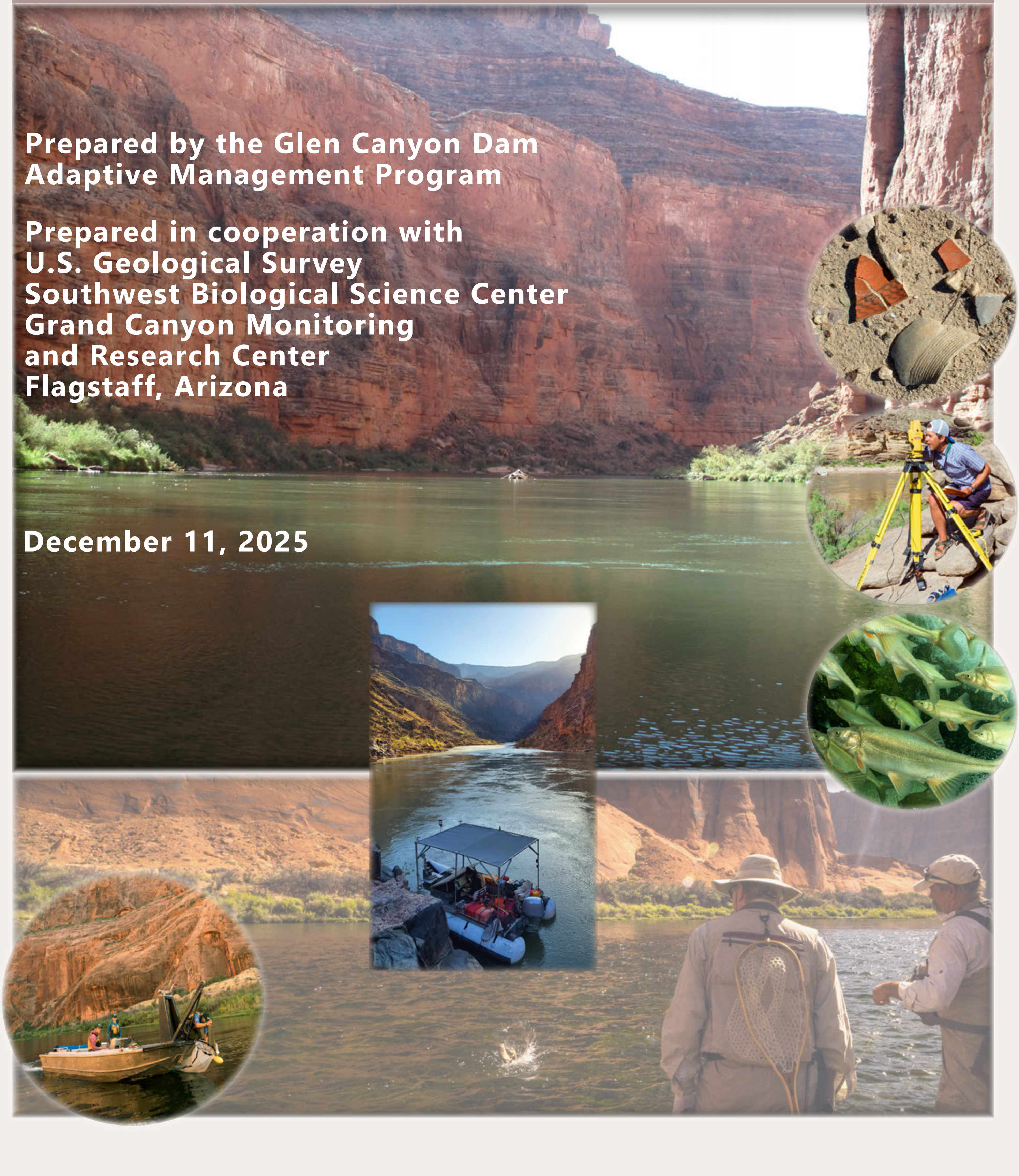
Status Report on the Long-Term Experimental & Management Plan (LTEMP) Metrics for the Glen Canyon Dam Adaptive Management Program



Prepared by the Glen Canyon Dam Adaptive Management Program

Prepared in cooperation with
U.S. Geological Survey
Southwest Biological Science Center
Grand Canyon Monitoring and Research Center
Flagstaff, Arizona

December 11, 2025

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Top background photo: A USGS research boat in eastern Grand Canyon, Colorado River, below Basalt Creek. Photo by Mike Moran, USGS.
Lower background photo: Two fisherman at Lees Ferry. Photo by David Herasimtschuk, Freshwaters Illustrated.
Inset lower left: Researchers on a boat in Grand Canyon. Photo by David Herasimtschuk, Freshwaters Illustrated.
Inset center: A research boat parked on the bank of the Colorado River. Photo by Katie Chapman, USGS.
Top inset, right: Pot shards in Grand Canyon. Photo by Amy East, USGS.
Top inset, middle: A USGS scientist conducts lidar scans of sandbars in Grand Canyon. Photo by Josh Caster, USGS.

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Abbreviations

- AC: Active Channel
- AF: Active Floodplain
- AGFD: Arizona Game and Fish Department
- APE: Area of Potential Effect
- Bug Flows: Macroinvertebrate Production Flows
- CPUE: Catch-per-unit-effort
- CRe: Colorado River ecosystem
- CRSP: Colorado River Storage Project
- DNF: Deviation from Natural Flow Metric
- DOI: U.S. Department of Interior
- EIS: Environmental Impact Statement
- EPT: Aquatic insect orders Ephemeroptera, Plecoptera, Trichoptera
- ESA: Endangered Species Act
- FEIS: Final Environmental Impact Statement
- FNU: Formazin Nephelometric Units
- FSC: Fluvial Sediment Connectivity
- ft^3/s: cubic feet per second (cfs)
- FY: Fiscal year
- GCD: Glen Canyon Dam
- GCDAMP: Glen Canyon Dam Adaptive Management Program
- GCMRC: Grand Canyon Monitoring and Research Center
- GCNP: Grand Canyon National Park
- GCPA: Grand Canyon Protection Act
- GCNRA: Glen Canyon National Recreation Area
- GPP: Gross primary production
- HBC: Humpback Chub
- HFE: High-Flow Experiment
- IF: Inactive Floodplain
- JCM: Juvenile chub monitoring
- LCR: Little Colorado River
- LTEMP: Long-term Experimental and Management Plan
- LTEMP EIS: Long-Term Experimental and Management Plan Environmental Impact Statement
- LTEMP ROD: Long-term Experimental and Management Plan Record of Decision
- NHDPlus: National Hydrography Dataset Plus
- NHPA: National Historic Preservation Act
- NNASMP: Nonnative Aquatic Species Management Plan
- NPS: National Park Service
- NRHP: National Register of Historic Places
- RKM: River Kilometers
- RM: River mile
- ROD: Record of Decision
- SBSC: Southwest Biological Science Center
- SDS: Sub-Daily Stage Metric
- SLI-Load Index
- TL: Total length
- TRGD: Trout reproductive and growth demographics
- TWP: Triennial Work Plan
- USFWS: U.S. Fish and Wildlife Service
- USGS: U.S. Geological Survey

Introduction

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In 2016, the U.S. Department of the Interior (DOI), through the Bureau of Reclamation (Reclamation) and the National Park Service (NPS), completed a Final Environmental Impact Statement (FEIS) for the Long-Term Experimental and Management Plan (LTEMP) for the Colorado River between Glen Canyon Dam and Lake Mead (DOI, 2016a). The purpose of the LTEMP is to provide a comprehensive framework for adaptively managing Glen Canyon Dam for a 20-year period consistent with the Grand Canyon Protection Act of 1992 (GCPA; Public Law 102-575 Title XVIII, October 30, 1992, 106 Stat. 4669) and other provisions of applicable Federal law. The LTEMP identifies specific options for dam operations (including hourly, daily, and monthly release patterns), non-flow actions, and appropriate experimental and management actions that meet GCPA requirements. Development of the LTEMP stemmed from the need to incorporate new scientific information developed since 1996 to better inform DOI decisions on Glen Canyon Dam operations and other management and experimental actions so that the Secretary of the Interior could continue to meet statutory responsibilities for protecting, mitigating impacts, and improving downstream natural and cultural resources for future generations, conserving species listed under the Endangered Species Act (ESA; Public Law 93–205, Approved December 28, 1973, 87 Stat. 884, as Amended Through P.L. 117–286, Enacted December 27, 2022), avoiding or mitigating impacts to National Register of Historic Places-eligible properties, and protecting the interests of American Indian Tribes, while meeting obligations for water delivery and generation of hydroelectric power (DOI, 2016a, b).

Eleven resource-specific goals were identified in the FEIS, which define the "fundamental objectives" of LTEMP (*sensu* Runge and others, 2015). Seven potential alternative management and experimental strategies were evaluated by the FEIS in terms of their ability to achieve those goals. Alternative D was determined to provide the best balance of performance among downstream resources to comply with the GCPA mandate to protect, to mitigate adverse impacts, and to improve the natural and cultural resources and visitor use values in the Grand Canyon National Park (GCNP) and Glen Canyon National Recreation Area (GCNRA) park units while continuing to comply with GCPA 1802 (b) applicable laws. The selection of Alternative D as the preferred alternative was codified in the Record of Decision (ROD) signed by the Secretary of the Interior on December 15, 2016 (DOI, 2016b).

Purpose of Metrics

Section 6.1 (c) of the 2016 LTEMP ROD states that "The DOI, in consultation with the Adaptive Management Working Group, will develop monitoring metrics for the goals and objectives using those in Appendix C of the FEIS as a starting point" (DOI, 2016b). Appendix C refers to a suite of "performance metrics" developed by Runge and others (2015) for evaluating the relative merits of the seven alternatives evaluated in the 2016 EIS. Those metrics were "intended to be objective measures of the performance of alternatives relative to goals for each affected resource being evaluated in the LTEMP EIS" (Runge and others, 2015). Comparisons of the performance of those metrics, among other considerations, ultimately led to selection of Alternative D as the preferred alternative.

The performance metrics developed by Runge and others (2015), while suitable for comparing draft alternatives, were not designed nor intended to measure resource outcomes based on empirical observations of resource condition following implementation of the LTEMP. In 2019, the DOI's Assistant Secretary for Water and Science directed the Bureau of Reclamation Glen Canyon Dam Adaptive Management Program (GCDAMP) to identify a set of "metrics" for assessing how well the selected alternative was meeting the goals and objectives stated in the LTEMP (DOI, 2016b). In fiscal years 2021-2023, Reclamation coordinated with the U.S. Geological Survey (USGS) Grand Canyon Monitoring and Research Center (GCMRC) to work with GCDAMP stakeholders in developing a new set of performance metrics for tracking progress towards attaining the resource goals specified in the LTEMP, using the previous performance metrics as a starting point.

For the purposes of this project, performance metrics are defined as objective, quantifiable measurements collected using standardized protocols that are indicative of resource condition and are an indicator of whether and how well the previously defined LTEMP goals are being achieved (Fairley, 2021). The scope of this effort encompasses the "fundamental objectives" of the LTEMP and the GCDAMP (*sensu* Runge and others, 2015) within the Colorado River ecosystem (CRe), as described in the LTEMP ROD. The focus is on tracking progress towards attaining the fundamental objectives (goals) of the program, as influenced by Glen Canyon Dam operations or management actions undertaken as part of the LTEMP.

The LTEMP FEIS (DOI, 2016a) and ROD (DOI, 2016b) identified the following 11 "fundamental objectives" or "goals" for the LTEMP:

1. **Archaeological and Cultural Resources**. Maintain the integrity of potentially affected National Register of Historic Places (NRHP)-eligible or listed historic properties in place, where possible, with preservation methods employed on a site-specific basis.

2. **Natural Processes**. Restore, to the extent practicable, ecological patterns and processes within their range of natural variability, including the natural abundance, diversity, and genetic and ecological integrity of the plant and animal species native to those ecosystems.
3. **Humpback Chub**. Meet Humpback Chub (*Gila cypha*) recovery goals, including maintaining a self-sustaining population, spawning habitat, and aggregations in the Colorado River and its tributaries below the Glen Canyon Dam.
4. **Hydropower and Energy**. Maintain or increase Glen Canyon Dam electric energy generation, load following capability, and ramp rate capability, and minimize emissions and costs to the greatest extent practicable, consistent with improvement and long-term sustainability of downstream resources.
5. **Other Native Fish**. Maintain self-sustaining native fish species populations and their habitats in their natural ranges on the Colorado River and its tributaries.
6. **Recreational Experience**. Maintain and improve the quality of recreational experiences for the users of the Colorado River ecosystem. Recreation includes, but is not limited to, flatwater and whitewater boating, river corridor camping, and angling in Glen Canyon.
7. **Sediment**. Increase and retain fine sediment volume, area, and distribution in the Glen, Marble, and Grand Canyon reaches above the elevation of the average base flow for ecological, cultural, and recreational purposes.
8. **Tribal Resources**. Maintain the diverse values and resources of traditionally associated Tribes along the Colorado River corridor through Glen, Marble, and Grand Canyons.
9. **Rainbow Trout Fishery**. Achieve a healthy high-quality recreational Rainbow Trout fishery in GCNRA and reduce or eliminate downstream trout migration consistent with National Park Service fish management and ESA compliance.
10. **Nonnative Invasive Species**. Minimize or reduce the presence and expansion of aquatic nonnative invasive species.
11. **Riparian Vegetation**. Maintain native vegetation and wildlife habitat, in various stages of maturity, such that they are diverse, healthy, productive, self-sustaining, and ecologically appropriate.

The overall purpose of the following chapters is to report on the status of a focused suite of performance metrics for each of the LTEMP goals so that federal managers, GCDAMP stakeholders, scientists, and the public, can assess progress towards achieving the resource goals and objectives defined in the LTEMP ROD (2016b). Most of the goals focus on maintaining or improving the condition of specific types of resources such as several fish species, cultural resources, sediment, or riparian vegetation.

Thus, most metrics focus on assessing the LTEMP's "performance" relative to those specific resource outcomes, such as whether current dam operations are in fact maintaining or increasing sediment above average base flow or attaining a population size for Humpback Chub consistent with the U.S. Fish and Wildlife Service (USFWS) recovery goals for that species. In other words, this report does not attempt to convey the status of every metric currently monitored by GCMRC and its cooperators on behalf of the GCDAMP, but rather, it reports on only a few key measurements for each goal that can allow managers and other stakeholders to assess progress in achieving the stated outcomes for each LTEMP goal.

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LTEMP Resource Goal 1. Archaeological Sites and Cultural Resources

> *"Maintain the integrity of potentially affected NRHP-eligible or listed historic properties in place, where possible, with preservation methods employed on a site-specific basis" (U.S. Department of the Interior [DOI], 2016a).*

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Background

The Long-Term Experimental and Management Plan (LTEMP) goal (Goal 1) for Archaeological and Cultural Resources is to maintain the National Historic Preservation Act (NHPA) integrity of historic properties *in situ* (U.S. Department of the Interior [DOI], 2016a). Beyond individual locations, LTEMP acknowledges that the region "should not be conceptualized merely as multiple discrete or detached archeological sites, traditional cultural properties, and/or sacred places; but rather viewed as interconnected, culturally symbiotic areas of traditional religious and cultural value" (DOI, 2016a; page 1). In this regard, Project D of the FY2025-27 Triennial Work Plan (TWP; DOI, 2024) addresses Goal 1 by quantifying changes in the physical condition of river corridor archaeological sites in Grand Canyon and the associated surrounding landscape as a function of i) dam operations, ii) experimental vegetation management, and iii) interacting natural processes and visitor impacts. Many cultural sites have physically deteriorated due to both human and non-human influences, with cumulative sediment loss (i.e., erosion) tied to regulated flows from Glen Canyon Dam being a primary driver of recent site degradation (Figure 1.1).

The dam and its operation are not the only sources of change affecting the Colorado River ecosystem (CRe) and associated archaeological sites. However, Glen Canyon Dam Adaptive Management Program's (GCDAMP's) Project D (DOI, 2024) focuses on studying and monitoring dam effects, in keeping with the mandates of the Grand Canyon Protection Act (GCPA) and consistent with the monitoring plan developed in 2015 and Reclamation's 2017 Historic Preservation Plan (U.S. Bureau of Reclamation, 2018). We evaluate these effects as they relate to Goal 1 using three metrics.

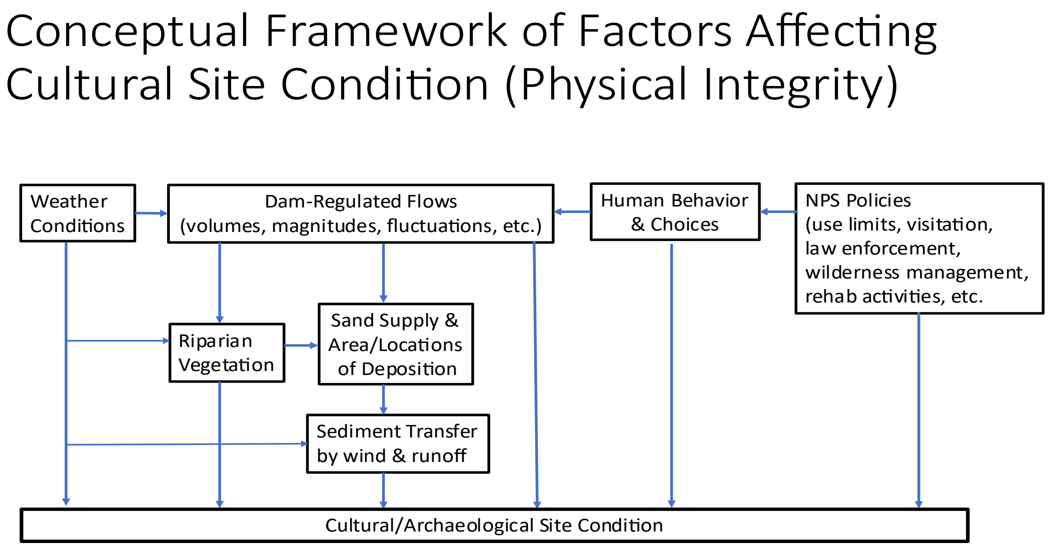

Figure 1.1. Conceptual model of drivers affecting archaeological site condition and National Historic Preservation Act (NHPA) integrity. National Park Service (NPS).

Metric 1.1, a qualitative assessment of site integrity, is determined based on monitoring conducted by the National Park Service (NPS) and is not directly measured as part of Project D; however, we periodically compare physical changes measured through Project D relative to NPS integrity assessments. Metric 1.2 is a quantitative summary of geomorphic changes at a sample of river corridor archaeological sites updated on an annual basis. Metric 1.3 is a categorical summary of geomorphic changes for all river corridor archaeological sites in the Grand Canyon Area of Potential Effect (APE) updated on the same schedule as the U.S. Geological Survey (USGS) Grand Canyon Monitoring and Research Center (GCMRC) (Project L; DOI, 2024) aerial overflight imagery collection.

