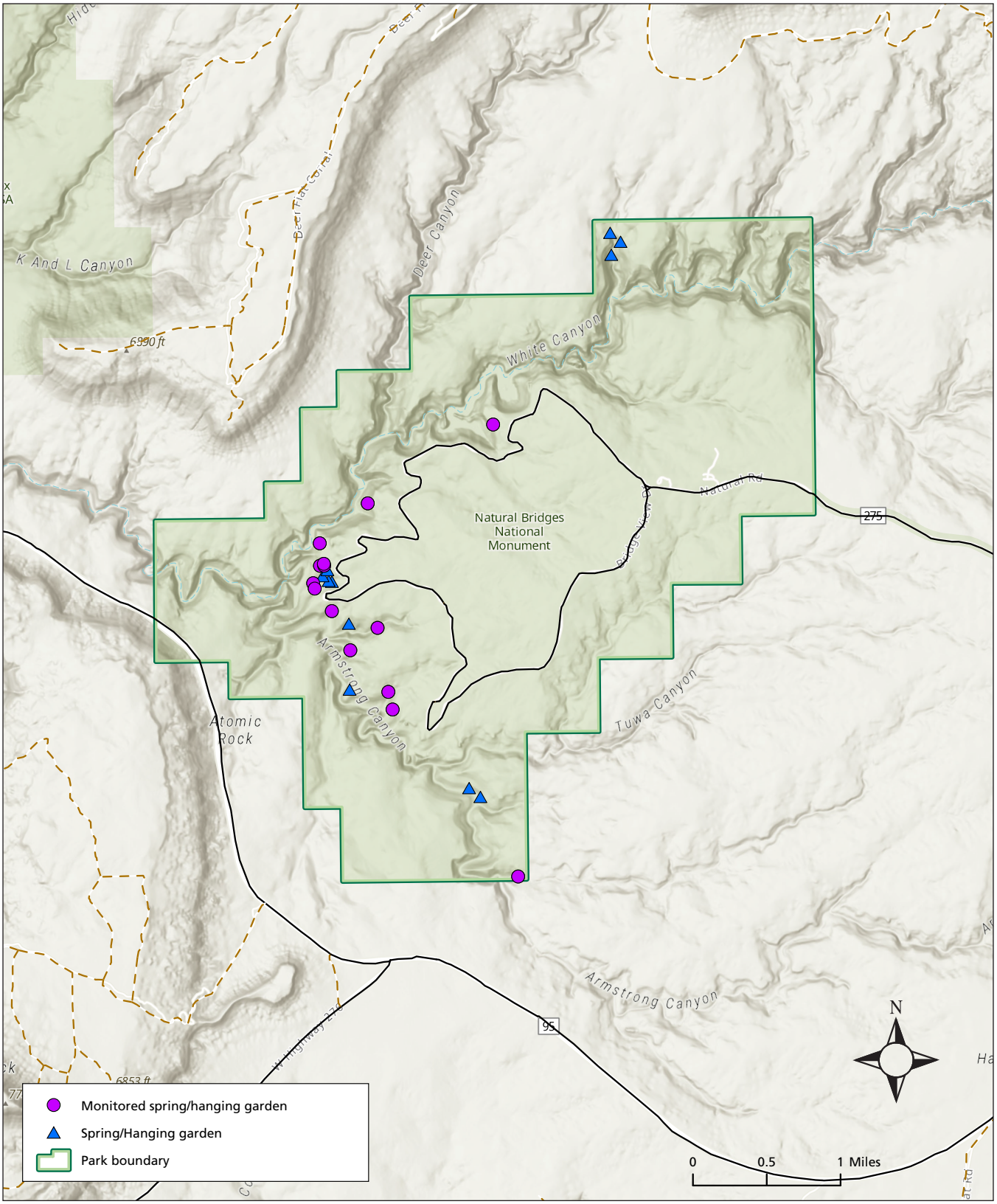


Figure 2-3. Hanging gardens at Natural Bridges National Monument, with 14 of 19 sites selected for monitoring.

of a bedrock bench near the elevation of the backwall, or the steepness of the colluvial slope. Photographs are taken and annotated to show the extent of the mapped hanging-garden area. The sketch map and photos are used to delineate the area that is sampled during long-term monitoring to enhance repeatability. A list of vegetation species is also completed for each site. A complete list of all species observed, along with the number of sites at which the species was found, is available in Appendix A.

The number of endemic plants by species is recorded in count classes for each hanging garden's backwall, alcove floor, and colluvial slope, and then totaled for the spring. Seven perennial-plant species were selected for pilot monitoring (Table 2-1). Because of their differing growth habits, including some rhizomatous and stoloniferous species, the unit of what constitutes an "individual" to be counted is also specified. In many cases, these are countable objects rather than strict genetic individuals.

Up to 500 individuals are counted for each species per geomorphic surface. Larger populations are estimated in count classes of 501–1,000, 1,001–10,000, and >10,000 individuals (Table 2-2). After completing the geomorphic surfaces, the total count for each species is estimated for the total spring area.

2.3 Double-observer sampling

Because of differing growth habits of the endemic species and the lack of a standardized search area, observer repeatability is a concern. We investigated repeatability through double-observer counts during the same site visit and by comparing counts from the initial and second-year site visits, usually by the same observer (Observer 1). The exceptions were Natural Bridges NM in 2018 and Arches NP in 2019, which were sampled by Observer 2. The ARCH sites were always sampled during May–June, and NABR sites were always sampled in September–October. Sites at Canyonlands NP were sampled in August–September. The selected endemic plants are all perennial. Sampling early or late in the growing season should not greatly affect observer variability.

Table 2-1. Endemic species, their counting unit, and their presence in Southeast Utah Group parks.

Family	Species	Counting unit	Growth habit	Presence*		
				ARCH	CANY	NABR
Asteraceae	*Cirsium rydbergii*	Individual	–	P	P	P
Asteraceae	*Erigeron kachinensis*	Rosette	Rhizomatous	NP	A	P
Liliaceae	*Zigadenus vaginatus*	Individual	–	P	P	P
Orchidaceae	*Habenaria zothecina*	Individual	–	P	P	P
Primulaceae	*Primula specuicola*	Rosette	Rhizomatous	P	P	U
Ranunculaceae	*Aquilegia micrantha*	Individual	–	P	P	P
Scrophulariaceae	*Mimulus eastwoodiae*	Stem	Stoloniferous	P	P	P

*P = Present in Park, U = Unconfirmed but suspected to occur in park, A = present in county, NP = not known to be present.

Table 2-2. Count classes for endemic species at hanging gardens in Southeast Utah Group parks.

Count class	Definition
N	Species not seen and presumed absent
1–500	Number of individuals counted
A	501–1,000 individuals
B	1,001–10,0000 individuals
C	>10,000 individuals

Double-observer sampling was not explicitly included in the springs sampling protocol, and was conducted as personnel were available. Four observers assisted with fieldwork over time; all had multiple years of experience monitoring vegetation. Observer 1 was the primary observer and conducted all fieldwork except when noted. Observer 1 had over 10 years of specialized experience working in hanging gardens on the Colorado Plateau. Observer 2 was the secondary observer and completed monitoring fieldwork when Observer 1 was unavailable. Observer 2 had 1–3 years of experience conducting vegetation-monitoring fieldwork in upland systems on the Colorado Plateau. Observer 3 had four years of experience conducting vegetation-monitoring fieldwork in upland systems on the Colorado Plateau, and Observer 4 had one year of experience conducting vegetation-monitoring fieldwork in upland systems on the Colorado Plateau and additional experience in other ecosystems.

Double-observer data were collected at three ARCH sites in May 2018, for a total of 12 site/species comparisons by observers 1 and 4. They were also collected at seven NABR sites and one CANY site in September 2016 by observers 2 and 3, for a total of 32 site/species comparisons.

2.4 Data summary and analysis

For data summary and analysis, we converted the total plant-count data to ordinal count classes (Table 2-3). Numeric data were combined into classes of 1–100 individuals and 101–500 individuals. The upper two field classes were collapsed into one ordinal count class of >1,000. We also investigated whether it was possible to analyze numeric data for populations with fewer than 500 individuals total. Because of the small sample size, data for Canyonlands NP are included in species-based analyses of repeatability and trend but were not analyzed at the park scale. Graphics and summaries were produced in R (R Core Team 2021) using the tidyverse (Wickham et al. 2019) and lubridate (Grolemund and Wickham 2011) packages.

Table 2-3. Ordinal count classes used for data summary and analysis.

Count class	Number of plants	Status
0	0	Not detected
1	1–100	Vulnerable
2	101–500	Low
3	501–1,000	Moderate
4	>1,000	High

Because of the lack of a standardized sampling area and the lack of replication within a site, approaches such as ordinal regression or Bayesian hierarchical modeling are not appropriate for analyzing the data. Instead, the metric of interest is the distribution of hanging-garden endemic-plant populations across the count classes over time. Because different hanging gardens are sampled each year over five years, the data must be pooled to compare earlier and later time periods.

We pooled the data into two samples of site/species pairs: the initial year a site was sampled (Yr0) and the most recent year the site was sampled (YrMax). The two ordinal samples were then tested for differences in count-class distributions using a two-sided dependent-samples sign test. This test collapses the comparisons to a plus, minus, or tie, and can only provide an analysis of change or no change; there is no estimate of the magnitude or rate of change.

Most of the sites had at least one plant population with fewer than 500 individuals. We were able to further test the numeric data using a paired Wilcoxon test for any species with 20 or more paired observations. We investigated both two-sided (increase or decrease) and one-sided (decrease) analyses to test for changes and then, more specifically, for declines in populations. Similar to the sign test, there is no estimate of the magnitude or rate of change. Analyses were performed in R 4.1.0 (R Core Team 2021).