Metric 1.1: Integrity

The term “integrity”, when used in the context of historic preservation, has a very specific meaning. Under the Secretary of Interior’s guidance for implementing the National Historic Preservation Act (NHPA), integrity is defined as “the ability of a historic property to convey its significance” (Federal Register, Vol. 48, No. 190, September 29, 1983). In this context, Metric 1.1 is an accounting of the number of archaeological sites and other historic properties within the Grand Canyon portion of the APE for Glen Canyon Dam (GCD) operations that have not maintained their integrity during a single year, and the cumulative number of those losses over multiple years. At the start of the LTEMP (December 2016), and as of August 2023, all 362 known archaeological sites within the Grand Canyon portion of the APE were determined to retain integrity; therefore, during the first 7 years of LTEMP, this metric value remained constant at zero.

Metric Summary Information

- *Metric Type:* Count of archaeological sites retaining integrity. This metric relies on periodic qualitative assessments of site integrity by Grand Canyon National Park (GRCA) cultural resource managers.
- *Data Required:* This metric relies on periodic qualitative judgments by National Park Service (NPS) cultural resource management professionals as to whether any site(s) in the Area of Potential Effect (APE) have lost integrity and therefore, are no longer significant under NRHP eligibility standards.
- *Metric Calculation:* A simple count of the number of sites that have lost integrity in a given year.
- *Frequency:* Annually, for a variable subset of the 362 sites located within the APE. Re-assessed approximately every five years for the entire population of sites in the APE.
- *Presentation:* Results are presented with summary text.

Data Collection

National Park Service archaeologists collect field observations on site integrity and make all official determinations for changes in this metric. During the Summer of 2024, USGS GCMRC scientists joined NPS archaeologists from Glen Canyon National Recreation Area (GCNRA) to collect photography and ground-based lidar during site visits to three river corridor archaeological sites. Although these sites are not included in the Grand Canyon APE, they had been monitored prior to the completion of the LTEMP, and there was interest in evaluating to what degree they might have changed during the past decade.

Results

While there were no determinations of change in integrity for any of the archaeological sites in the Grand Canyon APE in 2024 (Ellen Brennan, NPS archaeologist, oral communication, 2025), it is possible that changing physical conditions may contribute to changes in site integrity in the future. For example, NPS monitoring at site AZ C:02:0075 in the Glen Canyon APE indicated that at least one important feature, a fire pit (uncalibrated radiocarbon age = 2040 +/- 40 years before present) had been destroyed by erosion (Amy Schott, NPS archaeologist, oral communication, 2024), and subsequent repeat photography and ground-based lidar collected by USGS GCMRC during the May 2024 site visit confirmed that a large volume of sediment had eroded from a gully wall that contained the previously documented fire pit feature (Figure 1.2). Changes like these have the potential to impact and destroy site integrity.

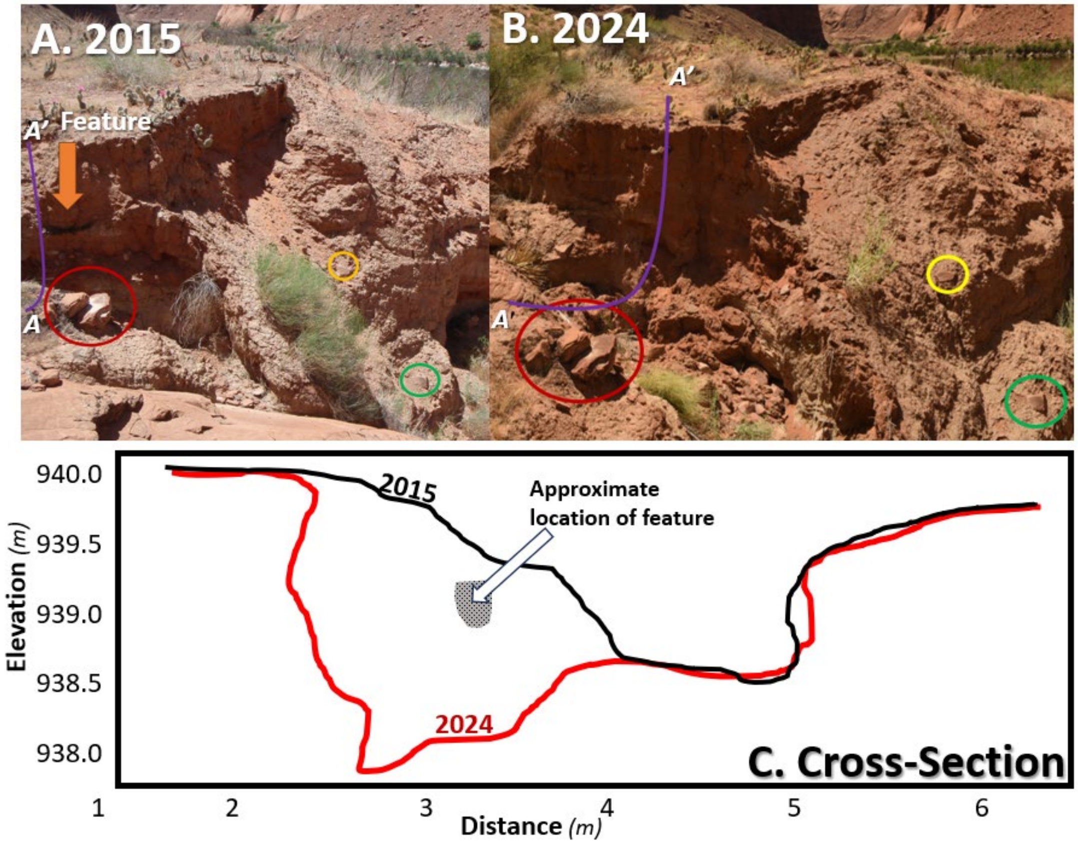

Figure 1.2. Field observations and measurements of geomorphic change and archaeological feature loss at site AZ C:02:0075 in Glen Canyon National Recreation Area between 2015 and 2024. Field photographs collected in 2015 (A) and 2024 (B) showing the location of a fire pit feature and a topographic cross-section (C) from ground-based lidar surveys collected on the same days. The approximate location of the feature is shown for reference. The gully wall has eroded laterally by more than a meter (~3 ft), removing visible evidence of the feature.

Summary and Interpretation

No sites were determined to have lost integrity in 2024 and so the metric remains unchanged. However, there are no “levels” or “degrees” of integrity. As explained in National Register Bulletin 15 (p. 44), “Historic properties either retain integrity (this is, convey their significance) or they do not.” Nevertheless, changes can and are taking place at some sites that have the potential to impact site integrity in the future (DOI, 1997). Measured losses of sediment and associated cultural features, such as those reported here for site AZ C:2:75, diminish the cultural record within the river corridor landscape. Dam operations like high-flow experiments (HFEs) can increase sediment availability and subsequently reduce erosion of archaeological sites in Marble and Grand Canyons (Sankey and others, 2018a, b; Caster and others, 2022) thereby lowering potential for site integrity changes. However, large flow fluctuations, such as those that occur with HFEs (e.g., most recently conducted in April 2023) can also increase erosion in some areas (Collins and others, 2014), particularly for sites in the Glen Canyon river corridor upstream of the Paria River confluence, which could contribute to site integrity loss.

Metric 1.2: Topographic Change at a Sample of Cultural Sites

Metric 1.2 relies on topographic change measured using repeat lidar surveys at a sample of the 362 river corridor archaeological sites in Grand Canyon National Park. Each site in the sample (currently 36 archaeological loci) is measured at least once every three years throughout Marble and Grand Canyons. Topographic change measurements at these locations are indicative of deposition and erosion within cultural sites, as well as sediment movement within the surrounding landscape. Measured topographic changes are largely driven by interactions between dam-regulated flows, river sediment, and local geomorphic processes that connect the river and its sediment to the surrounding landscape. This metric is directly reflective of the effects of dam-regulated flows on the sediment matrix that contains and protects cultural sites. It is based on a long-term dataset that includes consistent and repeatable measurements collected since 2010 (Collins and others, 2012, 2014). The measurements are normalized in relation to site size (in area) and survey interval (in years), which allows comparisons among sites (Caster and others, 2022). The sample includes sites of various dam-related classifications (refer to Metric 1.3 below) to ensure that the sample is generally representative of locations that are vulnerable to loss of integrity under varying flow and sediment supply conditions. This metric is specifically relevant for assessing the performance of the LTEMP with respect to Goal 1, because the main performance metric used for comparing LTEMP alternatives in the LTEMP FEIS was the "wind transport of sediment index" which sought to predict the likelihood of a positive response (increased sediment deposition by wind) at cultural sites in relation to the various proposed LTEMP alternatives (DOI 2016a, b). Furthermore, the fine resolution at which changes in sediment condition are measured with repeat lidar surveys (i.e., centimeter scale) have been shown to be sensitive to effects of HFEs (Sankey and others, 2018b), which are predicted to be a major factor for improving the condition of sand bars, aeolian source areas, and cultural sites over the LTEMP lifespan (DOI 2016a, b).

Metric Summary Information

- *Metric Type:* Continuous numeric variable. This metric is based on differences between repeat lidar-surveyed measurements of surface elevation changes at a sample of sites.
- *Data Required:* This metric relies on repeat measurements of surface topography at ~36 cultural sites using terrestrial lidar within and immediately surrounding the sites. Measurements are derived by comparing 3-dimensional mapped surfaces of each site approximately once every three years.
- *Metric Calculation:* For each site survey, the volume and location of sediment added or eroded since the previous survey is measured and mapped. Volumetric change is then normalized by site area. Specific methods of measuring topographic change are described in multiple documents (Collins and others, 2016; East and others, 2016; Caster and others, 2022).

- *Frequency:* Annually, for about one third of the total sample each year, typically in April-May. This schedule permits calculation of the metric approximately once every three years.
- *Presentation:* Data are presented with bar charts of measured change (refer to Figure 1.3 below).

Data Collection

The USGS GCMRC conducted ground-based lidar surveys at 16 archaeological sites in September 2024 (C:05:0031, C:05:0037, C:13:0006, C:13:0334, C:13:0336, C:13:0098, C:13:0099, C:13:0101, C:13:0321, C:13:0092, C:13:0009, B:14:0105, B:10:0237, A:15:0005, G:03:0058, and G:03:0072. See Table 1.1 in Appendix 1 of Caster and others, 2022, for more information about these sites). In addition to ground-based lidar surveys, observations on local weather conditions within the canyon were collected from six weather stations. These observations help to interpret the metric results by identifying potential geomorphic interactions with river sediment. These data are currently being processed for measuring and interpreting topographic changes for Metric 1.2.

Results

Metric 1.2 is a single value for each of the 36 archaeological sites or site loci representing a mean rate of net elevation change for the full record of observations through 2023 (Figure 1.3). Values greater than zero indicate a trend of increasing sediment (deposition) and values less than zero indicate decreasing sediment (erosion). For context, a measured change of -1 mm per year for an archaeological site covering 0.5 hectare (1.2 acres) means that, on average, approximately 8.4 metric tons of sediment were eroded each year from the first to the most recent survey interval. Between 2010 and 2023 the summary of net changes (Figure 1.3) indicates that 18 of the 36 sites have eroded and 18 have aggraded. Net changes are summarized by the fluvial sediment connectivity (FSC; a.k.a. “aeolian”) classification (refer to Metric 1.3 below for information about this classification) and show relatively large amounts of erosion for some Type 1 sites which have a more direct connection to dam-regulated flows (Type 1; Figure 1.3 and 1.4 left panel). This pattern correlates with larger total elevation changes (sum of absolute values of erosion and deposition) representative of sediment transport by wind (Caster and others, 2024; Figure 1.4 right panel).

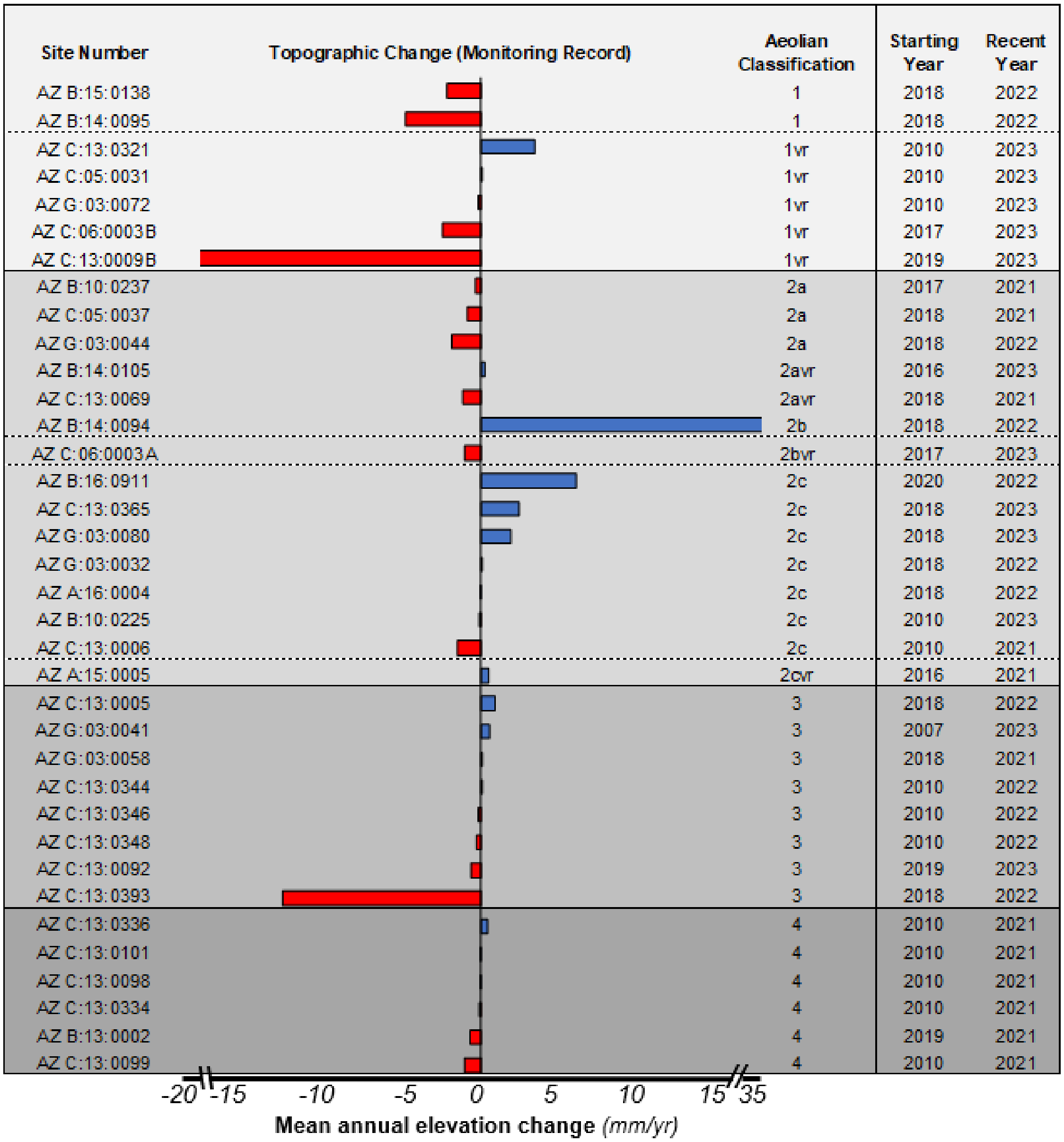

Figure 1.3. Bar graph showing annual mean topographic change for river corridor archaeological sites in the Grand Canyon Area of Potential Effect that have been monitored repeatedly with lidar between 2010 and 2023. Positive values (blue bars) indicate site physical integrity has been maintained over the timeframe (since starting year; e.g., 2010), whereas negative values (red bars) indicate sites that have degraded (lost surface sediment) during that same timeframe. Definitions of the aeolian classifications Types 1, 2a, 2b, 2c, 3, etc. are provided under Metric 1.3; "vr" designates sites where experimental vegetation removal has occurred. Provisional data, subject to change.

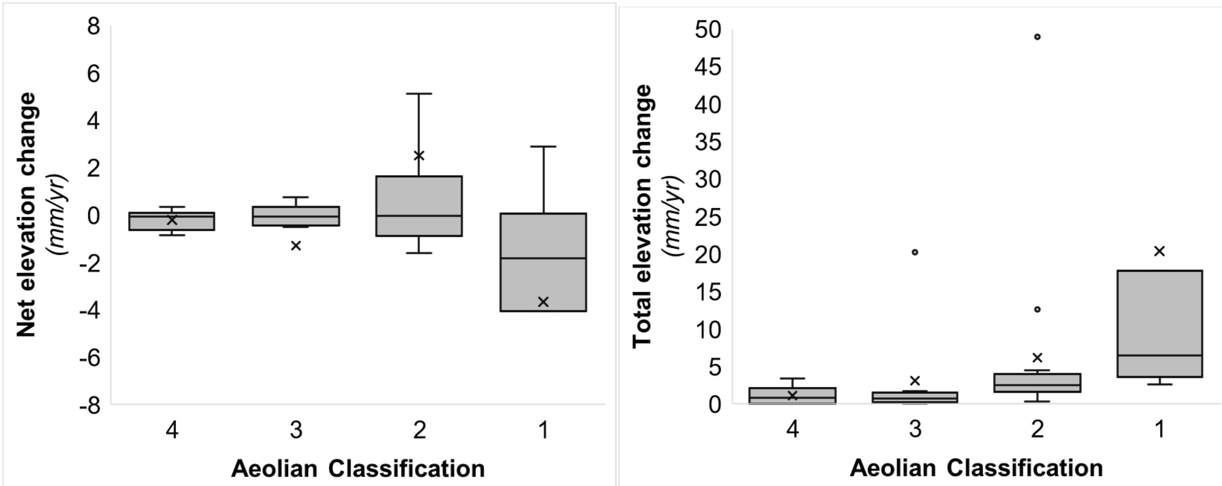

Figure 1.4. Figures showing (left panel) net elevation change (mm/yr), and (right panel) total elevation change (mm/yr) at archaeological monitoring sites within the Grand Canyon Area of Potential Effect, grouped by aeolian classification. Note that two extreme outliers, one in Class 1 and one in Class 2, are not shown in these plots; they are responsible for skewing the averages (marked with an 'x') in these plots. These plots summarize the monitoring data presented in Figure 1.3. Provisional data, subject to change.