2.5 Climate

Due to gaps in local weather-station data, climate summaries were calculated using historical gridMET data (Abatzoglou 2013) at the park centroid. Climate normals were calculated for the 30-year period from 1981 to 2010 by averaging temperature variables and summing precipitation and water defi-

cit. Gridded precipitation and temperature data were used to calculate water-balance variables accessed via the following URL: http://www.yellowstone.solutions/thredds/catalog.html (Tercek et al. 2021). Water deficit is the amount of water plants would use if it were available and is calculated as potential evapotranspiration minus actual evapotranspiration. Water deficit integrates water availability with ambient heat to provide a more meaningful metric of ecological drought than temperature or precipitation alone.

3 Results

3.1 Climate context

The three NCPN parks where hanging gardens are monitored stretch from Arches NP, in the north, to Natural Bridges NM in the south, with Canyonlands NP in the middle. Hanging-garden elevations range from 1,285 to 1,451 meters at ARCH, 1,509 to 1,728 meters at CANY, and 1,773 to 1,886 meters at NABR (Figure 3-1).

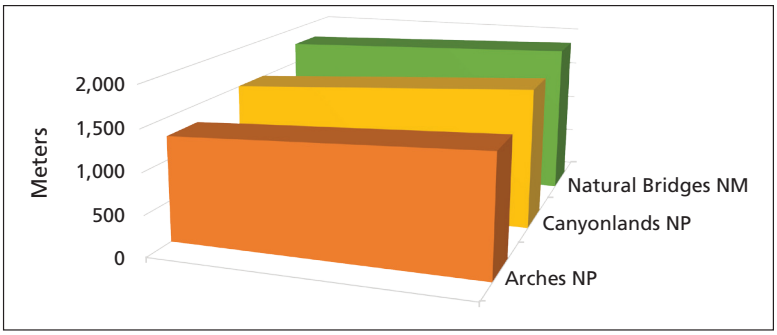


Figure 3-1. Elevation range of hanging gardens in NCPN parks where they are monitored.

For simplicity, we summarize climate variables at ARCH and NABR to represent the span of conditions across all three parks. Both parks showed similar patterns in climate. From 1979 to 2020, mean annual maximum and average temperatures had modest increases, but mean annual minimum temperatures increased at a quicker rate (Figure 3-2). Mean annual precipitation, although highly variable, trended downward over the same period, while mean annual water deficit increased steeply (Figure 3-3).

From 2013 to 2020, average annual temperatures were mostly higher than 30-year normals at both parks (Figure 3-4). There was a mix of years that were wetter and drier than normal, but the combination of higher temperatures and lower precipitation led to persistent drought conditions from 2016 to 2020.

3.2 Endemic plant count repeatability

3.2.1 Double-observer, same-year count classes

Of the 678 site/species pairs, 44 (6.5% of the dataset) had two observers present for sampling. Count classes matched for 32 of 44 pairs and were not in the same count class for 12 of 44 (25%; Table 3-1). Sample sizes for these comparisons were relatively low. The two observers counted *Cirsium rydbergii* (thistle) and *Erigeron kachinensis* (daisy) the most consistently, with all sites recorded in the same count class. *Habenaria zothecina* (orchid) and *Mimulus eastwoodiae* (monkeyflower) were recorded in different classes almost half the time. *Aquilegia micrantha* (columbine), *Primula specuicola* (primrose), and *Zigadenus vaginatus* (death camas) were classified differently 20–33% of the time.

3.2.2 Count classes from Yr0 to Yr1 (first sampling event to following year's sampling event)

To increase the number of comparisons available, we also compared data from two back-to-back sampling years. Because these species are perennial, large fluctuations in populations from year to year are not expected and have not been observed. However, this approach does risk including some actual change mixed in with observer variability.

In year-to-year sampling, the same count class was recorded for 166 of 219 observations (75.7%; Table 3-2). A decrease in one class was recorded for 27/219 (12.3%), a decrease in two count classes in 1/219 (0.5%), and an increase of one count class in 25/219 (11.4%). Of the species counted, *Erigeron kachinensis* (daisy) was the least repeatable across years within these count classes, with 60% of visits classified differently. The remaining species were classified differently on 4–35% of visits. The observer was Observer 1, except for at Natural Bridges NM in 2018 and Arches NP in 2019 (Observer 2).

Only *Cirsium rydbergii* (thistle) had high repeatability in both double-observer and year-to-year count scenarios.

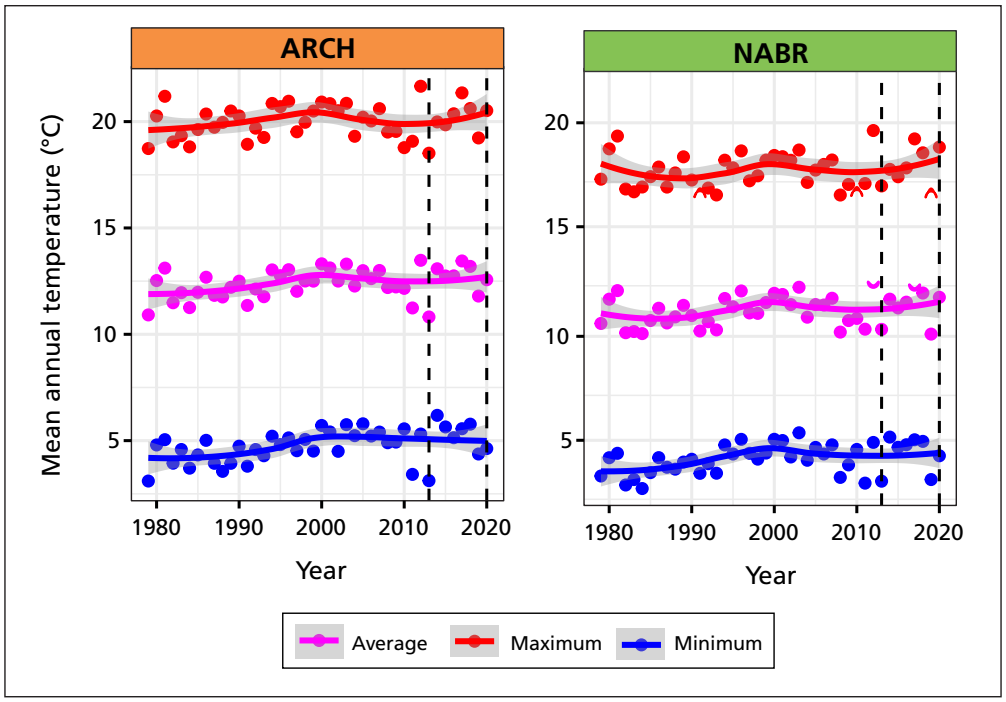


Figure 3-2. Mean annual average, maximum, and minimum air temperature data with LOESS smooths from historic gridMET model at the park centroid for Arches National Park and Natural Bridges National Monument, 1979–2020. The period of record for this report, from 2013 to 2020, is marked by dashed vertical lines on the graphs.

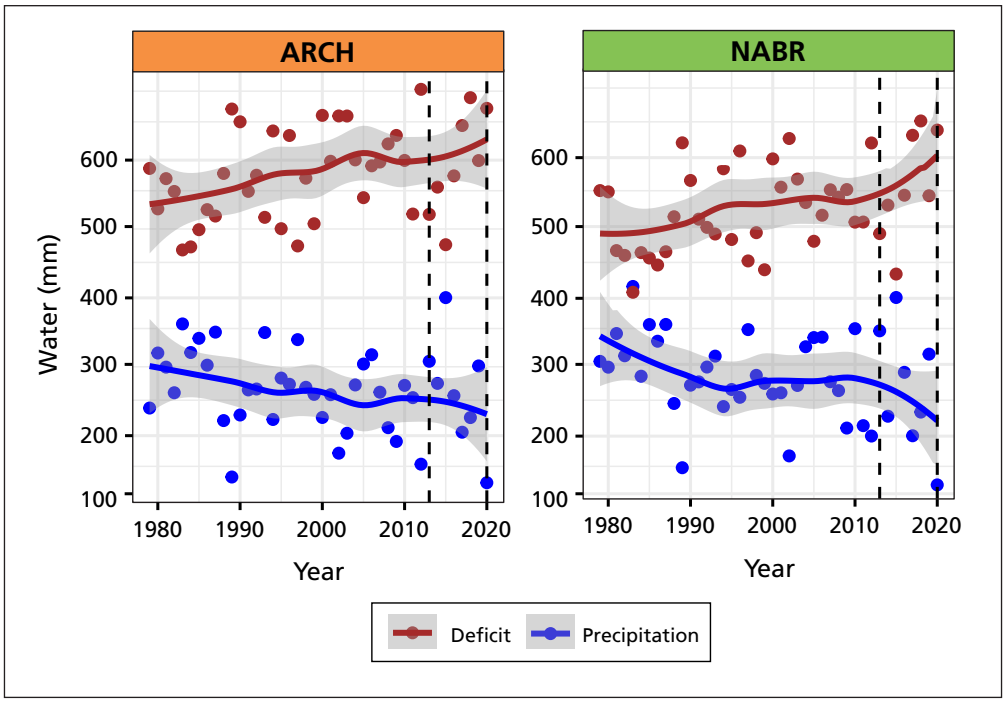


Figure 3-3. Mean annual cumulative precipitation and cumulative water-deficit data with LOESS smooths from historic gridMET model at the park centroid for Arches National Park and Natural Bridges National Monument, 1979–2020. The period of record for this report, from 2013 to 2020, is marked by dashed vertical lines on the graphs.