Summary and Interpretation

Metric 1.2 shows that between 2010 and 2023 half of the sample of 36 monitored sites eroded and half aggraded. Larger amounts of erosion have occurred at some sites that have a direct connection to river sandbars (Type 1) as of May 2023. Most Type 1 aeolian classification sites have upwind sandbars that have undergone experimental vegetation management implemented by the NPS annually since 2019. The purpose of the experimental vegetation management is to reduce vegetation cover on river sandbars to increase windblown sand transport to the downwind archaeological sites in an effort to achieve the NPS management goal of preservation in place (Little and others, 2000; NPS, 2006). While the net elevation change (Figure 1.4 left panel) indicates that Type 1 aeolian classification sites are eroding, on average, the total elevation change (Figure 1.4 right panel) indicates that significantly greater transport of sediment is occurring through those sites, likely due to the upwind vegetation management efforts. Sites with the less direct connection to river-sourced sediment supply (Types 3 and 4) tended to have lower rates of topographic change and likely slower responses to dam operations. This means that dam operations that could promote new sediment deposition for a Type 3 site like C:13:0393 (Figure 1.3) might take longer to be effective, increasing potential for degradation of cultural material. To continue evaluation of these sites, new experimental management strategies can be identified to help retain sand on sites that is blown by wind from the upwind sandbars.

Metric 1.3: Change in Vulnerability to Loss of Integrity

This metric combines two classification schemas based on dam operations and associated geomorphic interactions that are monitored on decadal scales. The first classification system, known as the fluvial sediment connectivity (FSC) classification (a.k.a. "aeolian classification"), describes the degree to which cultural sites can receive wind-blown sediment from fluvial sand bars (East and others, 2016, 2017; Caster and others, 2022; Sankey and others, 2023). The second classification system, called the drainage classification, describes the degree to which gullies that cut into or across archaeological sites are integrated with the dam-regulated Colorado River (local base level) (East and others, 2016, 2017; Caster and others, 2022; Sankey and others, 2023). Together, these two classifications describe changes to the vulnerability of cultural sites from past and future degradation that can lead to loss of integrity linked to dam operations.

The FSC classification system defines 5 classes or types of archaeological sites. Types 1–4 define those sites whose geomorphic context includes river-derived sand as an integral component—either fluvial, aeolian, or both. Type 5 defines sites at which river-derived sand is absent or, if present, is incidental to the geomorphic context. The site-type definitions are as follows (East and others, 2016, 2017):

- *Type 1:* Sites with an adjacent, upwind, recent subaerial fluvial sand deposit, and where there are no substantial barriers to impede aeolian sand transport from the flood deposit toward the archeological site.
- *Type 2:* Sites with an adjacent, upwind, recent subaerial fluvial sand deposit, but with a barrier separating the flood deposit from the archeological site. Barriers limit potential aeolian sand transport from the fluvial deposit toward the archeological site but may not eliminate sand movement entirely from sandbar to archeological site. We defined three subtypes:
    - *Type 2a:* Vegetation barrier present (may be riparian vegetation or higher-elevation, non-riparian upland vegetation).
    - *Type 2b:* Topographic barrier present (most commonly a tributary channel, but in several cases a steep bedrock cliff or large boulder deposit).
    - *Type 2c:* Both vegetation and topographic barriers present.
- *Type 3:* Sites at which an upwind shoreline exists for a recent high flow deposit, but where the recent high flow resulted in no open, unvegetated sandbar along the river margin.
- *Type 4:* Sites at which there is no upwind shoreline corresponding to a recent high flow, but whose geomorphic context involves river-derived sand.
- *Type 5:* Sites in the river corridor at which Colorado River-derived sand is absent or is only incidental to site context, such as sites situated entirely on bedrock or talus.

The drainage classification provides a periodic assessment of the degree to which gullies have "evolved" over time and become integrated with the mainstem river. Using aerial imagery and site visits, we evaluate drainage channels (rills, gullies, and arroyos, in order of increasing size) at each of the archaeological sites by noting whether such drainage systems are present within or adjacent to each site. We also document the downslope extent of the drainage, that is, the base level to which each drainage grades. Sites are classified into one of four categories (East and others, 2017; Figure 9):

- *Type D1:* no drainages
- *Type D2:* terrace-based drainages only
- *Type D3:* side-canyon-based drainages
- *Type D4:* river-based drainages

Metric Summary Information

- *Metric Type:* Categorical numeric. This metric is based on empirical observations of independently verifiable site characteristics using aerial images coupled with field verification for all sites located within the Grand Canyon portion of the Area of Potential Effect (APE) from dam operations, as defined in LTEMP.
- *Data Required:* Archaeological site boundaries plotted on new or recent aerial imagery throughout the river corridor, combined with field observation of drainage characteristics, changes in vegetation and sand bars, and direct or inferred knowledge of prevailing wind patterns; these data are compiled once every 5-10 years as part of Project D.
- *Metric Calculation:* Each site is categorized in terms of 1) its connectivity to fluvial sources of sediment through the medium of wind transport, based on a visual assessment (East and others, 2016), and 2) the degree to which drainages are present and/or integrated with the mainstem Colorado River. These classifications are then re-categorized in terms of whether and in what direction the classifications have changed since the last observation (i.e., increased FSC classification number and/or increased drainage classification number = increased vulnerability to integrity loss).
- *Frequency:* Approximately once every 5-10 years (decadal).
- *Presentation:* A chart (Figure 1.7) that synthesizes changes in classification status since the last time sites were classified (Figures 1.5 and 1.6).

Data Collection

During FY2024, the USGS GCMRC documented geomorphic conditions related to both classification systems during the ground-based lidar surveys at each of the 16 archaeological sites monitored last year (listed in Metric 1.2).

Site observations conducted by project staff during fieldwork in 2024 indicated that gully development had increased at many sites, and sandbars that had new deposition after the May 2023 high flows had lost sediment following sustained higher base flows during the summer and fall of 2023. Additionally, they collected 42 repeat photographs corresponding to historical collections from the late 1800s to the 1970s that permit further documentation of past and current landscape changes.

The last complete account of changes relative to the two classification systems was completed in 2022 following the most recent overflight (Figures 1.5, 1.6, and 1.7). The next complete update of these classifications and this metric can take place with the next overflight photography mission.

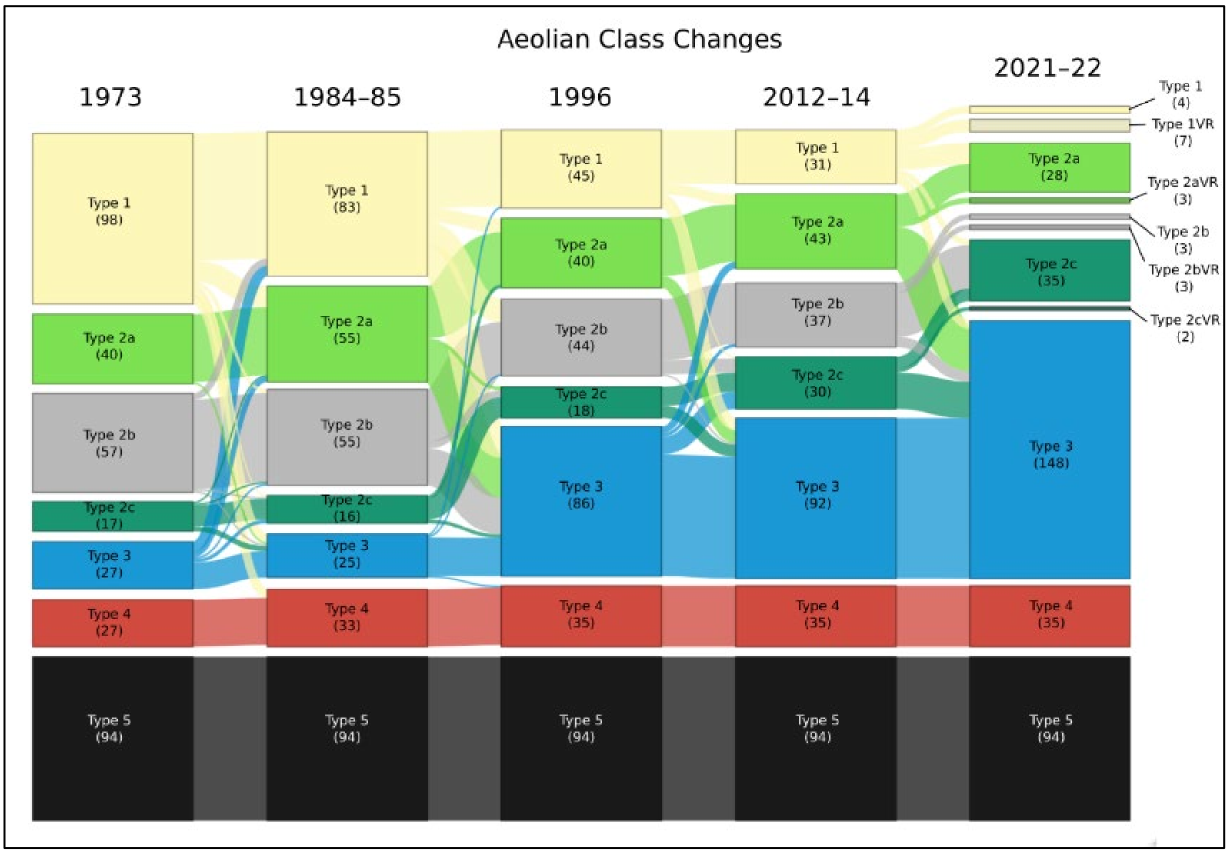

Figure 1.5. Changes and directions of change in fluvial sediment connectivity (FSC) classifications between 1973 and 2022 (from Sankey and others, 2023). Note the large decrease in Type 1 sites and concurrent increase in Type 3 sites since 1973. Vr refers to sites where experimental vegetation removal has occurred.

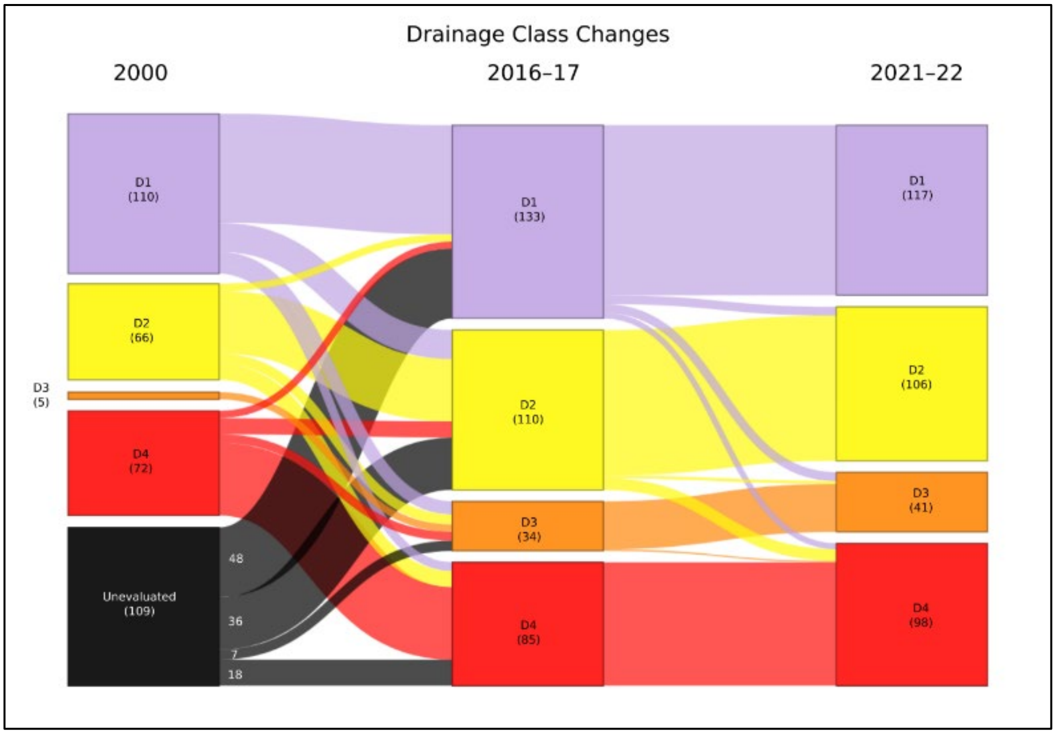

Figure 1.6. Changes in drainage classifications between 2000 and 2022 (from Sankey and others, 2023). Note that all changes in drainage classification since 2016 have moved from a numerically lower drainage class to a higher class, indicating a more eroded condition over time.

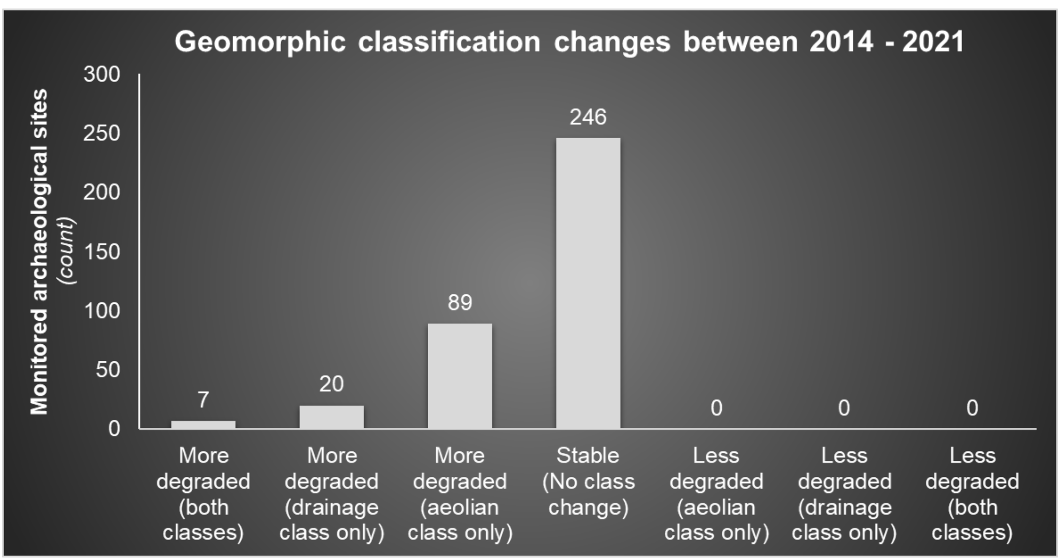

Figure 1.7. Changes in both fluvial sediment connectivity (FSC) classification status and drainage classification status for 362 sites in the Grand Canyon Area of Potential Effect (APE), showing that while the majority of sites (n=246) have not changed classification status, those that have changed are now in a more degraded condition, either due to a change in FSC class (n=89), a change in drainage class (n=20), or both (n=7). Provisional data, subject to change.

Summary and Interpretation

The multi-decadal record of observations related to Metric 1.3 shows that archaeological sites in Marble and Grand Canyons are losing river-sourced sediment and may be becoming more vulnerable to changes in site integrity.

The most recent update (2022; Figure 1.5) shows that the number of sites with the greatest connection to the river (FSC Type 1) are declining, primarily from increases in invasive riparian vegetation. This appears to correlate with increased development of gullies within archaeological sites that have potential to impact or remove cultural materials (Figure 1.6). Site observations during 2024 appear to support this interpretation as gully development was notably greater at many sites visited during the field surveys.

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LTEMP Resource Goal 2. Natural Processes

> *"Restore, to the extent practicable, ecological patterns and processes within their range of natural variability, including the natural abundance, diversity, and genetic and ecological integrity of the plant and animal species native to those ecosystems" (U.S. Geological Survey [DOI], 2016a).*

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Background

Key natural processes that sustain riverine and riparian ecosystems include disturbance patterns, water availability, water quality, geomorphic template, sediment regimes, and nutrient cycling (Poff and others, 1997; Ward and others, 2001; Wohl and others, 2015; Gurnell and others, 2016; Weigelhofer and others, 2018). The Long-Term Experimental and Management Plan (LTEMP) goal for natural processes identifies "ecological patterns and processes" and the resulting living communities as the key components of natural processes for the Colorado River ecosystem (CRe; DOI, 2016a). Here, we focus on elements of these natural processes that 1) are primary drivers of ecological patterns in the CRe or are indicators of biological response to primary drivers, 2) are influenced by management actions, and 3) are not already included in other LTEMP goals. We therefore focus on river flow patterns, which control disturbance patterns and water availability, and the flow of energy through food webs, which are indicators of water quality, nutrient cycling, and biotic response to river management.

The metrics proposed below focus on functional aspects of the hydrograph (Metrics 2.1– 2.2; Palmquist and others, 2025), the production of carbon for fueling aquatic food webs (2.3), and measures of dependent biological communities (2.4). Other key outcomes of the LTEMP Natural Processes goal are evaluated under other LTEMP goals, such as the sediment goal (increase and retain fine sediment, Goal 7), riparian vegetation goal (abundance and diversity of plants, Goal 11), and a suite of native and nonnative fish-related goals (abundance and diversity of fishes, Goals 3, 5, 10; DOI 2016a).

Choosing metrics is inherently value laden given they correspond to a human-defined management goal. This is further complicated when considering natural processes because enhancing a single natural process may not improve the status of every riverine resource (Richter and others, 1996; Schmidt and others, 1998).

Schmidt and others (1998) point out “the decision to manipulate ecosystem processes and components involves not only a scientific judgment that a restored or rehabilitated condition is achievable, but also a value judgment that this condition is more desirable than the status quo.”

Cultural perspectives inform these value judgements and may differ widely (Wantzen, 2024). For example, opinions on how human activities are (or are not) considered as part of natural processes guide the way metrics are defined here. Some unresolved ideas pertaining to natural processes include 1) metrics that can capture human impacts on terrestrial landscape processes; 2) alternate measures of ecosystem productivity (e.g., related to land-based carbon inputs, which historically drove production pre-dam); and 3) a metric that captures differences from the pre-dam thermal regime.

Here, we present a range of metrics that can be calculated given existing and currently collected data, are affected by Glen Canyon Dam Adaptive Management Program (GCDAMP)-related decision-making and have an impact on the river’s capacity to support diverse life. These metrics have not been fully finalized by the GCDAMP and are presented here to facilitate discussion and provide timely science that can be used for management decisions. While the process for developing and calculating the flow metrics is provided in a recent publication (Palmquist and others, 2025), additional metrics are described here for gross primary productivity (GPP) and insect diversity). Both the metrics themselves and their analyses are subject to change as the LTEMP metrics are reviewed and finalized.

Metric 2.1: Deviation from Natural Flow

The first metric, the deviation from natural flow metrics (DNF) is an integrative measure of how much observed flow patterns differ from expected natural flow patterns (deviation from natural flow, M2.1), which is intended to support holistic, whole-year, flow optimization efforts. Three sub-metrics are provided to highlight alterations to seasons within the overall annual hydrograph. The DNF metric and all three sub-metrics calculate the number of standard deviations that any daily discharge measurement is from an estimated natural discharge for that same day (Palmquist and others, 2025) 2000-2022.