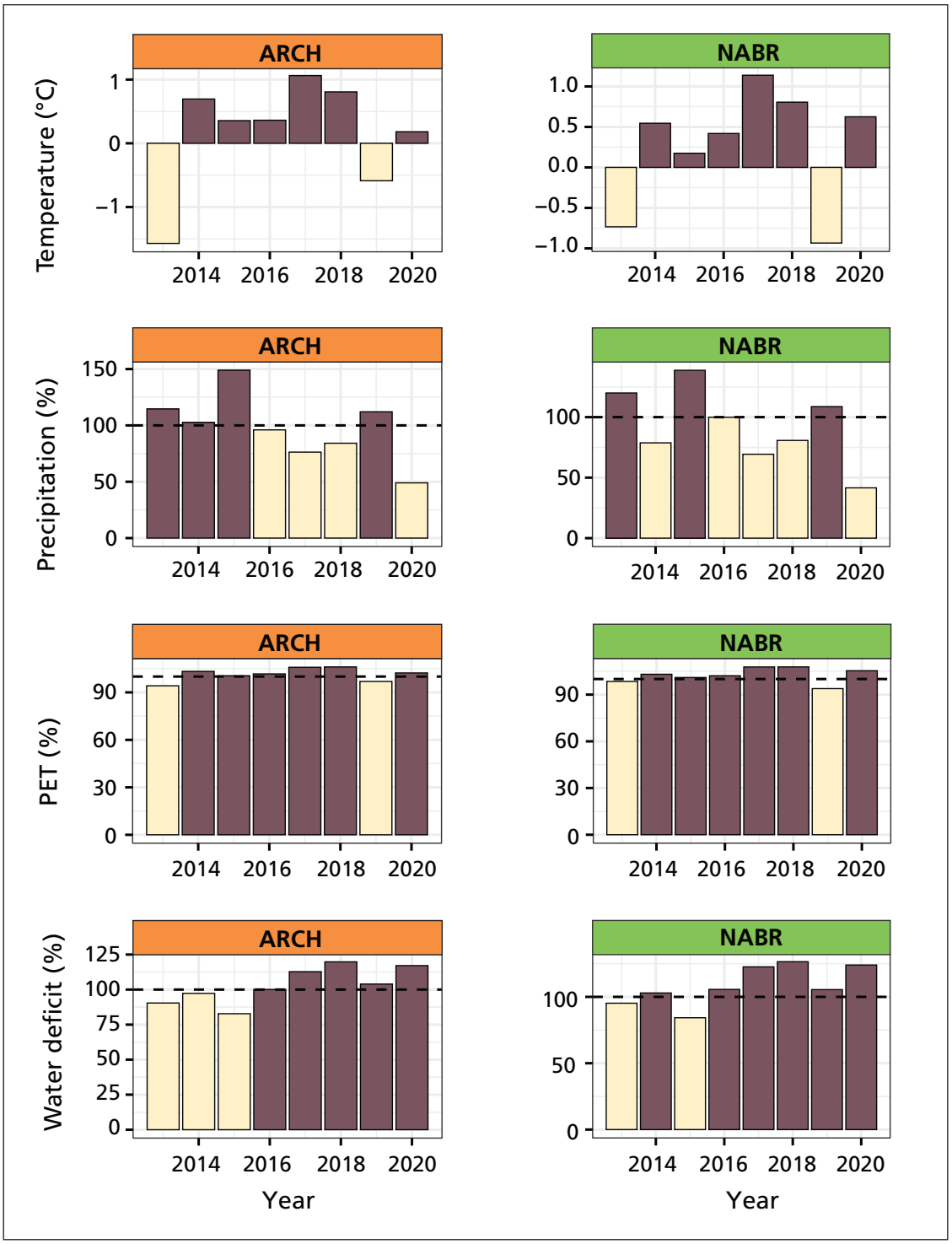


Figure 3-4. Departures from 1981–2010 climate normals for (A) average temperature, (B) precipitation, (C) potential evapotranspiration (PET), and (D) water deficit at Arches National Park (ARCH) and Natural Bridges National Monument (NABR), 2013–2020.

Table 3-1. Similarities and differences in count classification by two independent observers during the same visit to Southeast Utah Group parks, 2013–2020.

Species	Total number	Number		Percentage	
		Same	Different	Same	Different
Cirsium rydbergii	5	5	0	100.0	0.0
Erigeron kachinensis	5	5	0	100.0	0.0
Aquilegia micrantha	11	8	3	72.7	27.3
Primula specuicola	7	5	2	71.4	28.6
Zigadenus vaginatus	3	2	1	66.7	33.3
Habenaria zothecina	5	3	2	60.0	40
Mimulus eastwoodiae	8	4	4	50.0	50.0
Total	44	32	12	72.7	27.3

Table 3-2. Differences in year-to-year classification, 2013–2020.

Species	Total number	Number		Percentage	
		Same	Different	Same	Different
Cirsium rydbergii	25	24	1	96.0	4.0
Aquilegia micrantha	41	34	7	82.9	17.1
Mimulus eastwoodiae	50	41	9	82.0	18.0
Zigadenus vaginatus	35	25	10	71.4	28.6
Habenaria zothecina	27	18	9	66.7	33.3
Primula specuicola	31	20	11	64.5	35.5
Erigeron kachinensis	10	4	6	40.0	60.0
Total	219	166	53	75.8	24.2

3.2.3 Double-observer for populations of <500 individuals (numeric data)

In addition to data in count classes, we were interested in the repeatability of counts of up to 500 individuals. The number of double-observer comparisons was limited to 1–9 per species, but the data suggested some patterns. When comparing counts from independent observers during the same visit, *Cirsium rydbergii* (thistle) was still the most consistently counted (n=4). This was not surprising, because it is a large thistle with a growth habit that is relatively easily to distinguish. For the four species with more than one set of double-observer data in this population class, counts by the observer with less experience working in hanging gardens were consistently lower than those by the observer with more experience, and the difference between the two often did not overlap with zero (Figure 3-5). This suggests that the observer is probably an important variable to consider in results, although we don't have enough double-observer data to quantitatively estimate the effect.

3.2.4 Repeat visits from first year to second year for populations of <500 individuals (numeric data)

For visits from the first year of sampling to the second year of sampling, mostly with the same observer over time, medians approached zero for all species except *Erigeron kachinensis* (daisy). *Aquilegia micrantha* (columbine), *Cirsium rydbergii* (thistle), and *Habenaria zothecina* (orchid) had smaller interquartile ranges, while *Primula specuicola* (primrose) and *Zigadenus vaginatus* (death camas) had much wider interquartile ranges (Figure 3-6). This suggests that observer consistency helps improve repeat-

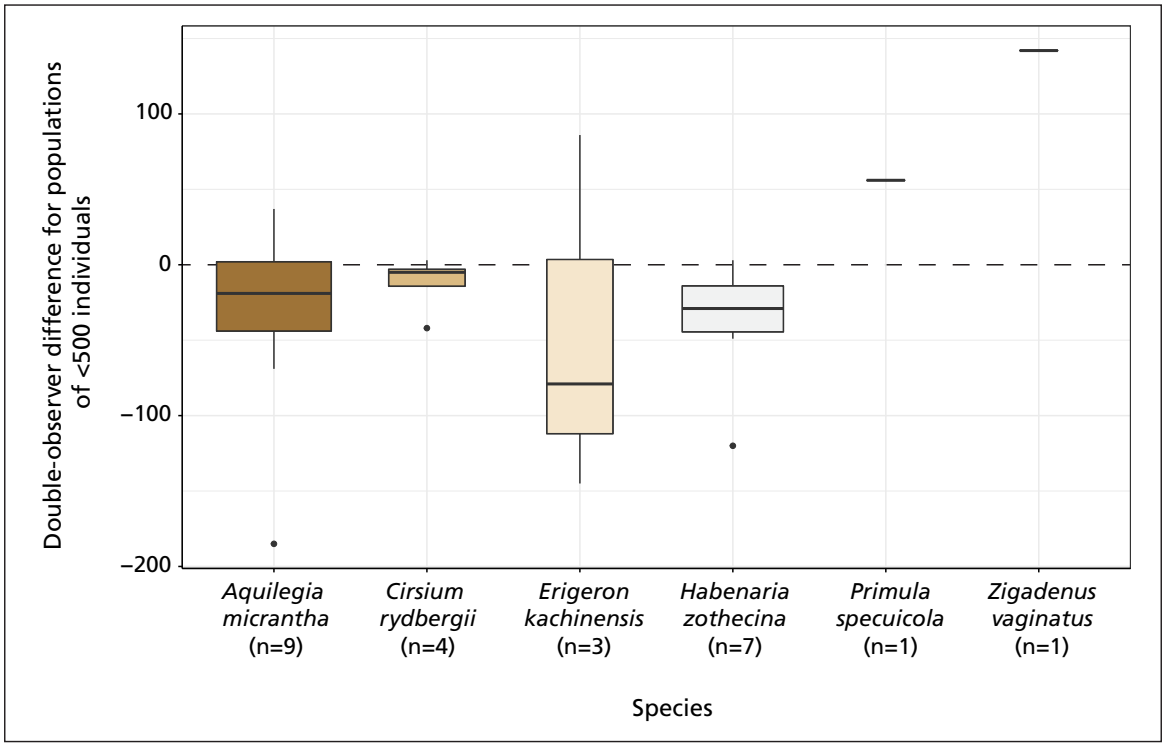


Figure 3-5. Differences in independent double-observer counts during the same site visit for populations with up to 500 individuals. *Mimulus eastwoodiae* (monkeyflower) tended to either be absent or to occur only in the highest count classes at both parks. The boxplot shows the median value as an interior line bounded by a box showing the interquartile range of the data. Whiskers represent the range of data within 1.5 × IQR. Dots lie outside this range.