A water year is defined as October 1 to September 30 of the following year, with the year designated by the calendar year in which the water year ends. The observed daily discharge for the CRe is represented by data from the U.S. Geological Survey Colorado River at Lees Ferry, AZ gage, 09380000 (U.S. Geological Survey, 2025a). The z-score for each day in water years 1963 through 2024 was calculated by subtracting the mean discharge of the baseline day from the observed daily discharge and dividing by the standard deviation of the baseline day. The absolute values of these daily z-scores are then averaged across a year to get an annual DNF value. Detailed methods for calculating the flow metrics can be found in Palmquist and others (2025).

Essentially, this is a measure of how many standard deviations the observed flow is from the natural flow. Values near 0 indicate more natural flows, while larger values indicate less natural flows. In a naturally flowing river, annual values would vary, but values of 2 would only occur about 5 percent of the time and even less for those greater than 2.

The same methods were used for the seasonal sub-metrics (functional flows *sensu* Yarnell and others, 2015), but average across certain time periods rather than the entire year. The peak flow metric is calculated slightly differently, in that only the peak discharge is used for each year, so days are not averaged within a year. Thus, absolute values are not used, and the metrics can be positive (greater than natural) or negative (lower than natural).

In 2024, the DNF and low flow metrics were relatively low (close to natural) as compared with the historical post-dam record. The DNF metric (M2.1) was 1.43 (Figure 2.1A). This is the third lowest natural flow metric score during LTEMP (water years 2022 and 2023 were 1.31 and 1.13 respectively). The score is also one of the lowest of the entire post-dam record. Note that 2023, which had a spring high-flow experiment, is one of the lowest DNF scores in the period of record. The only additional years with DNF metrics lower than 2024 are 1964-1968 and 1971. The low metric score is consistent with the historic pattern where years with low water releases tend to have annual hydrographs closer to the natural baseline and thus lower metric scores (Palmquist and others, 2025). The low flow metric (M2.1a) was 2.12, falling below the first quartile of the historic distribution (Figure 2.1B).

In 2024, the peak flow and monsoon variability metrics were relatively far from natural as compared with the historical post-dam record. The 2024 peak flow metric (M2.1b) was -1.74 (Figure 2.1C) which is the fourth furthest below the natural peak flow of any year on record (2022, 2010, and 2014 had the smallest observed peak flow relative to natural). Note that the 2023 spring high-flow experiment pushed this metric much closer to natural (close to 0). The 2024 monsoon season metric (M2.1c) was 1.48, which is above the third quartile of the historic distribution (Figure 2.1D). The 2024 monsoon metric was the highest (furthest from natural) of all the LTEMP water years.

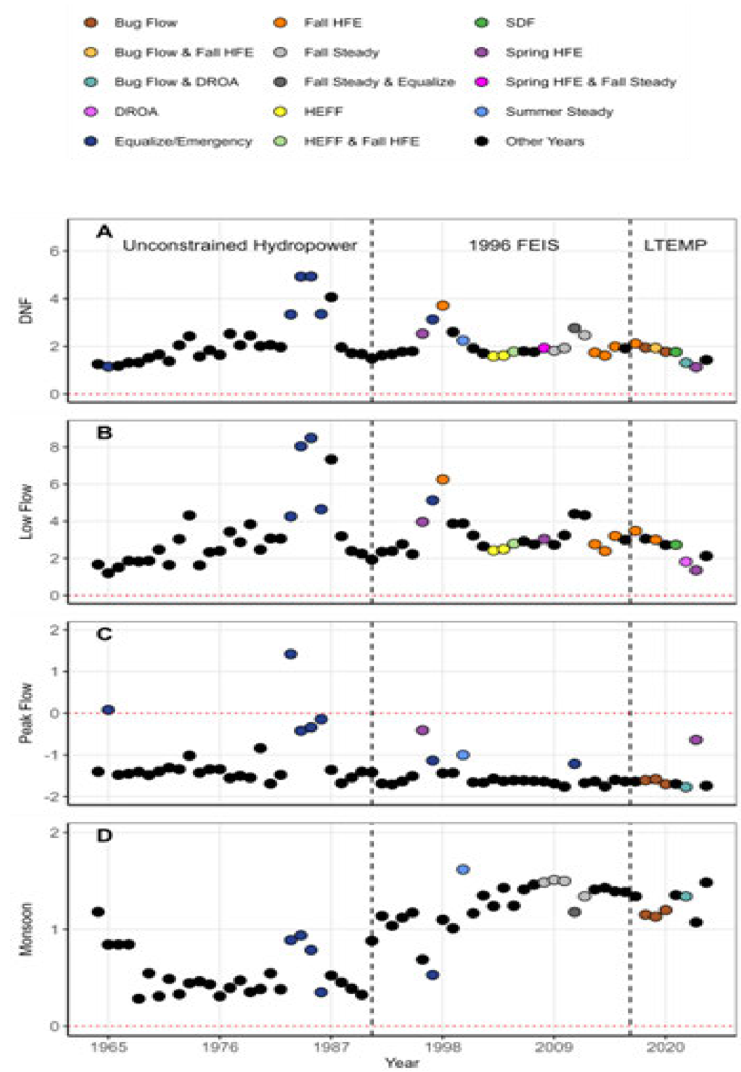

Figure 2.1. Annual metric values in the Colorado River ecosystem for the deviation from natural flow metric (A), the low flow sub-metric (B), the peak flow sub-metric (C), and the monsoon sub-metric (D). Vertical, dashed lines indicate the start of the 1991 agreement (Interim Operation Criteria for Glen Canyon Dam) which supported early implementation of modified low fluctuating flows under the 1996 Final Environmental Impact Statement (1996 FEIS; U.S. Department of the Interior, 1996) and the start of the Long-Term Experiment and Management Plan (LTEMP; U.S. Department of the Interior, 2016a). Colors indicate policy and experimental flow patterns that occurred during the time period of the metric: Bug Flow – macroinvertebrate production flow protocol, Equalize/Emergency – high releases to move water to Lake Mead, DROA – drought response operations agreement, Fall Steady Flows – low steady flows in September and October, HEFF – high experimental fluctuating flows, HFE – high-flow experiment, Summer Steady – steady flows in summer of 2000, and SDF – spring disturbance flow. Figure is modified from Palmquist and others, 2025; and updated metrics for water years 2023 and 2024 were calculated using discharge data from the Colorado at Lees Ferry, AZ USGS gage 09380000, available at https://www.gcmrc.gov/discharge_qw_sediment/station/GCDAMP/09380000 (U.S. Geological Survey, 2025a), and unregulated inflow estimates are available by selecting “Lake Powell” from the dropdown window and selecting “Unregulated Inflow” at https://www.usbr.gov/rsvrWater/HistoricalApp.html (U.S. Bureau of Reclamation Upper Colorado Region, 2025).

Metric 2.2: Subdaily Stage Fluctuation

Subdaily changes in river flow are a characteristic of flow management for hydropower (McManamay and others, 2016), yet this aspect of flow alteration is not captured in the daily discharge data that are used to compute the M2.1 metrics. Thus, we included the subdaily stage fluctuation metric (SDS; M2.2) that compares observed and expected daily stage change at Lees Ferry, AZ. This location has a lengthy (>100 years) period of record and is a comparatively wide part of the river, which means these values are a conservative estimate of stage change throughout the CRe (smaller stage change than narrow parts of the river). Both expected and observed daily stage change at Lees Ferry are represented by stage data from the U.S. Geological Survey Colorado River at Lees Ferry, AZ gage, 09380000 (Topping and others, 2003; U.S. Geological Survey, 2025a). Expected daily stage change is based on pre-dam stage data collected at Lees Ferry from water years 1925 through 1945. Observed daily stage change is based on water years 1963 through 2024.

First, daily stage change is calculated as the maximum stage minus the minimum stage for each day for both the natural flow period and the post-dam flow period. The mean and standard deviation of stage change is calculated for each Julian day of the natural flow period.

The same z-score approach is used by subtracting the mean stage change of the baseline Julian day from the observed daily stage change and dividing by the standard deviation of the baseline day. The absolute values of these daily z-scores are then averaged across a year to get an annual SDS value. Detailed methods for calculating the flow metrics can be found in Palmquist and others (2025). As for the M2.1 metrics, values near 0 indicate more natural flows, while larger values indicate less natural flows. In a naturally flowing river, annual values would vary, but values of 2 would only occur about 5 percent of the time and even less for those greater than 2.

In 2024, the SDS metric was relatively low (closer to natural) as compared with the historical post-dam record. The SDS metric (M2.2) was 6.85, falling below the first quartile of the historic distribution (Figure 2.2). With the exception of 1984 and 1985, annual SDS metric scores for the Colorado River have been higher than the DNF metric across all the years since the dam was closed (Palmquist and others, 2025). Across the full historical record, the average annual SDS metric score is also further from 0 than any of the three DNF sub-metrics (low flow, peak flow, and monsoon). In 2024, the SDS metric was the highest (furthest from natural) of any of the flow metrics calculated for this study.

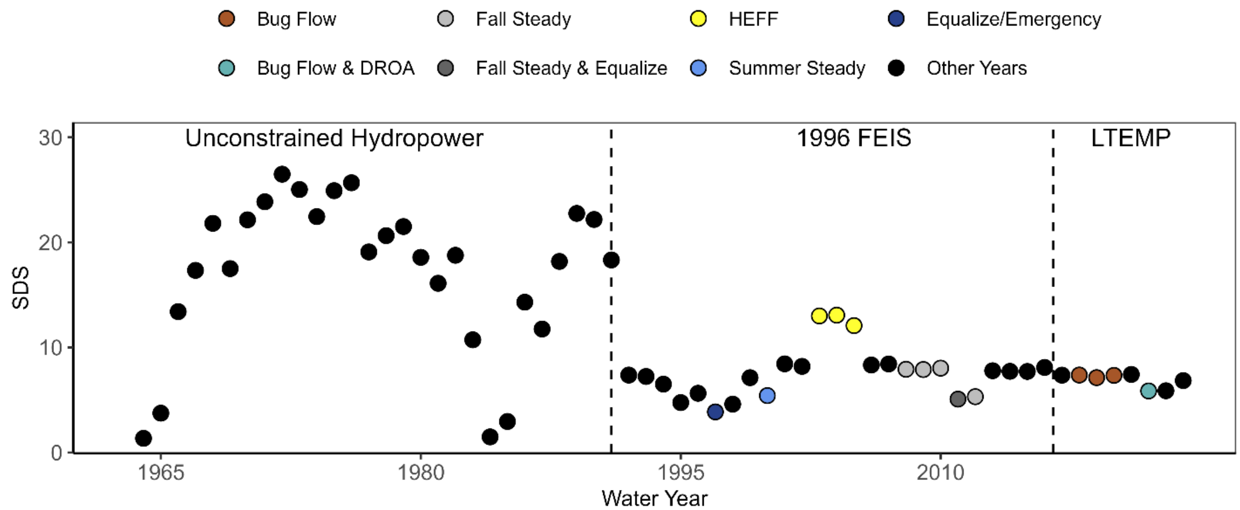

Figure 2.2. Annual metric values in the Colorado River ecosystem for subdaily stage fluctuations. Vertical, dashed lines indicate the start of the 1991 agreement (Interim Operation Criteria for Glen Canyon Dam) which supported early implementation of modified low fluctuating flows under the 1996 Final Environmental Impact Statement (1996 FEIS; U.S. Department of Interior, 1996) and the start of the Long-Term Experiment and Management Plan (LTEMP; U.S. Department of Interior, 2016). Colors indicate policy and experimental flow patterns that occurred during the time period of the metric: Bug Flow – macroinvertebrate production flow protocol, DROA – drought response operations agreement, Equalize/Emergency – high releases to move water to Lake Mead, Fall Steady Flows – low steady flows in September and October, HEFF – high experimental fluctuating flows, and Summer Steady – steady flows in summer of 2000. Figure is modified from Palmquist and others, 2025, and updated metrics for water years 2023 and 2024 were calculated using discharge data from the Colorado at Lees Ferry, AZ USGS gage 09380000 available at https://www.gcmrc.gov/discharge_qw_sediment/station/GCDAMP/09380000 (U.S. Geological Survey, 2025a), and data from the inflow gages (Colorado River near Cisco, UT, 09180500; Green River at Green River, UT, 09315000; San Rafael River near Green River, UT, 09328500; and San Juan River near Bluff, UT, 09379500) are available from the U.S. Geological Survey National Water Information System at https://waterdata.usgs.gov/nwis (U.S. Geological Survey, 2025b).

Metric 2.3: Springtime Gross Primary Productivity in Marble and Grand Canyons

Dam-induced changes in flow, temperature, and sediment regimes have caused a shift in the processes that fuel CRe aquatic food webs. Namely, growth of the invertebrate prey base and native fish populations is fueled by in-river primary production of algae (Stevens and others, 1997; Cross and others, 2013) whereas food webs of the pre-dam river were likely fueled by decomposition of terrestrial leaf-litter and detritus (Woodbury and others, 1959; Kennedy and Ralston, 2012). Although riparian vegetation in the CRe was historically sparse (Turner and Karpiscak, 1980; Kennedy and Ralston, 2012), debris-laden floods contained substantial leaf litter and wood accumulated from vast upstream sources (Haden and others, 2003). Conversely, turbidity levels in the pre-dam river never dropped below 30 formazin nephelometric units (FNU; Voichick and Topping, 2014), functionally eliminating the process of aquatic primary production as a potential food source for riverine food webs.

Present-day analyses of downstream trends in dissolved organic carbon support this inference and show a marked shift from terrestrially derived carbon upstream of Lake Powell to algae-derived dissolved organic carbon signatures below Glen Canyon Dam (Miller, 2012). In the present-day food web, periodic tributary floods do supply some allochthonous carbon inputs that propagate through food webs (Sabo and others, 2018; Behn and Baxter, 2019), but macroinvertebrates and fishes derive most of their production from algae owing to higher assimilation efficiencies (Wellard-Kelly and others, 2013). Diatoms, a widespread class of microscopic algae, dominate the trophic basis of invertebrate production (Cross and others, 2013). In addition to directly supporting macroinvertebrate production, primary producers are an important food source for native fishes such as the Flannelmouth Sucker (*Catostomus latipinnis*) and Razorback Sucker (*Xyrauchen texanus*) whose downturned, sub-terminal, mouths allow them to more easily consume algae and other food types that are on the river-bottom (Hansen and others, 2023). Owing to the important role that algae production plays in fueling Colorado River food webs, we identified gross primary production as a natural process metric.

Aquatic gross primary production (hereafter GPP) is a measure of the total amount of oxygen produced via photosynthesis in the river (typically in units of g O_2 m^{-2} d^{-1}). Recent developments in both oxygen sensor technology and modeling methods support estimation of GPP rates at the daily time step (Appling and others, 2018; Bernhardt and others, 2018). GPP is expected to be a good metric of aquatic food availability in Marble Canyon and Grand Canyon where 1) algae photosynthesis represents the foundation of aquatic food webs in these segments, and 2) there are not large standing stocks of unpalatable primary producers, such as the macrophytes that dominate river-bottom habitats in Glen Canyon and other vascular plants that can contribute to overall GPP in a river segment (Rüegg and others, 2021). Recent analysis shows that GPP is positively related to the growth of Flannelmouth Sucker in Marble Canyon and Grand Canyon (Hansen and others, 2023) and is expected to be positively related to the growth of other native fishes.

Here we present a time series of springtime GPP data from three long-term sites between 2012 and 2024 (Figure 2.3). Methods for estimating GPP in the Grand Canyon are described in Deemer and others (2022a) and the data used for calculating the 2012-2019 GPP metrics are in Deemer and others (2022b). At the time of publication, data from 2020-2024 were provisional data and subject to change.

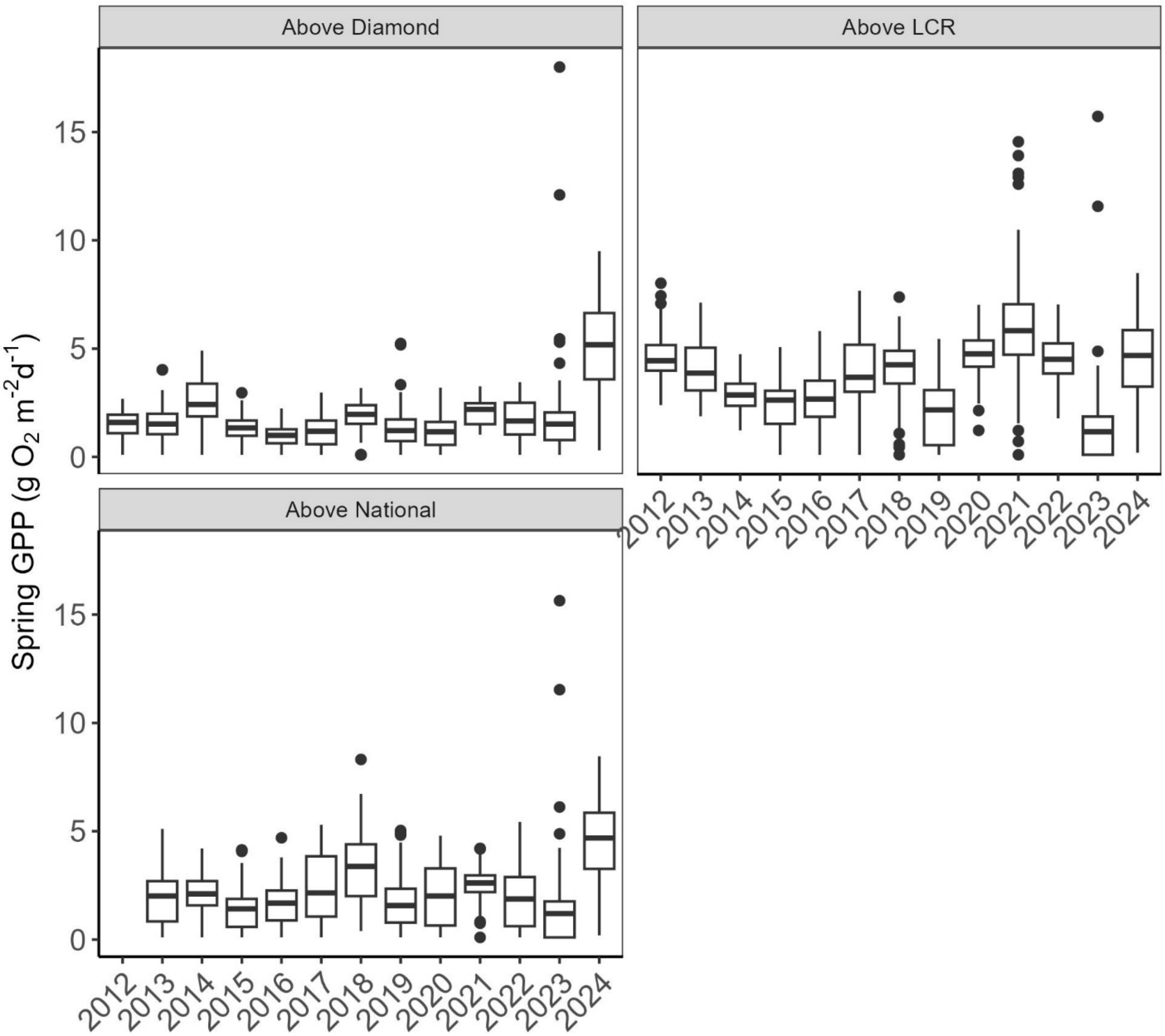

Figure 2.3. Springtime (March, April, May and June) gross primary production (GPP) across three reaches in the Grand Canyon. The "Above Diamond" reach encompasses the 27 river kilometers upstream from the Colorado River above Diamond Creek near Peach Springs, AZ, 09404200 U.S. Geological Survey gage. The "Above National" reach encompasses the 34 river kilometers upstream from the Colorado River above National Canyon near Supai, AZ, 09404120 U.S. Geological Survey gage. The "Above LCR" reach encompasses the 36 river kilometers upstream from the Colorado River Above Little Colorado River, AZ, 09383100 U.S. Geological Survey gage. Data through 2019 are published in Deemer and others (2022b). At the time of publication data from 2020 to 2024 were provisional and subject to change.

Metric 2.4: Percent EPT

The species of aquatic plants and animals present in the CRe have shifted toward species that are more resilient to regulated rivers. Aquatic macroinvertebrate assemblages were likely diverse in the pre-dam river and included mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera), "EPT" hereafter (Vinson, 2001; Haden and others, 2003; Kennedy and others, 2016). Invertebrate abundance was likely seasonally variable and greatest during long periods of low flows (Minckley, 1991; Haden and others, 2003; also see Woodbury and others, 1959). In the post-dam environment, the invertebrate assemblage has low diversity and is dominated by open-water egg-laying insects (Kennedy and others, 2016). This change in composition reflects river processes and has implications for the myriad of animal species that feed on aquatic insects as a primary food source (Nakano and Murakami, 2001; Baxter and others, 2005). The proportion of the invertebrate community comprised of EPT taxa (Percent EPT) has been used to describe the ability of a river to support life and is used internationally as a metric of stream and river health (Carlisle and others, 2013). Theory and empirical studies show that food webs that are biodiverse are more stable and more resilient to disturbance than simplified food webs, thus have greater ecological integrity (Rooney and others, 2006; Rooney and McCann, 2012). Therefore, we identified the proportion of the invertebrate community comprised of EPT taxa as a metric evaluating the biotic response to natural processes.

We compute the Percent EPT metric using community science light trap monitoring of aquatic insects (refer to Kennedy and others, 2016 for light trap monitoring methods). The Percent EPT metric represents the number of EPT taxa collected in a sample divided by the total number of aquatic insects present in the sample. In 2024, 37 percent of the aquatic insects captured in community science light trap samples on average were EPT taxa, representing the highest value for the EPT metric in the 13-year period of record (Figure 2.4). This high Percent EPT value for 2024 was driven by a relatively high absolute abundance of EPT taxa (i.e., caddisfly abundance in 2024 was the third highest in the 13-year record) and a relatively low absolute abundance of non-EPT insects (i.e., midge abundance in 2024 was the third lowest in the 13-year record).

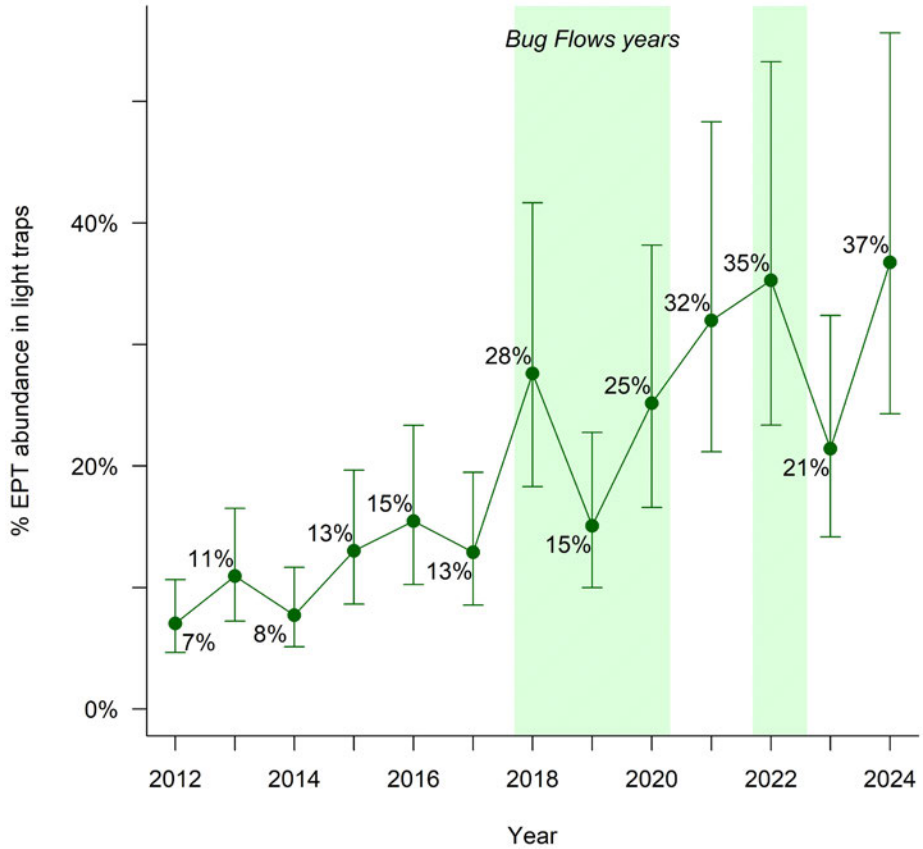

Figure 2.4. Graph showing the mean percentage of Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies; collectively EPT) taxa in the Colorado River ecosystem (CRe) captured in community science light trap sampling by year relative to the total number of aquatic insects captured. Bars represent one standard error, and green bands represent years when the bug flow experiment was tested. It should be noted that this plot largely represents percent Trichoptera captured in light traps—a total of 52 mayflies (Ephemeroptera) and zero stoneflies (Plecoptera) have been captured in light traps between 2012 and 2024. Data from 2012 through 2021 are published in Kennedy and others, 2023. At the time of publication, data from 2022 to 2024 were provisional and subject to change.

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LTEMP Resource Goal 3. Humpback Chub

"Meet Humpback Chub recovery goals, including maintaining a self-sustaining population, spawning habitat, and aggregations in the Colorado River and its tributaries below the Glen Canyon Dam" (U.S. Department of the Interior [DOI] (2016a).

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Background

Humpback Chub (*Gila cypha*) is a federally listed fish species endemic to the Colorado River basin. Of the six recognized Humpback Chub populations, Grand Canyon (including mainstem and tributary components) is home to the largest population (U.S. Fish and Wildlife Service [USFWS], 2002, 2018). Monitoring metrics for Humpback Chub were chosen to represent the status of the historic stronghold for the species in Grand Canyon (i.e., the Little Colorado River-spawning component; Metric 3.1) as well as to reflect the recent population increase and range expansion within the mainstem and other tributaries (Metrics 3.2, 3.3).

The tributary components include the Little Colorado River (LCR) and Havasu Creek, though other tributaries (e.g., Shinumo Creek, Bright Angel Creek) have potential to play more important roles in the future if Humpback Chub are able to establish. For the decades following construction of Glen Canyon Dam, Humpback Chub that spawned in the LCR were the most numerous group within Grand Canyon (and thereby across the entire species, as the Grand Canyon population had the highest abundance of the six recognized populations) (Kaeding and Zimmerman, 1983; USFWS, 2002; Coggins and Walters, 2009). The LCR-spawning component includes both LCR residents, which reside year-round in the LCR, and LCR migrants, which move from the mainstem in eastern Grand Canyon into the LCR to spawn in spring (Yackulic and others, 2014; Dzul and others, 2021a, c). Humpback Chub from the LCR were translocated into other tributaries and started successfully reproducing in Havasu Creek in 2013 (Healy and others, 2020) after translocations commenced in 2011, increasing the redundancy and resiliency of the species (Persons and others, 2017; Van Haverbeke and others, 2017; Rogowski and others, 2018).

In contrast to the tributary component in the LCR where Humpback Chub were more common, Humpback Chub were rare in the mainstem when water temperatures decreased following construction of Glen Canyon Dam (~1970s-1990s; Valdez and Ryel, 1997). Furthermore, spatial distribution of Humpback Chub in the mainstem was patchy, so that Humpback Chub were only captured in disjunct, localized sites with little movement between sites (hereafter 'aggregations sites'; Valdez and Ryel, 1997). In western Grand Canyon (i.e., from Havasu Creek to Pearce Ferry), catch at aggregation sites started to increase slightly in the late-2000s (Persons and others, 2017) and by 2016, adult Humpback Chub were abundant and widespread throughout western Grand Canyon (Van Haverbeke and others, 2017; Rogowski and others, 2018). Currently, preliminary data suggest high abundance of adult Humpback Chub in western Grand Canyon. It is hypothesized that this increase in western Grand Canyon Humpback Chub occurred due to warming water temperatures (Van Haverbeke and others, 2017) associated with drought-induced declines in Lake Powell elevations (Dibble and others, 2021) and formation of Pearce Ferry rapid in 2007 (which formed a physical barrier between Lake Mead and the Colorado River in the Grand Canyon).

Metric 3.1: Current Tier of Humpback Chub in the LCR Aggregation as described in the 2016 LTEMP Biological Opinion

As the most established group of Humpback Chub in Grand Canyon, the population dynamics of LCR-spawning Humpback Chub (i.e., LCR aggregation) are relatively well understood and provide important baseline data that can be used to compare against future conditions. To help protect and conserve LCR-spawners, the 2016 Biological Opinion (BiOp; U.S. Department of the Interior (DOI), 2016b) was established to define target minimums for adult Humpback Chub abundances (i.e., tiers) and to link tiers to management actions. The population model used to estimate abundances of LCR-spawners and determine the current tier is a multistate mark-recapture model that includes states for size class and location (i.e., LCR, JCM-east sampling site, in the mainstem outside JCM-east sampling site, JCM="Juvenile Chub Monitoring"; Dzul and others, 2022) so that state transitions correspond to growth and(or) movement between rivers (Figure 3.1). For details on abundance estimation methods, readers should refer to Yackulic and others (2014) and Dzul and others (2023).

By providing the tier instead of the abundance estimates, this metric provides context as to whether the LCR-spawning adults (≥ 200 mm total length [TL]) are in good health (Tier 0), show signs of possible future decline (Tier 1), or have low abundance (Tier 2). Specifically, the 2016 BiOp identifies 9,000 as the minimum number of adults where management actions are not needed (Tier 0), 7,000-8,999 as the number of adults where management actions excluding trout removals (e.g., increased translocations) are warranted (Tier 1), and <7,000 as the number where removal of nonnative trout is required (Tier 2; DOI, 2016b).

Additionally, assessment of Tier 1 includes large subadults (150-199 mm TL) to determine whether subadult abundances are on track to offset the adult mortality associated with a target adult abundance of 9,000.

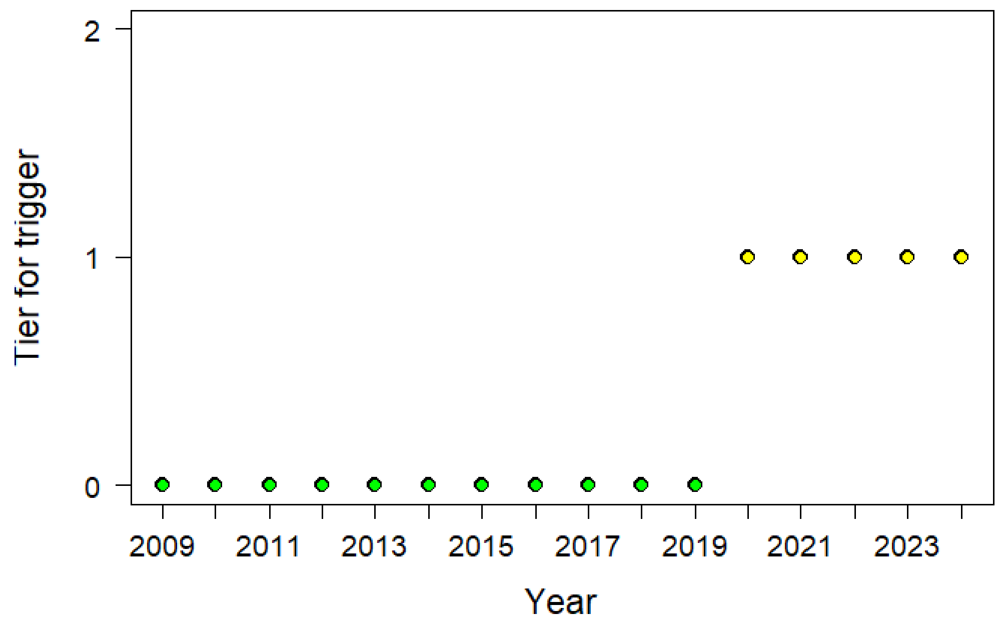

Figure 3.1. Tier for Little Colorado River-spawning Humpback Chub, as defined in the 2016 Biological Opinion (DOI, 2016b) from 2009-2024. Data from Dzul and others, 2021b.

Metric 3.1 shows the LCR-spawning group to be in good health (Tier 0) until 2019, at which point the abundances of large subadults drops to put LCR-spawners in Tier 1 from 2020-2024, which indicates that adult Humpback Chub abundance may fall below 9,000 in the future. Specifically, since 2020, the rolling 3-year average of large subadult abundance in the JCM-east sampling reach during fall has been below 810 to suggest recruitment is not sufficient to maintain adult abundance of 9,000. However, adult abundances have remained above 9,000 during the last 5 years, likely due to adult abundances having a high starting point (much greater than 9,000) and the effects of warming water temperatures on accelerating Humpback Chub growth in the Colorado River (Dzul and others, 2023). Specifically, the numbers for the trigger calculation were based on data from cold water years and assumed that it took three years for large subadult Humpback Chub to grow to adult size. More recently (i.e., post-2020), warmer water temperatures mean subadults may take 1-2 years to grow to adult size, which leads to a lower standing stock of subadults. High numbers of juvenile Humpback Chub in the mainstem in 2022 and 2023 suggest that future increases to subadults and adults are likely under current conditions (i.e., low predation levels).

Metric 3.2: Grand Canyon-wide Abundance of Adult Humpback Chub

Metric 3.2 (DOI, 2016b) compiles abundance estimates of adult Humpback Chub across different spatial locations within Grand Canyon to provide a simple measure of overall abundance for the entire region. Currently, this abundance estimate is the sum of the two most numerically dominant groups (LCR-spawning and western Grand Canyon). Metric 3.2 is the sum adult abundance estimates from a mark-recapture model of LCR spawners (same as used in Metric 3.1; Dzul and others, 2022), estimates from a state-space closed population model of western Grand Canyon, and estimates from major tributaries (e.g., Havasu Creek), assuming all abundance estimates are independent. While this metric can be intuitive due to its simplicity, we caution that it may be too coarse to provide information about site-specific abundances and trends. In actuality, Metric 3.2 is most reflective of sites with high abundances (in this case, the western Grand Canyon) so that trends in other sites with lower abundances (e.g., Havasu Creek) may be missed. Metric 3.2 is calculated every five years. Other publications suggest large increases in catch of Humpback Chub in western Grand Canyon over the last 1-2 decades (Van Haverbeke and others, 2017; Rogowski and others, 2018). Preliminary data suggest Metric 3.2 could reflect these increases to evaluate if the Humpback Chub resource in the Grand Canyon is currently considered in good condition.

Metric 3.3: Proportion of the Grand Canyon Ecosystem with Evidence of all Three Life Stages of Humpback Chub

Metric 3.3 (DOI, 2016b) describes how Humpback Chub are distributed spatially within Grand Canyon at different life stages (juvenile [<100 mm TL], subadult [100-199 mm TL], and adult [>200 mm TL]) (Tables 3.1 and 3.2). Presence of all three life stages provides an indication of a healthy, self-sustaining population. Juveniles are an important indicator of successful reproduction and absence of juveniles provides an early warning of a (likely) future decline in adults. Accordingly, monitoring presence/absence of juveniles across the river network in Grand Canyon can provide a broad spatial overview of whether or not the Humpback Chub population is self-sustaining and potentially identify hotspots where the population may be expanding or contracting. Having subadults and adults present, in addition to juveniles, could provide further evidence that the population is self-sustaining in a particular stream segment.

We note that Metric 3.3 is sensitive to sampling methods, so that changes in effort and gear types used to sample river sections may affect the metric. For example, juvenile Humpback Chub can be more readily captured in seines and unbaited hoop nets compared to adult Humpback Chub. We therefore note some caution in interpretation of segment-to-segment level changes, particularly if effort and gears vary across years and segments. Instead, this metric is intended to evaluate large, dramatic changes over the short term and more gradual changes over longer terms.

Table 3.1. List of habitat segments (defined by 12-13 river kilometer (rkm) sections starting at Glen Canyon Dam) and the number of life stages of Humpback Chub (*Gila cypha*) detected by sampling efforts in 2024. Life stages of Humpback Chub are defined by total length (TL), with life stages corresponding to juveniles (<100 mm TL), subadults (100-199 mm TL), and adults (>199 mm TL). Score is binary, with score=1 reflecting all stages present. Provisional data, subject to change.

Metric 3.3: Proportion of the Grand Canyon Ecosystem with Evidence of all 3 Life Stages of Humpback Chub				
Species	**River**	**Habitat Segment (rkm from dam)**	**Life Stages Detected**	**Score**
HBC	mainstem	0-13	0	0
HBC	mainstem	13-25	0	0
HBC	mainstem	25-37	0	0
HBC	mainstem	37-49	0	0
HBC	mainstem	49-61	1	0
HBC	mainstem	61-73	0	0
HBC	mainstem	73-85	2	0
HBC	mainstem	85-97	3	1
HBC	mainstem	97-109	1	0
HBC	mainstem	109-121	1	0
HBC	mainstem	121-133	3	1
HBC	mainstem	133-145	3	1
HBC	mainstem	145-157	3	1
HBC	mainstem	157-169	2	0
HBC	mainstem	169-181	3	1
HBC	mainstem	181-193	1	0
HBC	mainstem	193-205	2	0
HBC	mainstem	205-217	3	1
HBC	mainstem	217-229	3	1
HBC	mainstem	229-241	3	1
HBC	mainstem	241-252	3	1
HBC	mainstem	252-265	2	0
HBC	mainstem	265-277	3	1
HBC	mainstem	277-289	3	1
HBC	mainstem	289-301	3	1
HBC	mainstem	301-313	3	1
HBC	mainstem	313-325	3	1
HBC	mainstem	325-337	3	1
HBC	mainstem	337-349	3	1
HBC	mainstem	349-361	3	1
HBC	mainstem	361-385	3	1
HBC	mainstem	385-397	3	1
HBC	mainstem	397-409	3	1
HBC	mainstem	409-421	3	1
HBC	mainstem	421-433	3	1
HBC	mainstem	433-445	2	0
HBC	mainstem	445-457	3	1
HBC	mainstem	457-469	2	0
HBC	mainstem	469-481	3	1
Final Score: Proportion of Habitat Segments with all 3 Life Stages in Mainstem				**0.62**

Table 3.2. List of habitat segments (defined as river kilometers [rkm] upstream of the confluence with the Colorado River) and the number of life stages of Humpback Chub (HBC; *Gila cypha*) detected in major tributaries of Grand Canyon during sampling efforts in 2024. Life stages of Humpback Chub are defined by total length (TL), with life stages corresponding to juveniles (<100 mm TL), subadults (100-199 mm TL), and adults (>199 mm TL). All scores refer to habitat segments in calendar year 2024. The final score indicates the proportion of habitat segments where all three life stages of Humpback Chub were detected. Provisional data, subject to change.