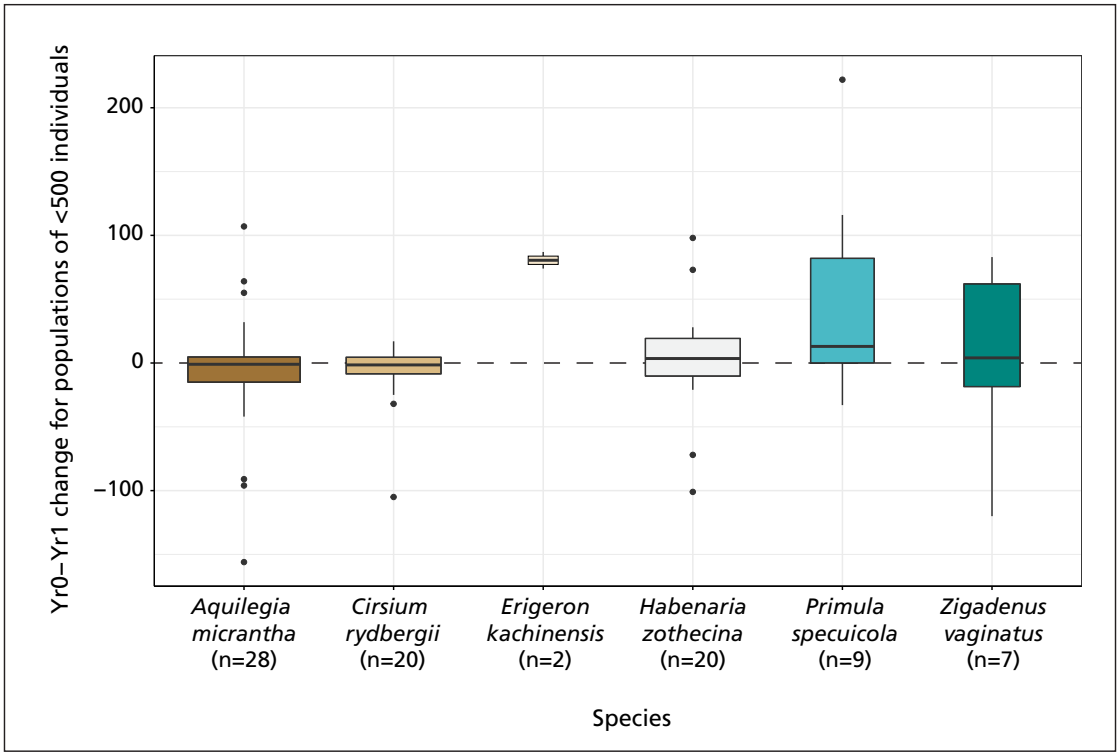


Figure 3-6. Differences in counts from repeat visits over two years for populations with up to 500 individuals. *Mimulus eastwoodiae* (monkeyflower) tended to either be absent or to occur only in the highest count classes at both parks. The boxplot shows the median value as an interior line bounded by a box showing the interquartile range of the data. Whiskers represent the range of data within 1.5 × IQR. Dots lie outside this range.

ability in counts compared to the independent observer comparisons above. A slight increase in counts could be expected as the observer gains familiarity with the site.

3.3 Characterization of species and sites in our datasets

To show changes in the distribution of population sizes over time, it's important to have sites with populations in various count classes at the beginning of the period of record. In the first year of sampling (Yr0), populations were distributed throughout the count classes for most species at Arches NP and several species at Natural Bridges NM (Figures 3-7 and 3-8). (*Erigeron kachinensis* (daisy) is absent at ARCH and *Primula specuicola* (primrose) is absent at NABR.) *Cirsium rydbergii* (thistle) only occurs at two locations in NABR. *Mimulus eastwoodiae* (monkeyflower) tended to either be absent or occur only in the highest count classes at both parks. This suggests that only new colonizations or very large reductions in monkeyflower would be captured by our methods. Sample sizes were too low to test count-class distributions at Canyonlands NP.

3.3.1 Number of species per site

When several endemic-plant populations are moving in the same direction over time, it bolsters our confidence in trends. At the beginning of sampling (Yr0), almost all sites at all three parks had more than one endemic species present at each site (Figure 3-9). At all parks, only six of the seven species were potentially present at each site.

Because of high variability in repeat counts and low sensitivity to count-class bins, counts of *Erigeron kachinensis* (daisy) and *Mimulus eastwoodiae* (monkeyflower) had relatively poor potential to detect trends over time. We investigated whether sites would still have several species present if counts of those two species were removed from the monitoring protocol. This would leave five potential species at ARCH and CANY and four potential species at NABR. Even in this scenario, almost all sites had more than one endemic species present. Removing these two species would not inhibit the detection of population trends at most sites.

3.4 Trends in endemic-plant counts over time

3.4.1 Park-based trends

To analyze park-based trends in plant counts, data were pooled into two time periods: the first year of sampling (Yr0) and the most recent year of sampling (YrMax). The rotating-panel design of temporal visits to sites means that the full panel of sites is visited over the course of four years; only a subset of sites is visited during any single year. For this reason, the earliest year of sampling (Yr0) is spread out over several years at each park. Yr0 at Arches NP ranged from 2014 to 2018, and Yr0 at Natural Bridges NM ranged from 2013 to 2017. For this report, YrMax ranges from 2017 to 2020 at both parks. Note that at ARCH, there is some overlap in Yr0 and YrMax. Sample sizes were too low for statistical analysis at Canyonlands NP.

Of the trends in endemic-plant count classes at Arches NP or Natural Bridges NM for our period of record (Figure 3-10, Tables 3-3 and 3-4), none were statistically significant. For 126 site/species pairs at ARCH, the dependent-samples sign-test value was $S = 23$, $p = 0.53$. For 39 site/species pairs at NABR, the dependent-samples sign-test value was $S = 8$, $p = 0.58$. There were also no statistically significant trends in small populations (<500 individuals) using a paired Wilcoxon test (ARCH: n=91 site/species pairs, V=2,161, p=0.40; NABR: n=27, V=194, p=0.21).

3.4.2 Species-based trends

Although aquifers at different parks are independent of each other, climate change is a regional stressor that is similar across the three parks in our study. To look at species-based trends in plant counts, we pooled data from all three parks and focused on any change in small populations (<500 individuals) using a two-sided (increase or decrease) test and then, more specifically, on declines using a one-sided (decrease) test. We tested data for any species with more than 20 paired observations, and we included any site/species pair where either Yr0 or YrMax

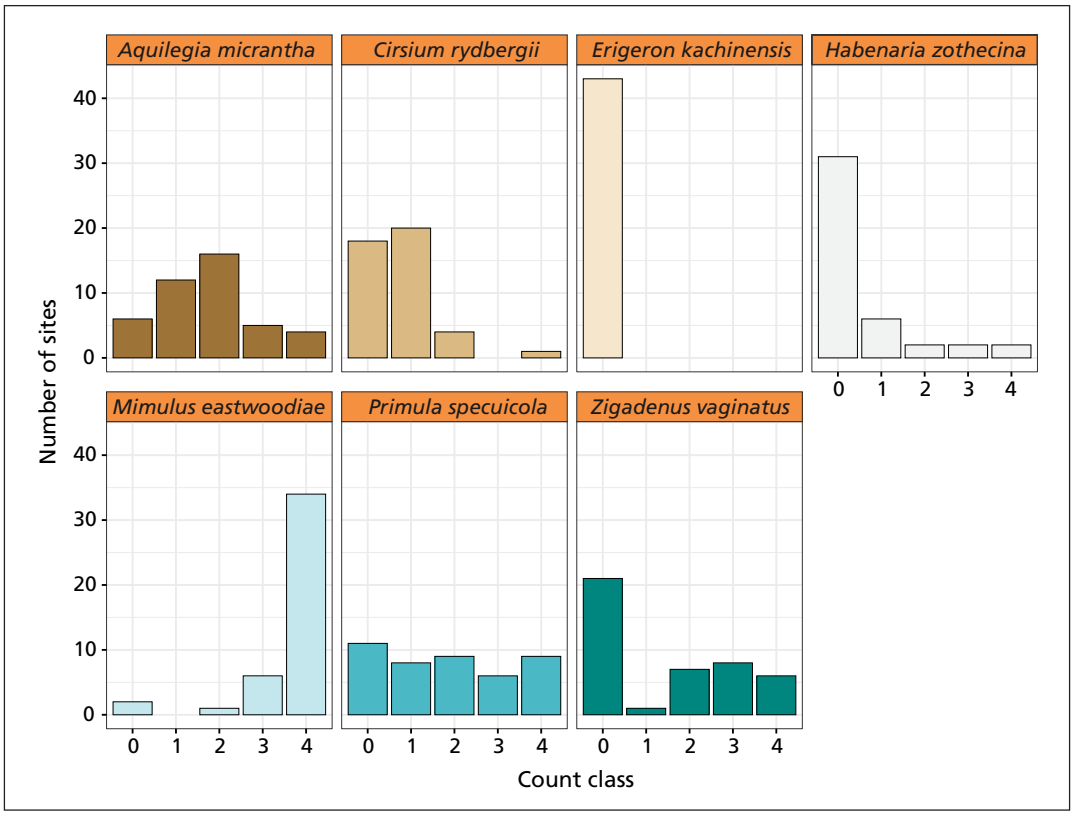


Figure 3-7. Distribution of species in count classes in Yr0 at different sites in Arches National Park. Classes are 0 (not observed), 1 (1–100 individuals), 2 (101–500), 3 (501–1,000) and 4 (>1,000).