Species	River	Habitat Segment (rkm from confluence)	Life Stages Detected	Score	Final Score
HBC	Little Colorado River	0-6.9	3	1	
HBC	Little Colorado River	7-13.6	3	1	
HBC	Little Colorado River	13.7-17.8	3	1	1.0
HBC	Kanab Creek	0-1	1	0	0.0
HBC	Shinumo Creek	0-1	0	0	0.0
HBC	Havasu Creek	0-1	3	1	1.0
HBC	Bright Angel Creek	0-2.9	0	0	
HBC	Bright Angel Creek	3-7.3	1	0	
HBC	Bright Angel Creek	7.4-10.3	0	0	0.0

To calculate Metric 3.3, habitat segments where all three life stages were sampled were given a score of 1, otherwise the segment received a score of 0. The final metric was obtained by summing scores across all segments and converting to a proportion by dividing by the total number of segments. Metric 3.3 indicates that in the mainstem Colorado River, from Glen Canyon Dam to Pearce Ferry, 62 percent of habitat segments have evidence of all three life stages of Humpback Chub in 2024. The occurrence of all three life stages is absent in Glen Canyon (rkm 0-25) and patchy in rkm 26-121 (i.e., mostly Marble Canyon) and increases downstream of rkm 121 (just upstream of confluence with the Little Colorado River in Grand Canyon, which is located ~rkm 124). Additionally, Humpback Chub were detected in all three habitat segments in the Little Colorado River, as well as in Havasu Creek, but not in Shinumo Creek. Only one life stage was detected in Bright Angel Creek (in one of three reaches studied) and in Kanab Creek. Incorporating more years into this metric can help document past and future range contractions and expansions for Humpback Chub.

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All animal procedures were reviewed and approved by the U.S. Geological Survey's Southwest Biological Science Center Institutional Animal Care and Use Committee (Project ID USGS-SBSC-2024-05, USGS-SBSC-2025-01).

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LTEMP Resource Goal 4. Hydropower and Energy

"Maintain or increase Glen Canyon Dam electric energy generation, load following capability, and ramp rate capability, and minimize emissions and costs to the greatest extent practicable, consistent with improvement and long-term sustainability of downstream resources" (U.S. Department of the Interior [DOI], 2016).

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Background

Hydropower generation within the Colorado River Storage Project (CRSP; 43 U.S.C. 620 et seq.) is a vital resource for the electricity sector in the Western Interconnect (western United States power grid). Hydropower facilities are highly flexible energy sources with low variable costs and are considered a relatively clean form of energy (Waldo and others, 2021). Beyond providing ancillary services, the primary role of CRSP energy—particularly at Glen Canyon Dam—is to replace more expensive energy generation during periods of peak demand. This strategy can maximize the economic value of hydropower, which can be represented as the difference between the variable cost of energy replaced by hydropower and the variable costs of hydropower itself. From an economic perspective, the timing of energy generation is more critical than the total amount of energy produced.

The ability to optimize the economic value of hydropower depends on operational constraints (Figure 4.1). When the operation of a hydropower facility deviates from the optimal economic dispatch or when additional constraints are imposed to achieve other resource goals, there is a corresponding loss in economic value. Furthermore, if a facility's energy-generating capacity is insufficient during periods when capacity requirements are critical, additional capacity is planned within the system, which can lead to increased economic costs. Additionally, external costs, such as emissions (CO_2, CH_4, SO_2, NO_X) within the electric sector, are influenced by the generation of hydropower. These externalities are also of consideration when evaluating the overall economic impacts of hydropower operations.

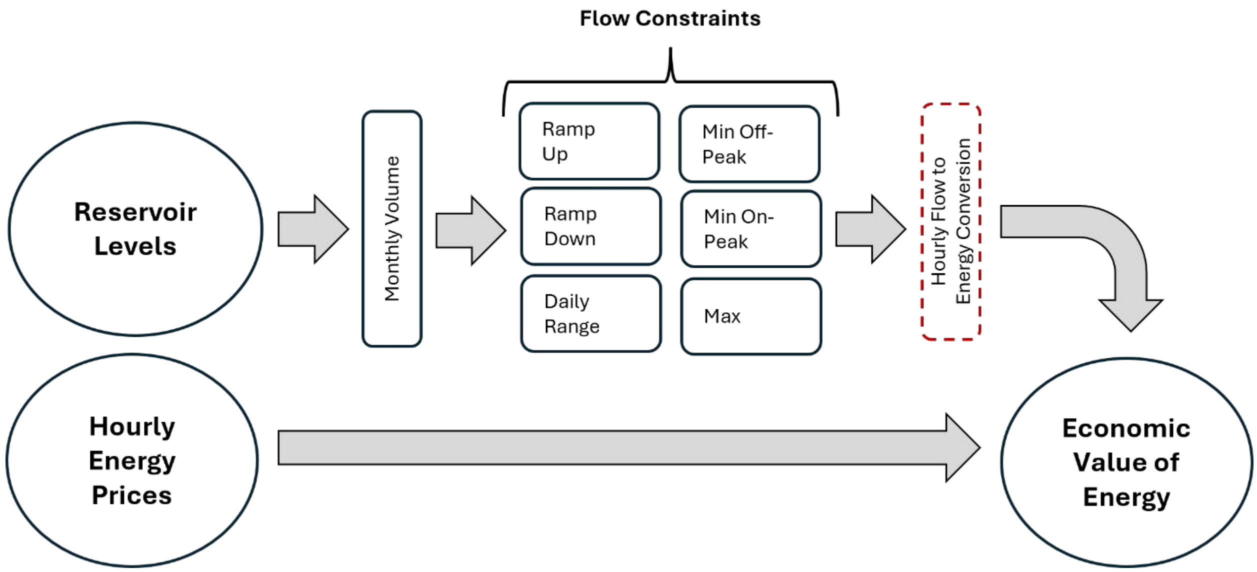

Figure 4.1. Conceptual model of drivers affecting the economic value of energy generation at Glen Canyon Dam. The economic value of energy generated at Glen Canyon Dam is constrained by drivers that are independent of within-year dam operations (reservoir levels, hourly prices, monthly releases) and drivers that are dependent on within-year dam operations (daily flow constraints).

Metric 4.1: Economic Value of Hydropower

In this retrospective analysis, energy generation and prices are observed over the period of interest. The economic value of hydropower is calculated as the difference between the marginal cost of energy generation during a specific period and the marginal cost of hydropower. For example, if 10 MWh of hydropower replaces 10 MWh of natural gas-fired energy generation, the economic value of hydropower equals the marginal cost of natural gas generation (primarily fuel costs) minus the marginal cost of hydropower generation (primarily maintenance and operational costs). Hydropower generation data for Glen Canyon Dam are reported by the U.S. Bureau of Reclamation (USBR, 2025). These data account for reservoir levels, hourly flow volumes, and water-to-power conversion calculations when estimating generation. Hourly electricity prices are sourced from the California Independent System Operator (CISO, 2025).

For illustrative purposes, Figure 4.2 presents the hourly energy generation at Glen Canyon Dam and the corresponding energy price at the Palo Verde Hub in Arizona during the week of March 1–7, 2024. The economic value of hydropower during this representative week is calculated by multiplying the hourly energy generation at Glen Canyon Dam (MWh) by the locational marginal price of energy in the system ($/MWh), then subtracting the marginal cost of hydropower generation, estimated at approximately $6/MWh (Power and others, 2016). By applying this methodology throughout 2024, it is possible to estimate the total value of energy generated at Glen Canyon Dam over the year.

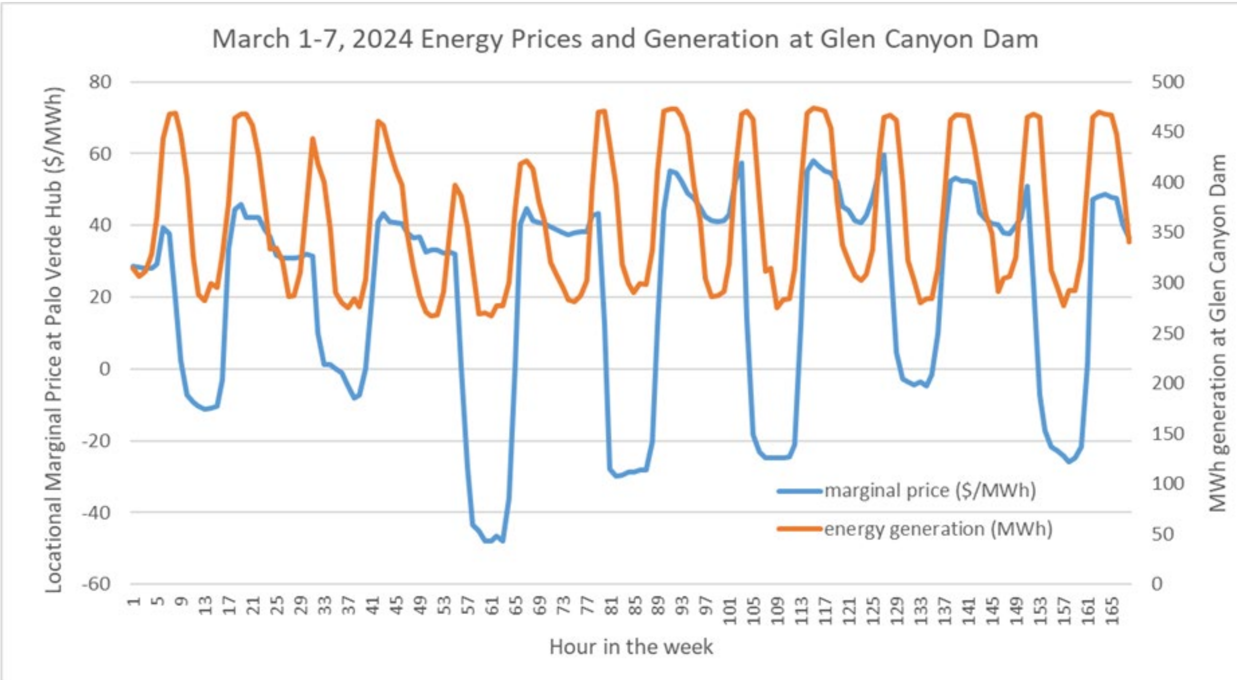

Figure 4.2. Hourly energy generation at Glen Canyon Dam (USBR, 2025) and the locational marginal price (CISO, 2025) of energy at the Palo Verde Hub for March 1–7, 2024. Energy prices exhibit a bimodal pattern over the 24-hour cycle, highlighting the significant influence of renewable energy, in this example solar energy, on hourly locational marginal prices during this time of year. Energy generation at Glen Canyon Dam closely follows the fluctuations in locational marginal prices. Provisional data, subject to change.

Energy prices and generation levels vary significantly throughout the year. For instance, both generation and locational marginal prices in August 2024 differ substantially from those in March (Figure 4.3). During this period, electricity sector demand is high, resulting in locational marginal prices exceeding $400/MWh. This week also coincides with an experimental release of water intended to cool river temperatures and prevent the establishment of nonnative warmwater fish species. The simultaneous operation of both generating units and the bypass at Glen Canyon Dam leads to considerable fluctuations in hydropower generation.

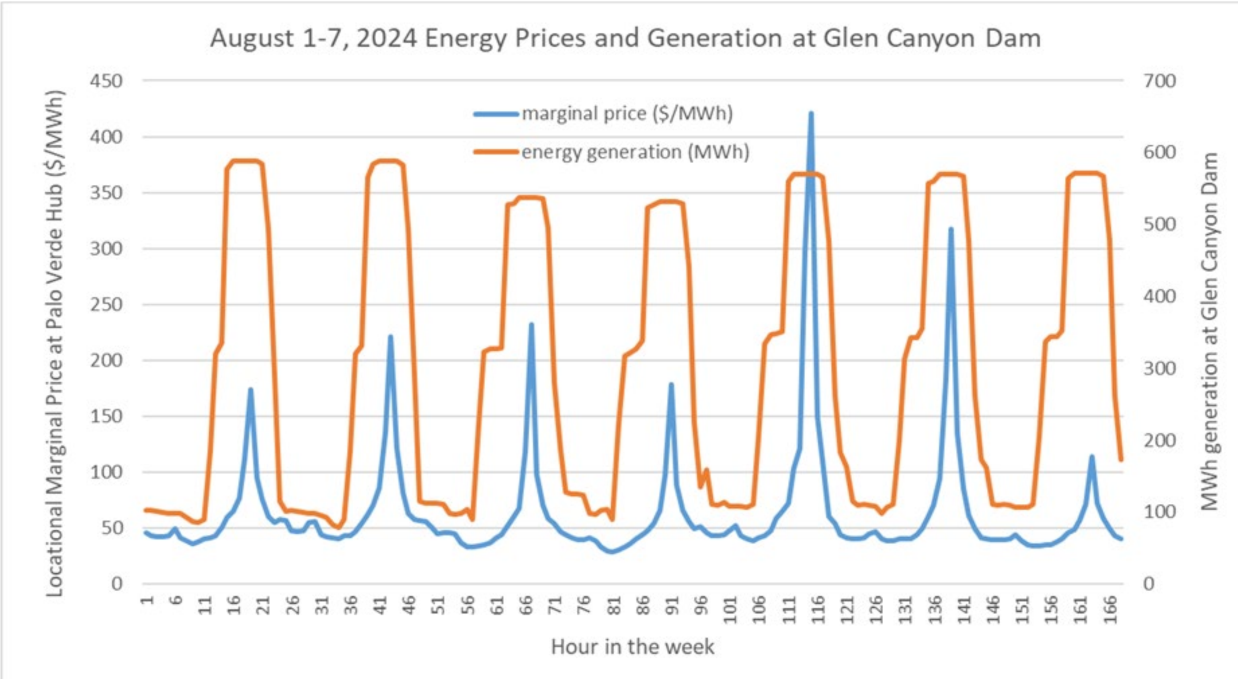

Figure 4.3. Hourly energy generation at Glen Canyon Dam (USBR, 2025) and locational marginal price of energy (CISO, 2025) at the Palo Verde Hub for August 1–7, 2024. Prices exhibit single daily peaks, in one instance reaching upwards of $400/MWh, driven by high energy demand during this time of year. Energy generation at Glen Canyon Dam aligns with the fluctuations in locational marginal prices. Provisional data, subject to change.

By applying the methods described above to the data shown in Figures 4.2 and 4.3 for the 2024 calendar year, the economic value of energy generated at Glen Canyon Dam can be estimated. The economic value of this energy demonstrates a pronounced seasonal variation (Figure 4.4). Energy prices are significantly higher during the winter and summer months, when increased demand can necessitate the use of more expensive generating units to meet energy needs.

Based on this analysis, the estimated total economic value of energy generated at Glen Canyon Dam in 2024 is $81 million.

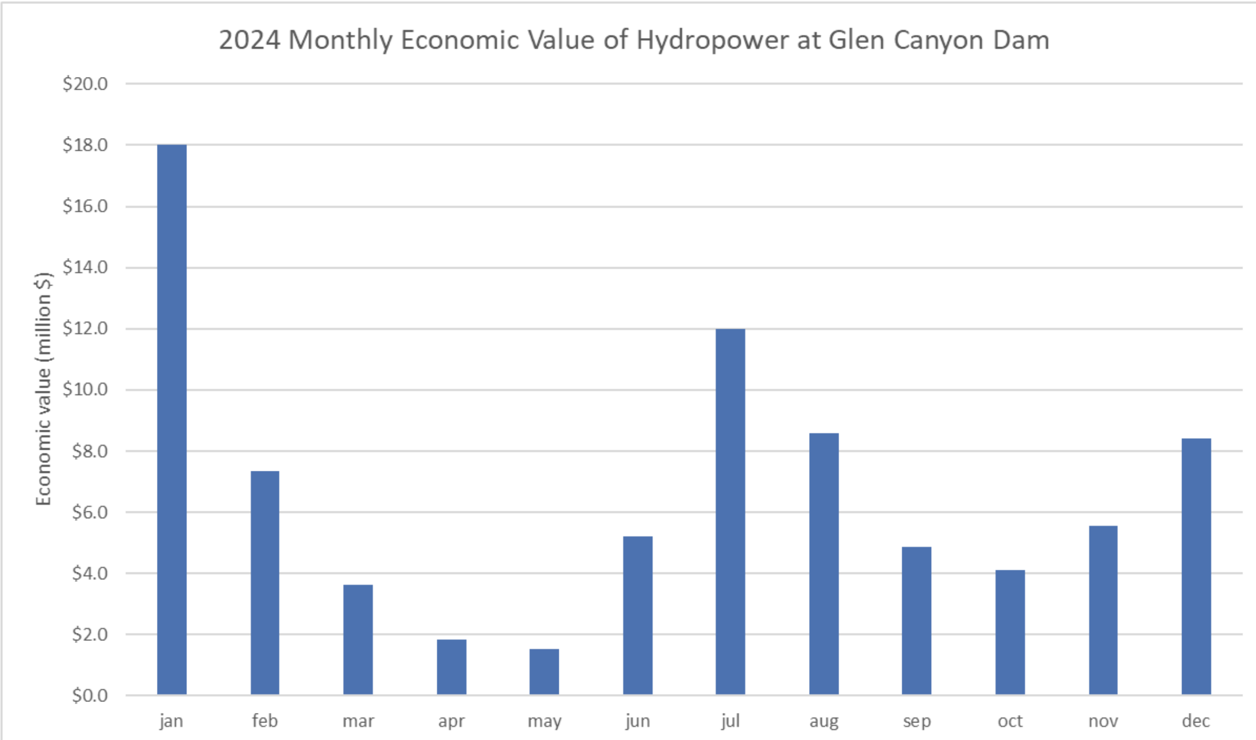

Figure 4.4. Monthly economic value of energy generated at Glen Canyon Dam in 2024 (data from USBR, 2025 and CISO, 2025). Provisional data, subject to change.