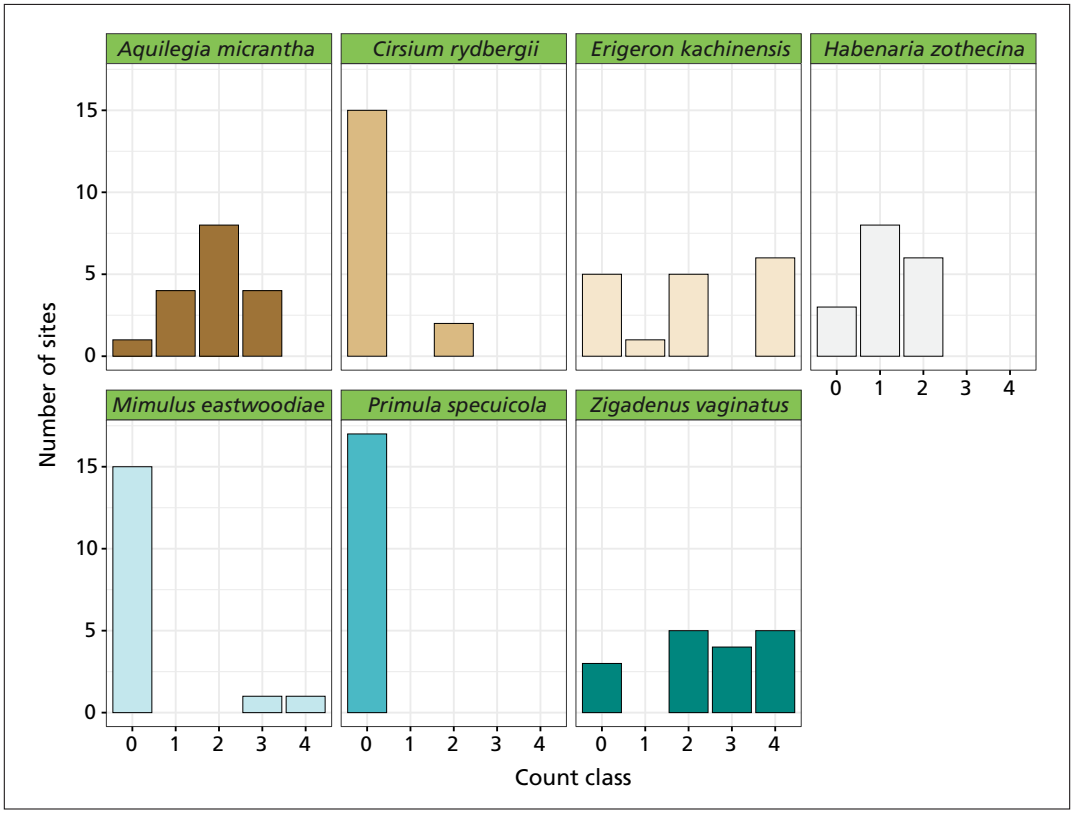


Figure 3-8. Distribution of species in count classes in Yr0 at different sites in Natural Bridges National Monument. Classes are 0 (not observed), 1 (1–100 individuals), 2 (101–500), 3 (501–1,000) and 4 (>1,000).

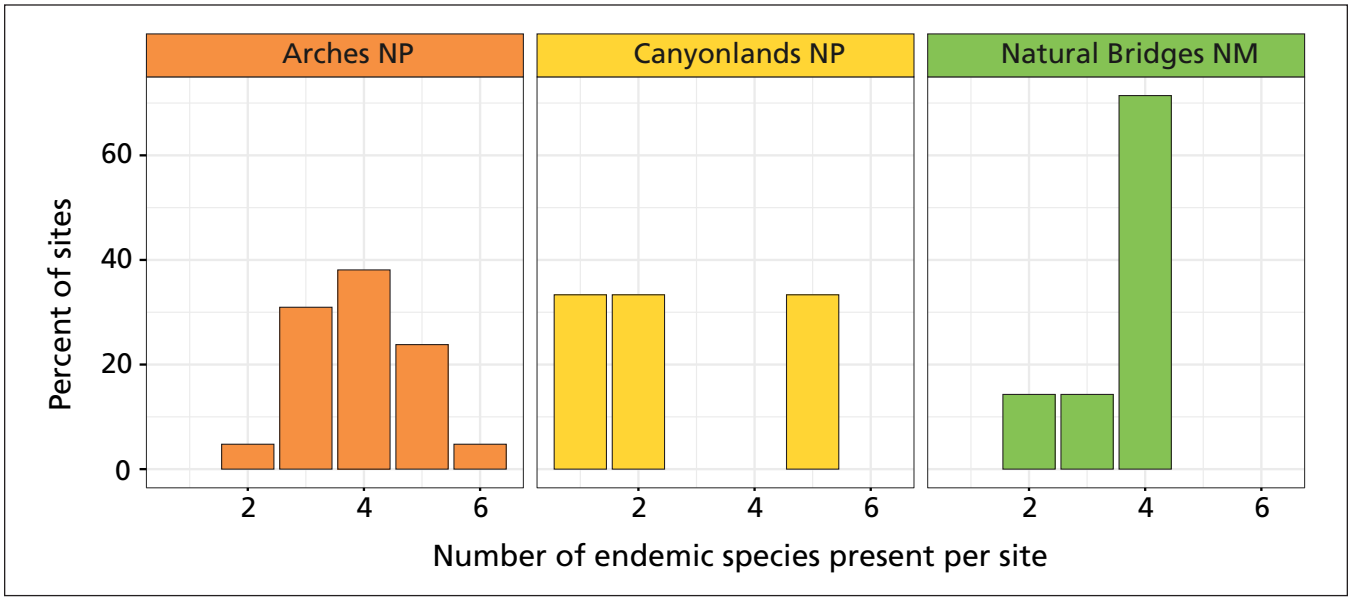


Figure 3-9. The percentage of sites at each park with 1–6 endemic species present. Each park has only six of the seven species present. Most sites have more than one endemic species present.

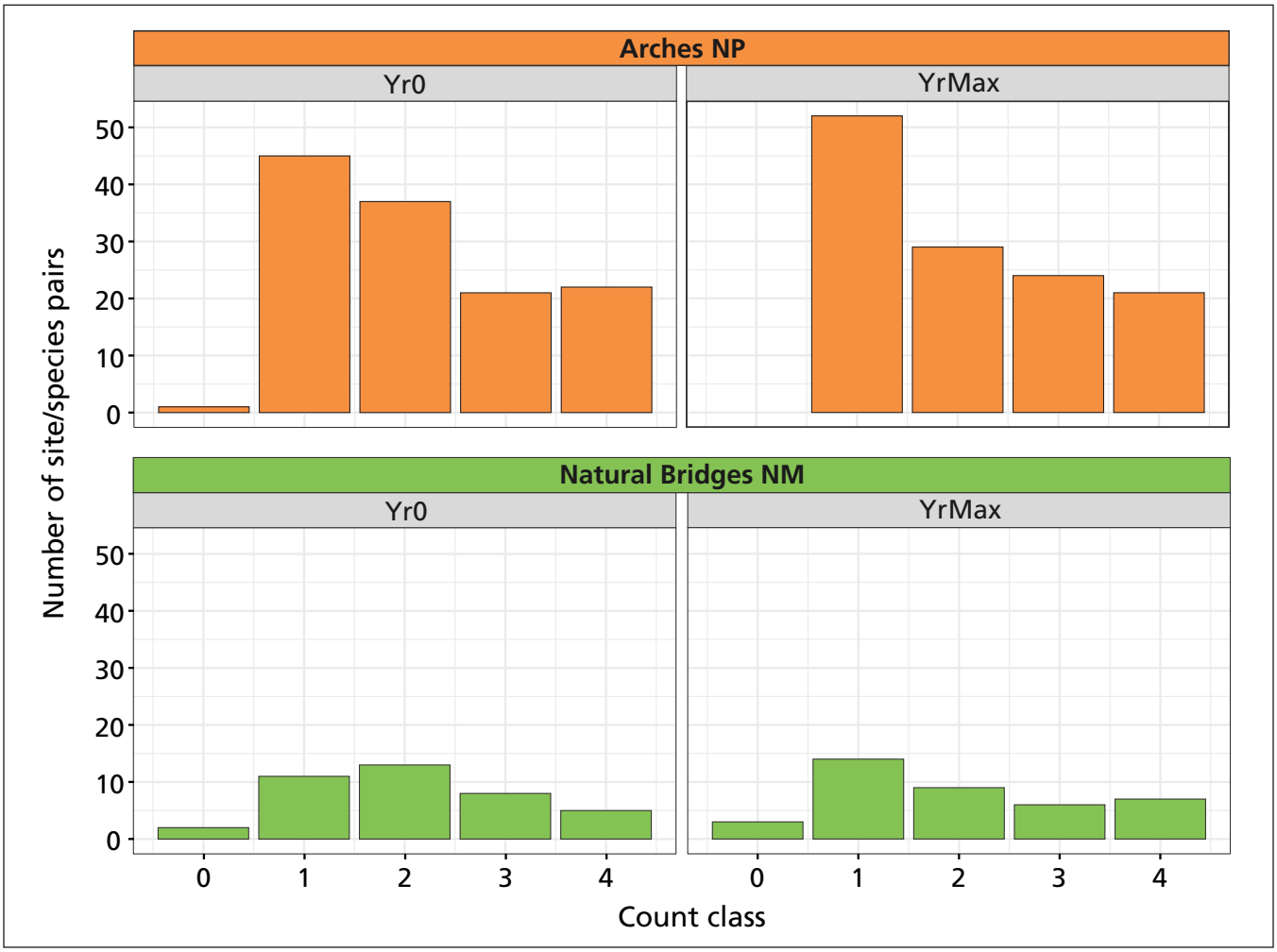


Figure 3-10. Distribution of site/species populations in count classes for the first year of sampling (Yr0) and the most recent year of sampling (YrMax) for each site.

were recorded as <500 individuals, or where both periods had <500 individuals. Only three species had enough observations to be included. One species, *Cirsium rydbergii* (thistle), had a change large enough to be detected with both a two-sided and one-sided test. When testing specifically for decreases in small populations, all three species (columbine, thistle, and orchid) had statistically significant declines (Table 3-5).

3.4.3 *Number of vulnerable sites*

Sites with low population numbers are of particular concern to park managers. At Arches NP and Natural Bridges NM, 51 of 57 sites have at least one endemic-plant population of fewer than 500 individuals, and 42 of 57 sites have at least one population with fewer than 100 individuals. Based on visits in the early 2000s that failed to relocate some populations noted in the 1990s (Weissinger et al. 2011), populations with fewer than 100 individuals may be more vulnerable to extirpation. When comparing the first and last year of sampling, almost every species at both parks had a higher number of vulnerable populations at the end of the period of record (Table 3-6). Most *Cirsium rydbergii* (thistle) populations and over half of *Habenaria zothecina* (orchid) populations had fewer than 100 individuals. In 2020, several populations of *Habenaria zothecina* at NABR monitoring sites were not found where they had previously been recorded. A combination of low population sizes and sensitivity to drought indicate that *Habenaria zothecina* and *Cirsium rydbergii* may be vulnerable to population reductions.

Table 3-3. Difference in count classes between Yr0 and YrMax for 126 site/species pairs at Arches National Park, 2013–2020.

Difference in count class	Observations	Percent of comparisons
-1	23	18%
0	85	67%
1	17	13%
2	1	1%

Table 3-4. Difference in count classes between Yr0 and YrMax for 39 site/species pairs at Natural Bridges National Monument, 2013–2020.

Difference in count class	Observations	Percent of comparisons
-1	8	21%
0	26	67%
1	5	13%

Table 3-5. Paired Wilcoxon test results for three hanging-garden endemic species at Southeast Utah Group parks, 2013–2020.

Species	Observations	Test statistic	Two-sided p-value (any change)	One-sided p-value (decrease)
Aquilegia micrantha	42	506	0.11	0.05*
Cirsium rydbergii	25	221.5	0.04*	0.02*
Habenaria zothecina	23	176.5	0.11	0.05*

*Statistically significant p-value.

Table 3-6. Number of sites with 1–100 individuals in first and most recent sampling events.

Park	Species	Total sites	Number (%) of sites with 1–100 individuals		Percent change
			Yr0	YrMax	
ARCH	*Aquilegia micrantha*	37	11 (30)	13 (35)	+5
ARCH	*Cirsium rydbergii*	24	19 (79)	20 (83)	+4
ARCH	*Habenaria zothecina*	13	6 (46)	8 (62)	+16
ARCH	*Primula specuicola*	31	8 (26)	8 (26)	0
ARCH	*Zigadenus vaginatus*	23	1 (4)	3 (13)	+11
CANY	*Aquilegia micrantha*	2	1 (50)	1 (50)	0
CANY	*Cirsium rydbergii*	1	0 (0)	0 (0)	0
CANY	*Habenaria zothecina*	2	1 (50)	1 (50)	0
CANY	*Zigadenus vaginatus*	2	0 (0)	1 (50)	+50
NABR	*Aquilegia micrantha*	13	4 (31)	5 (38)	+7
NABR	*Cirsium rydbergii*	1	0 (0)	1 (100)	+100
NABR	*Habenaria zothecina*	13	7 (54)	8 (62)	+8
NABR	*Zigadenus vaginatus*	12	0 (0)	0 (0)	0

4 Discussion

4.1 Count repeatability

Inter- and intra-observer variability is a well-known issue with vegetation-monitoring data, but there is no standard agreement on what level of variability is acceptable. In a recent review, Morrison (2016) reported that over 90% of 59 vegetation studies documenting observer variability found a statistically significant effect on species detection, identification, or estimates of prevalence (e.g., cover or abundance). NCPN's monitoring protocol attempts to limit variation in species detection and identification by (1) limiting the number of species to accurately identify at a site, (2) providing training material on species identification, (3) maintaining a consistent observer over time, and (4) providing lists of species previously observed at each site as a working list for counts.

The range of different classifications in count classes reported here is similar to what has been reported in other studies, although no other studies are known for this vegetation type. Cover estimates differed by one category for 33–48% of plots summarized in a review by Morrison (2016), who stated, "In studies of vegetation abundance utilizing category estimation, between one-third and one-half of all estimates are erroneous, but most of these are probably off by only one category." For example, Morrison and Young (2016) found that estimates of rare-plant densities in limestone glades were one category lower than true counts for about 30% of 365 accuracy assessments over 10 years. They also noted high observer variability and consistent bias for some observers.

In contrast, differences in assigned cover classes ranged from only 2% to 6% for sagebrush monitoring at Bighorn Canyon National Recreation Area but 2–21% at John Day Fossil Beds NM (Esposito et al. 2016). Vittoz and Guisan (2007) found that in larger (40 m^2) alpine meadow plots of, visual estimates were useful for detecting large changes, and estimates were improved by using a consistent observer over time. High variability is a common component of vegetation quantification.

There is no formal way to account for observer variability in the statistical tests used for this analysis. Aside from independent observations, most data were collected by a single observer, Observer 1. From year to year, Observer 1 increased and decreased classifications roughly equally, suggesting that intra-observer variability adds noise but not necessarily bias to the dataset. With a consistent observer, statistical trends are likely to lag behind actual changes in plant populations. In contrast, independent-observer comparisons suggest that greater experience working in hanging gardens leads to higher counts as observers gain familiarity with the sites and species. When there is a change in personnel, increasing trends in species counts may be a result of increasing expertise rather than a true change in populations.

4.2 Trends over time

When data were pooled for the region, several species showed declines in plant counts from 2013 to 2020 at sites with small populations. *Cirsium rydbergii* (thistle) had statistically significant changes using both a two-sided (increase or decrease) and one-sided (decrease) test (see Table 3-5). Both *Habenaria zothecina* (orchid) and *Aquilegia micrantha* (columbine) also showed statistically significant decreases. Because of small sample sizes, these are currently the only three species with enough data for species-specific trend analysis.

Our data did not indicate any statistical trends over time in endemic plant-count categories or across all species for small populations at hanging gardens in Arches NP and Natural Bridges NM. This is unsurprising, given that the period of record for this report only barely exceeds the five-year panes for the full panel of springs monitoring. As the amount of time lengthens, broad-scale changes are more likely to become apparent. It is also likely that the very wide bins in higher count classes reduced the sensitivity of changes in count-class distributions. We have already noted an increasing shift toward populations of fewer than 100 individuals.

Starting in 2020, we also noted potential die-offs of some endemic species at some sites, particularly *Habenaria zothecina* (orchid) at Natural Bridges NM. For example, at one site, no orchids were found in 2020, whereas previous counts had ranged from 48 to 183 individuals. This is likely due to increasing drought, as air temperatures continue to rise in the region. Drought effects were visible on other springs vegetation during site visits to SEUG parks in 2020 and 2021. Woody spring-associated species, such as hackberry and chokecherry, have experienced canopy-browning and limb-shedding, and some individual woody plants recorded in photo-points have died (Figure 4-1).

We have also noted mortality in spring-associated species at the edges of hanging-garden habitat (Figure 4-2). As climate change continues, we anticipate additional drought stress and drying in hanging gardens and other springs in southeast Utah, which will likely negatively affect endemic-plant populations at these sites. Continued monitoring can track these changes and highlight sites that appear less affected by climate change.

4.3 Recommended changes to the monitoring program

Endemic-plant count data are variable, and some populations are large enough that our methods will be slow to detect changes. However, NCPN has found no other viable metric for long-term monitoring at most hanging gardens. In addition, endemic-plant counts are more sensitive as populations get smaller, so counts can give relevant information to park managers more quickly as populations reach lower thresholds.

Based on this initial evaluation of the data, some modifications to the protocol seem beneficial at this time. We recommend:

1. Stop counting *Erigeron kachinensis* and *Mimulus eastwoodiae* in hanging gardens. The presence of these species is still captured in the vegetation species list recorded for each site during site characterization.
2. Continue to count individuals from 1 to 500, but estimate populations of 500–1,000 and >1,000 (remove the >10,000 count class). Break counts into classes of 1–100 and 101–500 for analysis, with special attention to populations of 1–100 as "vulnerable" populations.
3. Perform counts with long-term staff to minimize the effects of observer turnover. Abrupt changes in counts that coincide with a change in observer should be considered carefully. When possible, have the previous observer and new observer overlap with double-observer data for some sites.
4. Add analysis of small populations and use one-sided tests to test for decreases in populations, given that decreases are of the greatest management concern.



Figure 4-1. Examples of drought stress on woody species at Southeast Utah Group springs in 2020.



Figure 4-2. Die-off of spring-associated grasses at the downgradient extent of a hanging garden at Natural Bridges National Monument from 2017 to 2019.