The avoided capacity and emissions costs are not estimated in this report, as doing so would require access to production cost models like the proprietary models PLEXOS or AURORAxmp. These models simulate the operation of the electricity sector and could estimate how the operation of Glen Canyon Dam impacts annual costs related to capacity and emissions.

In future metric exercises, changes in the economic value of hydropower could be assessed on a monthly and yearly basis. Such assessments could provide insights for identifying trends within and across years due to changes in the electricity sector, rather than simply tracking overall hydropower generation trends resulting from reservoir elevation or monthly volume changes.

Reporting of metrics to characterize deviations from operational rules are outlined in the U.S. Department of the Interior (DOI) Long-term Experimental Management Plan Environmental Impact Statement (LTEMP EIS; DOI, 2016). For instance, during the summer of 2024, experimental flows were implemented to reduce Colorado River water temperature. This experiment involved moving significant volumes of water through bypass, foregoing hydropower generation. The resulting loss of energy generation had a substantial impact on the economic value of energy produced at Glen Canyon Dam in 2024 based on this analysis. Future operational decisions that affect the timing and quantity of energy generation, available capacity, and emissions in the electricity sector could be documented and included in metric reporting.

This approach can improve understanding of the broader implications of operational decisions on the Hydropower and Energy Goal, as specified in the beginning of this chapter (DOI, 2016).

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LTEMP Resource Goal 5. Other Native Fishes

"Maintain self-sustaining native fish species populations and their habitats in their natural ranges on the Colorado River and its tributaries" (U.S. Department of the Interior [DOI], 2016).

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Background

Colorado River native fishes evolved in an environment characterized by seasonally and annually varying river discharge, temperature, and turbidity, resulting in unique morphological and life history traits as well as high endemism. This natural environmental variability has been dampened and altered by the construction and operations of Glen Canyon Dam (GCD), leading to declines in native fishes in the Grand Canyon (Gloss and Coggins, 2005). Nonetheless, recent drought (2000-present; Udall and Overpeck 2017; Wheeler and others, 2022) and decreasing basin-wide reservoir water storage have facilitated increases in seasonal variability of water temperature that coincided with expansion of native fish populations (Dzul and others, 2023). Further, somatic growth in native fishes has benefited, in part, from enhanced gross primary production stemming from short-term stable flow experiments (i.e., "bug flows"; Hansen and others, 2023). The distribution and abundance of native fishes in the Colorado River ecosystem (CRe) from GCD to Pearce Ferry rapids, and including tributaries, may also be driven by biotic interactions with introduced fishes (Yackulic and others, 2018; Healy and others, 2020) whose populations can also be influenced by Glen Canyon Dam operations as well as basin-wide water management (Korman and others, 2021; Yard and others, 2023). Thus, the extant native fishes in the CRe including Humpback Chub (*Gila cypha*; addressed using different metrics [Fairley, 2023]), Bluehead Sucker (*Pantosteus discobolus)*, Flannelmouth Sucker (*Catostomus latipinnis*), Razorback Sucker (*Xyrauchen texanus*), and Speckled Dace (*Rhinichthys osculus)*, are important indicators of ecosystem health, and may reflect the effects of dam operations and experimental flows, in addition to factors driven by climatic variability. The occurrence of species previously extirpated in the CRe (Bonytail *Gila elegans*, Colorado Pikeminnow *Ptychocheilus lucius,* Roundtail Chub *Gila robusta*) also contributes towards the "other native fishes" goal, reflects the efforts of management agencies to reintroduce these species, and are of concern to the Glen Canyon Dam Adaptive Management Program (GCDAMP; DOI, 2016).

A complete analysis of the five “other native fish” metrics is not available. These metrics include the following:

Metric 5.1 – Proportion of the Grand Canyon ecosystem with evidence of all three life history stages of bluehead sucker (juvenile [1-year lag], subadult, adult).

Metric 5.2 – Proportion of the Grand Canyon ecosystem with evidence of all three life history stages of flannelmouth sucker (juvenile [1-year lag], subadult, adult).

Metric 5.3 – Proportion of the Grand Canyon ecosystem with evidence of all three life history stages of razorback sucker (juvenile [1-year lag], subadult, adult).

Metric 5.4 – Proportion of the Grand Canyon ecosystem with evidence of speckled dace (any life stage).

Metric 5.5 – Proportion of the Grand Canyon ecosystem with evidence of extirpated species (bonytail, roundtail chub, Colorado pikeminnow; any life stage).

Native fish capture data, which are used to calculate metrics, can benefit the additional assessment of these metrics through collaborative monitoring and research efforts by the Arizona Game and Fish Department, National Park Service, and U.S. Geological Survey in multiple tributaries as well as the Colorado River.

All animal procedures were reviewed and approved by the U.S. Geological Survey’s Southwest Biological Science Center Institutional Animal Care and Use Committee (Project ID USGS-SBSC-2024-02, USGS-SBSC-2024-05, USGS-SBSC-2025-01).

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LTEMP Resource Goal 6. Recreational Experience

> *"Maintain and improve the quality of recreational experiences for the users of the Colorado River ecosystem. Recreation includes, but is not limited to, flatwater and whitewater boating, river corridor camping, and angling in Glen Canyon" (U.S. Department of the Interior [DOI], 2016).*

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Background

Recreational experiences in Glen and Grand Canyons are renowned worldwide (U.S. Department of the Interior [DOI], 2016). These regions host a wide range of outdoor recreational activities, with two of the most notable being trout fishing below Glen Canyon Dam (GCD) at Lees Ferry and whitewater rafting in the Grand Canyon. The quality of these activities—and others within the canyons—is influenced by a variety of attributes. These attributes include factors directly affected by GCD operations (e.g., river flow), those indirectly influenced by GCD operations (e.g., Rainbow Trout [*Oncorhynchus mykiss*] abundance, campsite area and quality), and those unrelated to GCD operations (e.g., scenic vistas and geology) (Figure 6.1).

Defining metrics for evaluating recreational experience involves identifying and measuring attributes connected to GCD operations. Ideally, these metrics are quantitative, commensurable (allowing aggregation), and reflective of the experiences of diverse recreational groups.

An individual's economic value for outdoor recreation is based on the combination of attributes that make an outdoor activity desirable. Using the economic value of recreational experience as a metric enables the simultaneous evaluation of all key attributes and their given state during an outdoor recreational trip —such as river flow, beach area that can be used for camping, campsite quality, and Rainbow Trout catch rates—while providing the flexibility to aggregate this information across various trips and outdoor recreation activities. Specifying the appropriate suite of metrics and their interactions can benefit the analysis because recreationists have varying preferences for different attributes, which often conflict. For example, higher river flows may reduce beach areas that can be used for camping by submerging them more frequently.

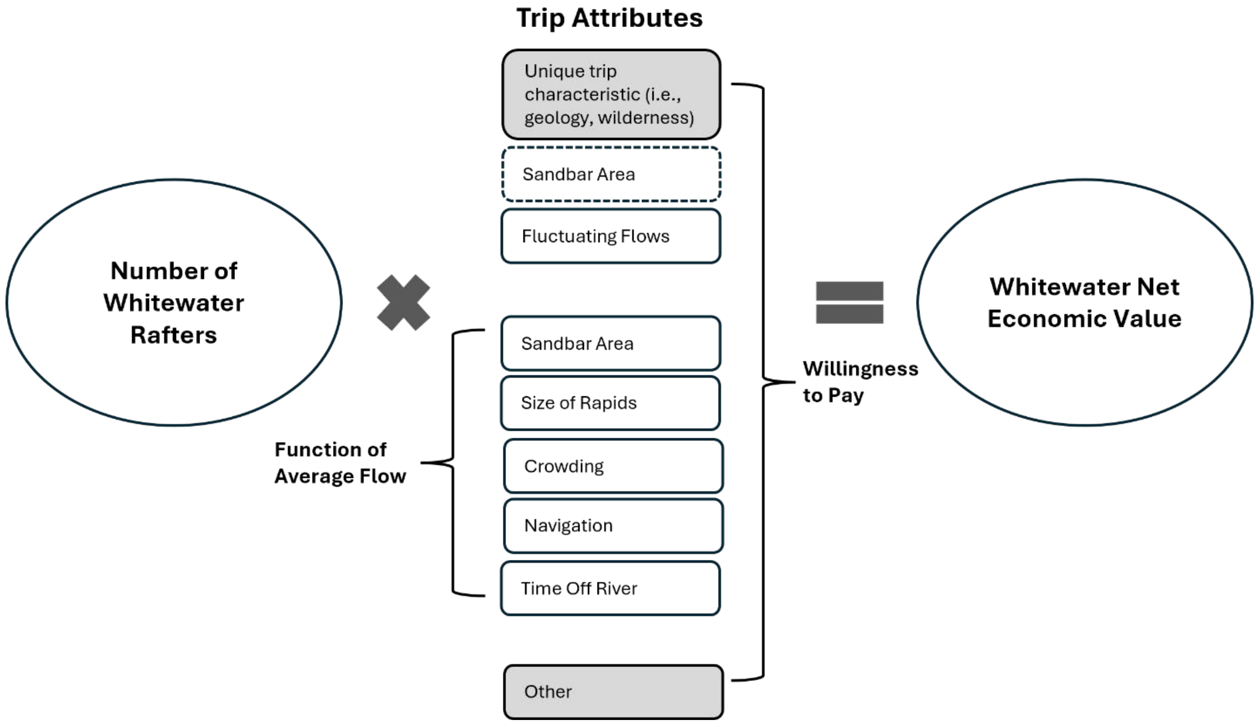

Figure 6.1. Conceptual model of drivers influencing the recreational experience or economic value of whitewater rafting in Grand Canyon National Park. Economic value is determined by the annual number of whitewater rafters and the individual economic value of a trip that is influenced by attributes that depend on dam operations (e.g., sandbar area, daily flow fluctuation, and average daily flow) as well as attributes independent of dam operations. Dashed lines for the sandbar attribute indicate that its importance is not accounted for in the existing function used to estimate the economic value of recreational experience of whitewater rafting trips.

Economists have developed methods to estimate the economic value of a nonmarket activity, such as outdoor recreation (e.g., Neher and others, 2019). Since recreational activities in Glen and Grand Canyons are not market-traded, survey methods are used to estimate economic value based on variations in attributes. Estimating this value requires understanding individuals' willingness to pay for trips under specified conditions.

This value reflects recreationists' willingness to pay beyond their trip costs, offering a measure of trip value while accounting for the role of various attributes. It is important to distinguish this from spending by recreationists, which impacts the local economy. Willingness to pay represents the economic value to a recreationist but is not the cost of an individual's trip. Because of this, changes in GCD operations may affect the recreational experience and its economic value without influencing regional spending, assuming the number of trips remains constant.

The economic value of recreation incorporates the inherent trade-offs recreationists face between attributes during their trips. These trade-offs have been shown to remain stable over time (Neher and others, 2019).

While not all aspects of recreation in Glen and Grand Canyons can be fully captured through measured attributes that result in an economic value of an individual trip, this approach serves as a foundational framework for understanding the condition of the recreational resource goal (Neher and others, 2019). This framework can be used to evaluate experimental flows at GCD, or other management actions, and assess their impact on recreational experiences.

Metric 6.1: Economic Value of Recreational Experience

The recreational experience metric is modeled using willingness to pay (value above and beyond individual trip costs) from survey data that estimate the economic value of recreational trips under specific Colorado River flow and other conditions (Bishop and others, 1987; Bair and others, 2016; Neher and others, 2019). The surveys can be used to estimate functions that map Colorado River flow conditions to economic value of angling trips in Glen Canyon National Recreation Area and whitewater rafting trips in Grand Canyon National Park. The final metrics are aggregate economic value estimates based on observed numbers of participants and per trip economic values based on trip attributes, such as Colorado River flow influenced by the operation of GCD, over the period of interest.

While the economic value of recreation arises from various attributes, including average river flow, daily flow fluctuations, beach area that can be used for camping, Rainbow Trout catch rates, and trout size, in this initial stage of metric development, river flow is the sole attribute used to estimate the economic value of recreational experiences for anglers in Glen Canyon and whitewater rafters in the Grand Canyon (Bishop and others, 1987; Neher and others, 2019).

After determining monthly Colorado River flows (in cfs), the number of angler trips (Figure 6.2), and whitewater trips for 2024 were estimated (Figure 6.3). Seasonal patterns in recreation can influence analyses (e.g., timing of flows and angler trips) for both Glen Canyon National Recreation Area and Grand Canyon National Park. In summary, separate functions estimate the economic value of an individual angler or whitewater trip within a given month based on flow conditions. The total monthly economic value is then calculated by multiplying the per-trip economic value by the number of participants in each recreational activity (Figures 6.2 and 6.3).

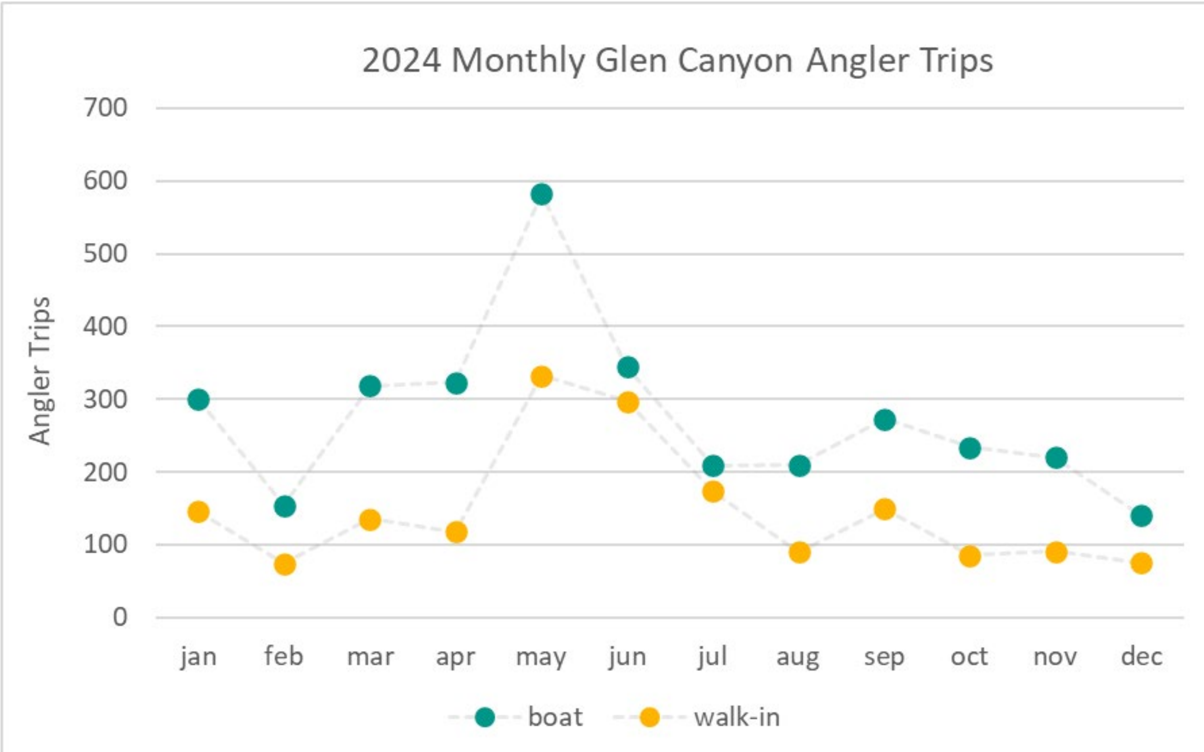

Figure 6.2. Estimated 2024 monthly angler trips to Lees Ferry in Glen Canyon National Recreation Area. Given that flow characteristics and angler experience vary by month, identifying monthly trip patterns is used to analyze the economic value of angling at Lees Ferry. Provisional data, subject to change.

The analysis, aggregating the per-trip economic values based on flow characteristics multiplied by monthly trips, provides an estimate of the economic value of angling and whitewater recreation in 2024 as a proxy for recreational experience. It is important to note that recreational whitewater trips require permits. While analyses of future Glen Canyon Dam operations often assume that all permitted trips will be filled, this is not always the case. In 2024, fewer trips were taken than the number of available permits (Figure 6.3). Therefore, the observed number of individual whitewater rafting trips are used to estimate the total economic value for whitewater rafting in 2024.

Using the methods described above, the estimated economic values for whitewater rafting and angling in 2024 were:

- **Whitewater trips**: $28.0 million in economic value (Figure 6.3).
- **Angling trips**: $1.8 million in economic value (Figure 6.4).

Together, these result in an estimated total economic value of $39.8 million for recreational angling and whitewater trips in 2024.

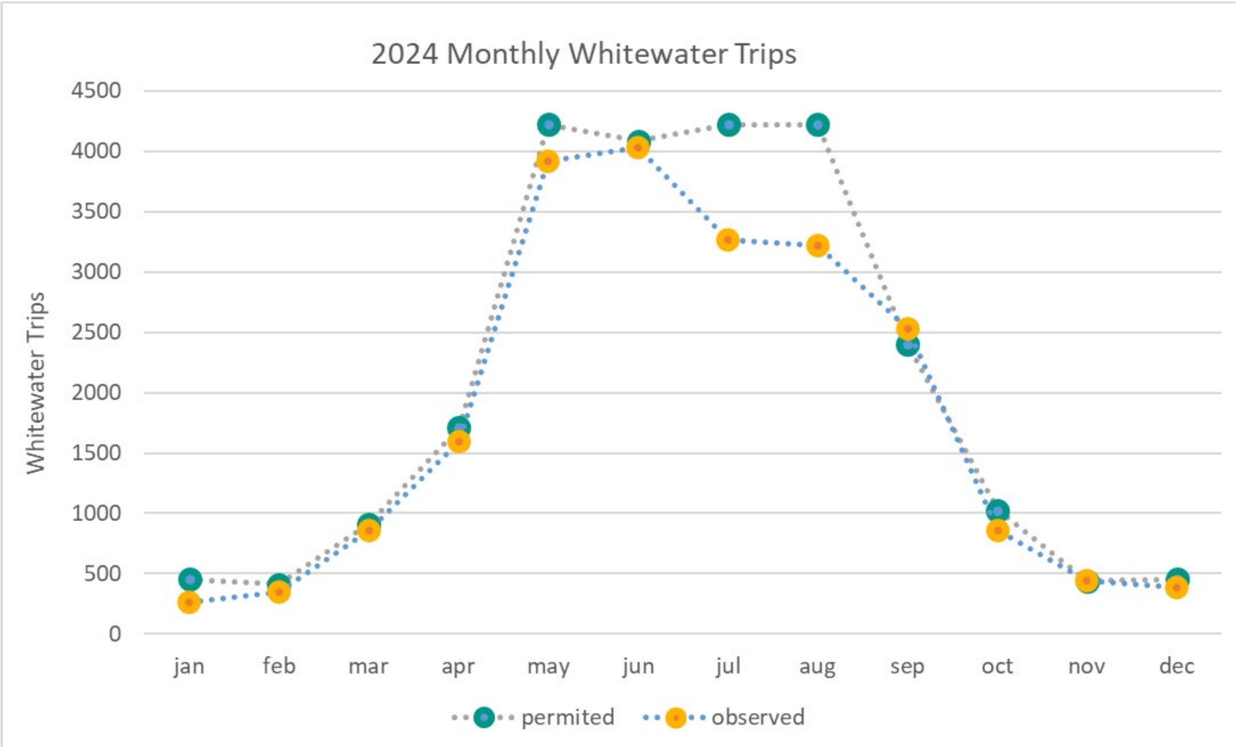

Figure 6.3. Estimated 2024 monthly whitewater trips in Grand Canyon National Park compared to permitted whitewater trips. Provisional data, subject to change.