The rotating panel of sampling sites for two years, followed by three years of rest, adds further complication to data analysis. It reduces the potential to detect trends quickly, because the analysis window moves in five-year panes. The monitoring program could opt to abandon the rotating panel and only sample a set number of springs every year. For logistical reasons, this would require reducing the total number of springs sampled to approximately 20 at Arches NP and 8 at Natural Bridges NM. In that scenario, data could be analyzed using a generalized linear model with a Poisson distribution and an offset for the size of the spring. However, with such low sample sizes, the results of such an analysis would be questionable.

On the other hand, sampling some sites each year makes it easier to understand when a significant change event may have occurred and can provide early detection of major shifts. There are also logistical benefits for long-term staffing to sampling fewer sites per year and having a more consistent effort each year. Most sites have at least one endemic-plant population of fewer than 500 individuals; prioritizing sites with small populations is not an effective criterion for reducing the number of springs that are sampled. Maintaining the current number of sites will allow more robust analysis of small populations.

While endemic-plant count data may not be precise, this information is still important, given the ecological value of these unique systems. Hanging-garden endemic-plant counts can also be useful in synthesis with the quantitative springflow monitoring we are able to conduct at a smaller set of springs.

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Appendix A. Hanging-Garden Species List

Table A-1. Number of sites at each park with species recorded during initial site characterization.

Family	Species[1]	Lifeform	Duration	Nativity	Wetland code[2]	Park[3]		
						ARCH (n=42)	CANY (n=3)	NABR (n=14)
Aceraceae	*Acer negundo*	Tree	Perennial	Native	FACW	4	–	–
Agavaceae	*Yucca angustissima*	DwarfShrub	Perennial	Native	UPL	–	1	5
Agavaceae	*Yucca harrimaniae*	DwarfShrub	Perennial	Native	UPL	1	–	–
Anacardiaceae	*Rhus aromatica*	Shrub	Perennial	Native	FACW	22	2	5
Anacardiaceae	*Toxicodendron rydbergii*	Shrub	Perennial	Native	FACU	20	2	–
Apocynaceae	*Apocynum cannabinum*	Forb	Perennial	Native	FAC	3	2	–
Asclepiadaceae	*Asclepias* sp.	Forb	Perennial	Native	–	1	–	–
Asclepiadaceae	*Asclepias speciosa*	Forb	Perennial	Native	FACW	–	1	–
Berberidaceae	*Berberis fendleri*	Shrub	Perennial	Native	UPL	–	1	–
Berberidaceae	*Mahonia fremontii*	Shrub	Perennial	Native	UPL	8	2	2
Berberidaceae	*Mahonia repens*	DwarfShrub	Perennial	Native	UPL	1	–	–
Betulaceae	*Betula occidentalis*	Tree	Perennial	Native	FACW	5	–	–
Betulaceae	*Ostrya knowltonii*	Tree	Perennial	Native	FACW	–	1	–
Boraginaceae	*Cryptantha* sp.	Forb	–	Native	UPL	2	–	–
Caprifoliaceae	*Sambucus caerulea*	Shrub	Perennial	Native	FACU	1	–	–
Caprifoliaceae	*Symphoricarpos longiflorus*	Shrub	Perennial	Native	FAC	1	–	–
Caprifoliaceae	*Symphoricarpos oreophilus*	Shrub	Perennial	Native	FACU	–	–	1
Caryophyllaceae	*Silene antirrhina*	Forb	Annual	Native	UPL	1	–	–
Celastraceae	*Pachystima myrsinites*	DwarfShrub	Perennial	Native	UPL	–	–	3
Compositae	*Ambrosia acanthicarpa*	Forb	Annual	Native	FAC	1	–	–
Compositae	*Artemisia bigelovii*	Shrub	Perennial	Native	UPL	1	–	1
Compositae	*Artemisia ludoviciana*	Forb	Perennial	Native	FACU	13	2	–
Compositae	*Aster ascendens*	Forb	Perennial	Native	FAC	1	–	2
Compositae	*Brickellia californica*	Shrub	Perennial	Native	UPL	1	–	–
Compositae	*Brickellia longifolia*	Shrub	Perennial	Native	FACW	3	2	–
Compositae	*Brickellia microphylla*	DwarfShrub	Perennial	Native	UPL	8	1	4
Compositae	*Chrysopsis villosa*	Forb	Perennial	Native	UPL	24	–	12
Compositae	*Chrysothamnus nauseosus*	Shrub	Perennial	Native	FAC	1	1	1

Table A-1, cont. Number of sites at each park with species recorded during initial site characterization.

Family	Species[1]	Lifeform	Duration	Nativity	Wetland code[2]	Park[3]		
						ARCH (n=42)	CANY (n=3)	NABR (n=14)
Compositae	*Cirsium calcareum*	Forb	Perennial	Native	FAC	2	1	11
Compositae	*Cirsium rydbergii*	Forb	Perennial	Native	OBL	24	–	1
Compositae	*Conyza canadensis*	Forb	Annual	Native	FACW	–	2	1
Compositae	*Erigeron kachinensis*	Forb	Perennial	Native	FACW	–	–	10
Compositae	*Erigeron pumilus*	Forb	Perennial	Native	UPL	14	–	–
Compositae	*Erigeron* sp.	Forb	–	Native	_	2	–	–
Compositae	*Erigeron utahensis*	Forb	Perennial	Native	UPL	1	–	–
Compositae	*Euthamia occidentalis*	Forb	Perennial	Native	OBL	1	–	–
Compositae	*Gutierrezia sarothrae*	DwarfShrub	Perennial	Native	UPL	5	1	6
Compositae	*Helianthella microcephala*	Forb	Perennial	Native	UPL	–	–	12
Compositae	*Hymenopappus filifolius*	Forb	Perennial	Native	UPL	–	–	4
Compositae	*Hymenoxys acaulis*	Forb	Perennial	Native	UPL	–	–	1
Compositae	*Machaeranthera canescens*	Forb	Perennial	Native	UPL	1	–	–
Compositae	*Oxytenia acerosa*	Forb	Perennial	Native	FACU	–	1	–
Compositae	*Senecio multilobatus*	Forb	Perennial	Native	UPL	2	–	2
Compositae	*Solidago missouriensis*	Forb	Perennial	Native	FACW	8	–	–
Compositae	*Solidago velutina*	Forb	Perennial	Native	FACU	23	2	6
Compositae	*Sonchus asper*	Forb	Annual	NonNative	FACU	5	–	–
Compositae	*Stephanomeria* sp.	Forb	_	Native	UPL	4	–	–
Compositae	*Stephanomeria tenuifolia*	Forb	Perennial	Native	UPL	4	1	1
Compositae	*Taraxacum officinale*	Forb	Perennial	NonNative	FACU	3	1	1
Compositae	*Viguiera multiflora*	Forb	Perennial	Native	UPL	5	–	–
Compositae	*Xanthium strumarium*	Forb	Annual	Native	FAC	1	–	–
Cornaceae	*Cornus sericea*	Shrub	Perennial	Native	FACW	2	–	–
Cruciferae	*Arabis selbyi*	Forb	Perennial	Native	UPL	1	–	–
Cruciferae	*Lepidium montanum*	Forb	Perennial	Native	UPL	1	–	1
Cruciferae	*Malcolmia africana*	Forb	Annual	NonNative	UPL	2	–	–
Cruciferae	*Stanleya pinnata*	Forb	Perennial	Native	UPL	1	–	–
Cruciferae	*Thelypodium integrifolium*	Forb	Perennial	Native	FAC		1	–
Cupressaceae	*Juniperus osteosperma*	Tree	Perennial	Native	UPL	19	2	2

Table A-1, cont. Number of sites at each park with species recorded during initial site characterization.

Family	Species[1]	Lifeform	Duration	Nativity	Wetland code[2]	Park[3]		
						ARCH (n=42)	CANY (n=3)	NABR (n=14)
Cyperaceae	*Carex aurea*	Graminoid	Perennial	Native	OBL	37	2	12
Cyperaceae	*Carex hystericina*	Graminoid	Perennial	Native	OBL	2	–	–
Cyperaceae	*Carex nebrascensis*	Graminoid	Perennial	Native	OBL	1	–	–
Cyperaceae	*Carex rossii*	Graminoid	Perennial	Native	UPL	–	–	5
Cyperaceae	*Carex* sp.	Graminoid	Perennial	Native	FACW	2	–	4
Cyperaceae	*Eleocharis palustris*	Graminoid	Perennial	Native	OBL	1	–	–
Cyperaceae	*Eleocharis rostellata*	Graminoid	Perennial	Native	OBL	–	1	–
Cyperaceae	*Eleocharis* sp.	Graminoid	_	Native	OBL	1	–	–
Cyperaceae	*Scirpus acutus*	Graminoid	Perennial	Native	OBL	2	–	–
Ephedraceae	*Ephedra viridis*	Shrub	Perennial	Native	UPL	14	1	2
Equisetaceae	*Equisetum hyemale*	Fern	Perennial	Native	FACW	1	–	–
Equisetaceae	*Equisetum laevigatum*	Fern	Perennial	Native	FACW	3	–	–
Ericaceae	*Arctostaphylos patula*	Shrub	Perennial	Native	UPL	–	–	1
Euphorbiaceae	*Euphorbia brachycera*	Forb	Perennial	Native	UPL	19	–	1
Fagaceae	*Quercus gambelii*	Shrub	Perennial	Native	UPL	27	1	7
Fagaceae	*Quercus welshii*	Shrub	Perennial	Native	UPL	7	–	–
Gentianaceae	*Swertia radiata*	Forb	Perennial	Native	UPL	–	–	11
Geraniaceae	*Erodium cicutarium*	Forb	Annual	NonNative	UPL	2	–	–
Gramineae	*Agrostis stolonifera*	Graminoid	Perennial	NonNative	FACW	4	1	–
Gramineae	*Aristida purpurea*	Graminoid	Perennial	Native	UPL	2	1	1
Gramineae	*Bromus tectorum*	Graminoid	Annual	NonNative	UPL	24	–	1
Gramineae	*Calamagrostis scopulorum*	Graminoid	Perennial	Native	FAC	16	3	12
Gramineae	*Elymus canadensis*	Graminoid	Perennial	Native	FACU	6	–	1
Gramineae	*Elymus elymoides*	Graminoid	Perennial	Native	UPL	8	–	1
Gramineae	*Elymus* sp.	Graminoid	Perennial	_	_	–	–	1
Gramineae	*Muhlenbergia andina*	Graminoid	Perennial	Native	FACW	3	–	2
Gramineae	*Muhlenbergia asperifolia*	Graminoid	Perennial	Native	FACW	3	1	4
Gramineae	*Muhlenbergia pauciflora*	Graminoid	Perennial	Native	UPL	–	–	1
Gramineae	*Muhlenbergia thurberi*	Graminoid	Perennial	Native	UPL	5	–	1
Gramineae	*Panicum acuminatum*	Graminoid	Perennial	Native	FACW	34	1	3

Table A-1, cont. Number of sites at each park with species recorded during initial site characterization.

Family	Species[1]	Lifeform	Duration	Nativity	Wetland code[2]	Park[3]		
						ARCH (n=42)	CANY (n=3)	NABR (n=14)
Gramineae	*Phragmites australis*	Graminoid	Perennial	Native	FACW	12	2	–
Gramineae	*Poa fendleriana*	Graminoid	Perennial	Native	UPL	1	1	–
Gramineae	*Poa pratensis*	Graminoid	Perennial	NonNative	FACU	6	1	–
Gramineae	*Poa* sp.	Graminoid	_	_	_	1	–	–
Gramineae	*Polypogon monspeliensis*	Graminoid	Annual	NonNative	FACW	6	–	–
Gramineae	*Schizachyrium scoparium*	Graminoid	Perennial	Native	FACU	21	1	3
Gramineae	*Sporobolus asper*	Graminoid	Perennial	Native	UPL	–	–	1
Gramineae	*Stipa comata*	Graminoid	Perennial	Native	UPL	5	–	–
Gramineae	*Stipa hymenoides*	Graminoid	Perennial	Native	UPL	2	1	8
Gramineae	*Stipa* sp.	Graminoid	Perennial	Native	UPL	1	–	1
Juncaceae	*Juncus arcticus*	Graminoid	Perennial	Native	FACW	6	1	6
Juncaceae	*Juncus articulatus*	Graminoid	Perennial	Native	OBL	1	–	–
Juncaceae	*Juncus bufonius*	Graminoid	Annual	Native	OBL	1	–	–
Juncaceae	*Juncus ensifolius*	Graminoid	Perennial	Native	FACW	2	–	–
Labiatae	*Hedeoma drummondii*	Forb	Perennial	Native	UPL	6	–	–
Leguminosae	*Astragalus* sp.	Forb	_	Native	_	1	–	–
Leguminosae	*Dalea oligophylla*	Forb	Perennial	Native	FAC	1	–	–
Leguminosae	*Lathyrus brachycalyx*	Forb	Perennial	Native	FAC	1	–	–
Leguminosae	*Melilotus officinalis*	Forb	Biennial	NonNative	FACU	8	1	–
Liliaceae	*Asparagus officinalis*	Forb	Perennial	NonNative	FACU	3	–	–
Liliaceae	*Smilacina stellata*	Forb	Perennial	Native	FAC	14	2	–
Liliaceae	*Zigadenus vaginatus*	Forb	Perennial	Native	OBL	23	2	12
Nyctaginaceae	*Mirabilis linearis*	Forb	Perennial	Native	FAC	3	–	–
Nyctaginaceae	*Mirabilis multiflora*	Forb	Perennial	Native	UPL	1	–	–
Nyctaginaceae	*Mirabilis oxybaphoides*	Forb	Perennial	Native	FAC	–	1	–
Oleaceae	*Forestiera pubescens*	Shrub	Perennial	Native	FACW	1	–	–
Oleaceae	*Fraxinus anomala*	Shrub	Perennial	Native	UPL	9	1	2
Onagraceae	*Epilobium ciliatum*	Forb	Perennial	Native	FAC	–	1	–
Onagraceae	*Oenothera caespitosa*	Forb	Perennial	Native	UPL	2	–	–
Onagraceae	*Oenothera longissima*	Forb	Perennial	Native	OBL	11	–	1

Table A-1, cont. Number of sites at each park with species recorded during initial site characterization.

Family	Species[1]	Lifeform	Duration	Nativity	Wetland code[2]	Park[3]		
						ARCH (n=42)	CANY (n=3)	NABR (n=14)
Orchidaceae	*Epipactis gigantea*	Forb	Perennial	Native	OBL	31	2	4
Orchidaceae	*Habenaria zothecina*	Forb	Perennial	Native	OBL	14	2	10
Pinaceae	*Pinus edulis*	Tree	Perennial	Native	UPL	14	2	12
Polemoniaceae	*Ipomopsis aggregata*	Forb	Perennial	Native	FACU	1	–	–
Polemoniaceae	*Ipomopsis roseata*	Forb	Perennial	Native	UPL	1	–	–
Polemoniaceae	*Ipomopsis* sp.	Forb	_	Native	_	–	–	1
Polygonaceae	*Eriogonum corymbosum*	Shrub	Perennial	Native	UPL	–	1	–
Polygonaceae	*Eriogonum microthecum*	DwarfShrub	Perennial	Native	UPL	–	–	1
Polypodiaceae	*Adiantum capillus-veneris*	Fern	Perennial	Native	FACW	26	–	4
Primulaceae	*Primula specuicola*	Forb	Perennial	Native	OBL	32	–	–
Pteridaceae	*Pellaea* sp.	Fern	Perennial	Native	UPL	2	–	–
Ranunculaceae	*Aquilegia micrantha*	Forb	Perennial	Native	OBL	36	2	13
Ranunculaceae	*Clematis ligusticifolia*	Vine	Perennial	Native	FACU	4	1	–
Ranunculaceae	*Ranunculus cymbalaria*	Forb	Perennial	Native	OBL	–	–	1
Rhamnaceae	*Rhamnus betulifolia*	Shrub	Perennial	Native	FAC	10	2	8
Rosaceae	*Amelanchier alnifolia*	Shrub	Perennial	Native	FACU	1	–	–
Rosaceae	*Amelanchier utahensis*	Shrub	Perennial	Native	UPL	3	1	4
Rosaceae	*Cercocarpus intricatus*	Shrub	Perennial	Native	UPL	2	1	3
Rosaceae	*Cercocarpus montanus*	Shrub	Perennial	Native	UPL	–	–	6
Rosaceae	*Coleogyne ramosissima*	Shrub	Perennial	Native	UPL	–	–	1
Rosaceae	*Holodiscus dumosus*	Shrub	Perennial	Native	UPL	1	–	–
Rosaceae	*Petrophytum caespitosum*	Forb	Perennial	Native	UPL	–	–	1
Rosaceae	*Rosa woodsii*	Shrub	Perennial	Native	FAC	6	2	–
Rubiaceae	*Galium multiflorum*	Forb	Perennial	Native	UPL	1	–	8
Rubiaceae	*Galium* sp.	Forb	_	Native	_	2	–	1
Salicaceae	*Populus fremontii*	Tree	Perennial	Native	FACW	1	–	–
Salicaceae	*Salix eriocephala*	Shrub	Perennial	Native	OBL	–	–	2
Salicaceae	*Salix exigua*	Shrub	Perennial	Native	OBL	3	–	–
Salicaceae	*Salix gooddingii*	Tree	Perennial	Native	FACW	1	–	–
Santalaceae	*Comandra umbellata*	Forb	Perennial	Native	UPL	13	–	6

Table A-1, cont. Number of sites at each park with species recorded during initial site characterization.

Family	Species[1]	Lifeform	Duration	Nativity	Wetland code[2]	Park[3]		
						ARCH (n=42)	CANY (n=3)	NABR (n=14)
Scrophulariaceae	*Castilleja linariifolia*	Forb	Perennial	Native	FAC	11	2	6
Scrophulariaceae	*Castilleja scabrida*	Forb	Perennial	Native	UPL	1	–	–
Scrophulariaceae	*Mimulus eastwoodiae*	Forb	Perennial	Native	OBL	41	1	1
Scrophulariaceae	*Penstemon eatonii*	Forb	Perennial	Native	UPL	1	–	–
Scrophulariaceae	*Penstemon* sp.	Forb	Perennial	Native	FACU	1	1	–
Scrophulariaceae	*Penstemon utahensis*	Forb	Perennial	Native	UPL	1	–	–
Typhaceae	*Typha* sp.	Forb	Perennial	Native	OBL	1	–	1
Ulmaceae	*Celtis reticulata*	Tree	Perennial	Native	FAC	12	–	–
Umbelliferae	*Aletes macdougalii*	Forb	Perennial	Native	UPL	–	–	3
Urticaceae	*Parietaria pensylvanica*	Forb	Annual	Native	FACU	1	–	–

[1]Taxonomy follows Welsh et al. (2008).
[2]Wetland Code UPL = upland, FACU = facultative upland, FAC = equally likely to occur in upland or wetland habitats, FACW = facultative wetland, OBL = obligate wetland.
[3]ARCH = Arches National Park, CANY = Canyonlands National Park, NABR = Natural Bridges National Monument.

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