In addition to the above metrics, the recreational experience goal metric can benefit from refinement. For example,

1. **Incorporating Additional Attributes**: Future estimates of recreational experience could include attributes such as Rainbow Trout catch rates for angling trips or the area of beach used for camping to improve accuracy of recreational experience estimates for whitewater trips.
2. **Baseline Comparisons**: Economic value estimates could be compared to a baseline condition that accounts for visitor numbers and other factors unrelated to GCD operations. This comparison isolates the impacts of GCD operations. Attributes such as beach area that can be used for camping or Rainbow Trout catch rates may require multi-year evaluations, but controlling for other factors can help to assess the long-term impacts of GCD operations.
3. **Historic Trends**: Incorporating historic data into metric results could enable trend analyses of changes in recreational experiences over time.
4. **Addressing Uncertainties**: While recreationists' preferences for attributes such as average river flow can be known, less is understood about how changes in beach area that can be used for camping or campsite quality may affect economic value. Existing

data—both socioeconomic and physical science—can help refine the relationships between key attributes and economic value.

5. **Broader Economic Context**: Broader economic trends in the United States, such as income changes and shifting preferences for goods and services, could influence the economic value of recreational experiences. Although studies (Bishop and others, 1987; Neher and others, 2017) suggest that these values have remained stable over time, future estimates may benefit from considering evolving economic conditions.

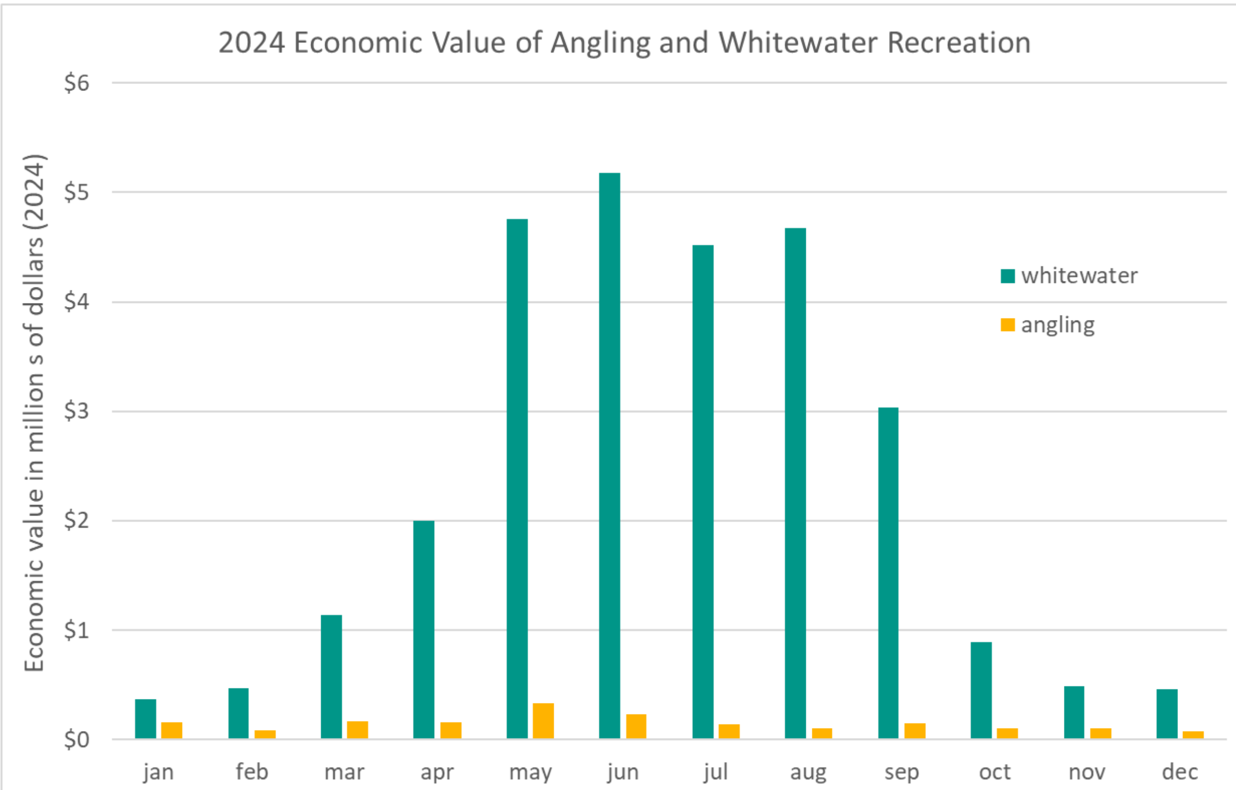

Figure 6.4. Monthly economic value of recreational angling and whitewater trips in 2024. Provisional data, subject to change.

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LTEMP Resource Goal 7. Sediment

"Increase and retain fine sediment volume, area, and distribution in the Glen, Marble, and Grand Canyon reaches above the elevation of the average base flow for ecological, cultural, and recreational purposes" (U.S. Department of the Interior [DOI], 2016).

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Background

The management goal for fine-sediment (sand, silt, and clay) in the Colorado River ecosystem (CRe) is based on utilizing dam operations and only the available tributary fine-sediment supply (i.e., no sediment augmentation) and may be achieved when the fine-sediment mass balance is neutral to positive in the Colorado River on a segment-by-segment basis over multi-year to decadal-plus timescales (DOI, 2016). Most of the fine sediment stored in the CRe is composed of sand-size sediment. Despite the erosion of over 28 million metric tons of sand from the CRe since the 1963 closure of Glen Canyon Dam, the amount of sand in the CRe still greatly exceeds the amount of sand annually resupplied by tributaries (Hazel and others, 2006; Topping and others, 2021). In contrast, the amount of silt and clay in the CRe is likely much less than that annually resupplied by tributaries. Although relatively large amounts of silt and clay have been observed seasonally in Colorado River eddies following tributary floods, the amount retained over multi-year periods in the CRe is generally negligible (Topping and others, 2021).

Management of sand is focused on maintaining/increasing the size of sandbars whereas management of silt and clay is focused on maintaining sufficient quantities of silt and clay for ecological processes (DOI, 2016). Sandbars erode under most dam operations and can only be rebuilt during high-flow experiments (HFEs) timed after tributary floods to coincide with the availability of the largest amount of finer sand (Rubin and others, 2002; Grams and others, 2015; Grams and others, 2025); sandbars have eroded when HFEs occurred when the sand was relatively coarse/depleted (Topping and others, 2019). The supply (that is, "bank account") of the finer sand utilized to rebuild/enlarge sandbars is a combination of newly supplied tributary sand and background sand in longer-term storage, some of which is relict from the pre-dam period (Barnhardt and others, 2001; Chapman and others, 2020). Sufficient sand may not be available to rebuild sandbars during HFEs in perpetuity if progressive depletion of this bank account continues (Topping and others, 2021), an especially important point given that some of this bank account predates Glen Canyon Dam (Barnhardt and others, 2001; Chapman and others, 2020) and thus cannot be readily replaced.

Most of the sand bank account is localized and stored at lower elevations in the river cross section (Grams and others, 2013, 2019) and is therefore continuously affected by dam operations, whereas the amount of sand stored in sandbars at higher elevation is smaller and less affected by dam operations (Schmidt and others, 2004; Hazel and others, 2006; Grams and others, 2019).

Achievement of the Long-Term Experimental and Management Plan (LTEMP) Resource Goal 7 (DOI, 2016) requires a sustainable source of sediment. Dam operations have resulted in net depletion of the sand bank account since 1963 (Topping and others, 2021), and sandbar deposition during HFEs depends on the amount of finer sand in this bank account (tributary resupply plus background storage). Therefore, the metric to evaluate performance for the sediment goal includes measures of the sand bank account and sandbar response. Measuring sandbar response alone does not inform on the sustainability of sand management because sandbar deposition during repeated HFEs could progressively deplete the sand bank account (Schmidt and Grams, 2011). Moreover, because high-elevation sandbars are less sensitive to dam operations than is the lower-elevation sand that comprises most of the bank account, this depletion could continue for some time before being detected in the sandbar data.

Because the amount of silt and clay stored in the CRe is much smaller than the amount of silt and clay annually resupplied by tributaries, the metric to evaluate silt and clay is different than that for sand. Instead of focusing on a measure of the amount of silt and clay in storage, an improved metric evaluates the effect of dam operations on the retention time of silt and clay following tributary resupply.

Metrics for evaluating the performance of LTEMP Goal 7 (DOI, 2016) can therefore track 1) changes in the sand bank account, that is, the sand supply to sandbars, 2) changes in high-elevation sandbar size, and 3) retention of silt and clay relative to ecologically critical thresholds. Although useful for predicting basic sandbar response for the various flow alternatives explored during the LTEMP process, the sand-load index (SLI) of Runge and others (2015) may not provide an effective evaluation of LTEMP Goal 7 performance. The Runge and others (2015) SLI can be used to evaluate these flow alternatives when this index is greater than 0 only when HFEs occur, and HFEs are required to rebuild sandbars. However, the SLI can only attain its maximum value of 1 when all sand transport during a year occurs during HFEs. Absent flows exceeding 31,500 ft^3/s continuously, the SLI would equal 1 only under the extremely coarse sand conditions where sand can only be transported during the highest flows, that is, when the sand bank account has been exhausted and sandbars are relatively small (Topping and others, 2021). Consequently, although a Runge and others (2015) SLI greater than 0 can indicate sediment management achieves LTEMP Goal 7, larger values of the SLI approaching 1 may indicate that this goal is not achieved.

The following metrics represent 1) the segment-by-segment change in the sand supply (bank account) for maintaining and rebuilding sandbars in the CRe, 2) the volume, area, and distribution of fine sediment in high-elevation sandbars in Glen, Marble and Grand Canyons along with a second metric to evaluate sandbar response to individual high flow events, and 3) the segment-by-segment duration of time when the silt-and-clay concentration causes turbidity to exceed 30 formazin nephelometric units (FNUs), which corresponds to a critical ecological threshold. The first metric addresses the sand supply, the second two address high-elevation sandbars, while the last measurement concerns silt-and-clay retention.

Goal 7, Performance Metric 7.1: Sand Supply by River Segment (the Sand "Bank Account")

Metric Description and Criteria for Selection

The metric chosen to evaluate whether LTEMP sand management can be sustainable is the segment-by-segment direct measurement of the sand supply, that is, the sand "bank account," utilized to rebuild sandbars during HFEs. This sand bank account consists of a mixture of sand of different ages ranging from pre-dam sand to recent tributary sand inputs (Barnhardt and others, 2001; Chapman and others, 2020). This metric is calculated on an annual basis for five river segments: upper Marble Canyon, lower Marble Canyon, eastern Grand Canyon, east-central Grand Canyon, and west-central Grand Canyon. It is extracted from the continuous mass-balance sand budgets for these segments, which are in turn calculated using the 15-minute discharge and suspended-sediment measurements (Rantz and others, 1982a, b; Topping and Wright, 2016) made at the U.S. Geological Survey (USGS) gaging stations that bracket these segments (refer to list and map of gaging stations and segments [referred to as "reaches"] in Topping and others, 2021, or at https://www.gcmrc.gov/discharge_qw_sediment/).

The sand-supply metric reflects the total change in sand mass in a river segment relative to a "zero time"; uncertainties accumulate relative to this zero time. The zero time is chosen as midnight preceding July 1, 2012, the time preceding sediment year 2013 and the initiation of the HFE Protocol (DOI, 2011). As developed by Topping and others (2000) and used extensively in the LTEMP Final Environmental Impact Statement (FEIS; DOI, 2016) and in Topping and others (2021), a sediment year is defined such that each year begins with the season of maximum tributary sediment supply (July 1 of preceding calendar year) and ends with the natural season of maximum sediment export, that is, the pre-dam snowmelt flood (June 30 of current calendar year). Refer to Topping and others (2021) for a detailed description of this metric and the nature of the uncertainties associated with it. For sand management to be sustainable in the CRe, long-term trends in the sand-supply metric (7.1) must be either zero or positive in each river segment (Figure 7.1).

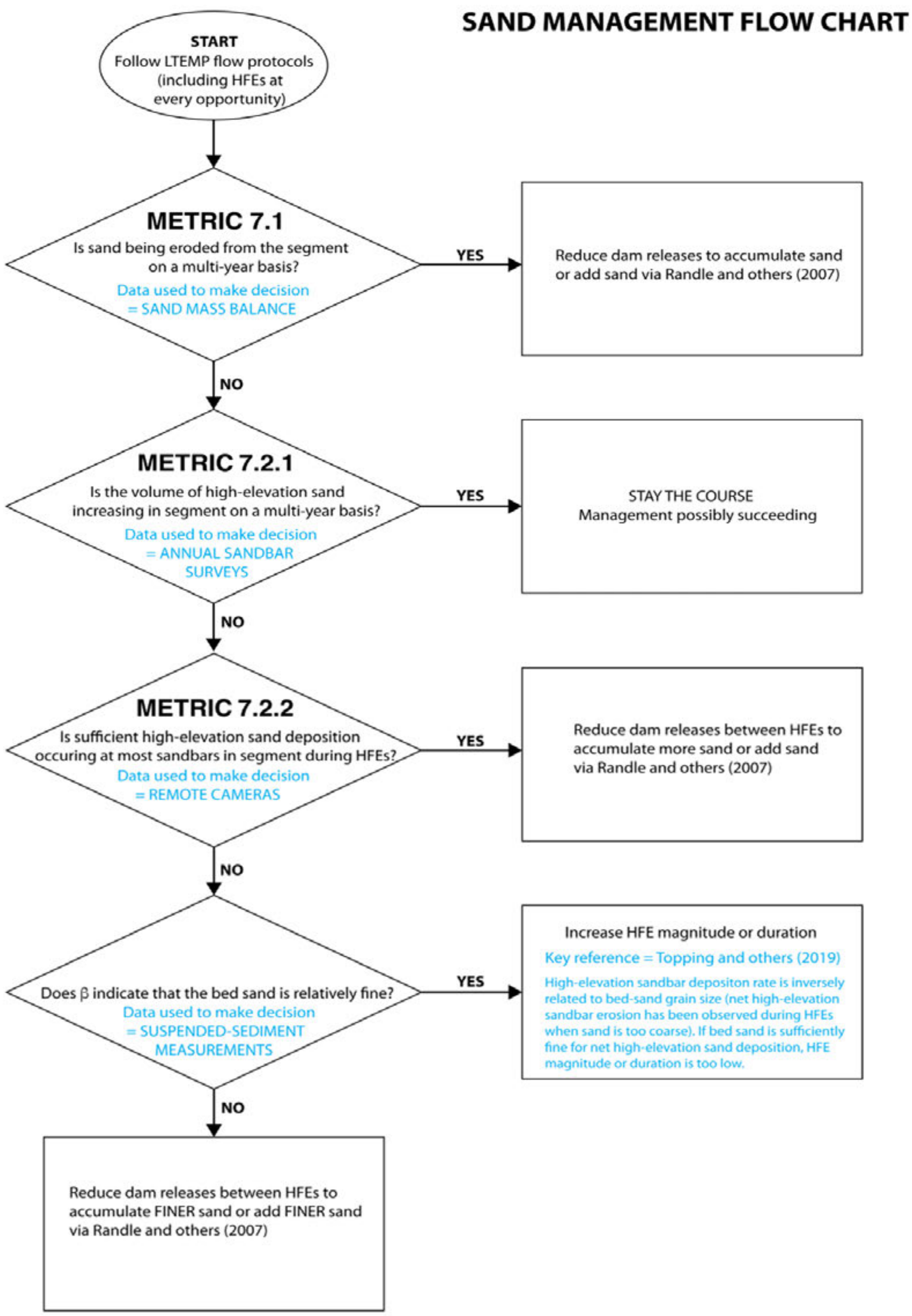
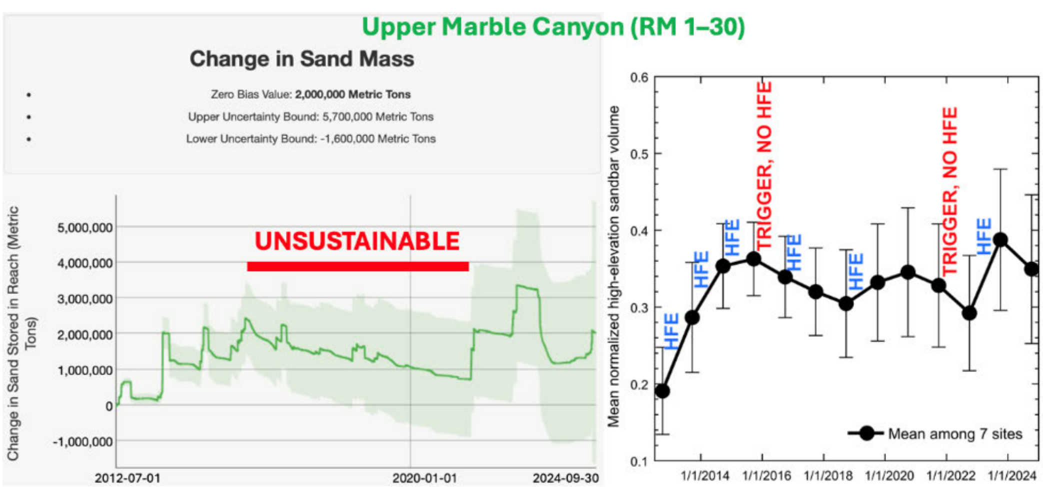


Figure 7.1. Sand management flow chart showing the decision points and management options that can be associated with Metrics 7.1, 7.2.1, and 7.2.2. β are defined in Rubin and Topping (2001, 2008) and applied to the Colorado River ecosystem (CRe) in Topping and others (2019, 2021). "Sufficient high-elevation sand deposition" under Metric 7.2.2 is defined as an amount of sand similar to that observed during previous high-flow experiments (HFEs) under similar sand-concentration and grain-size conditions.

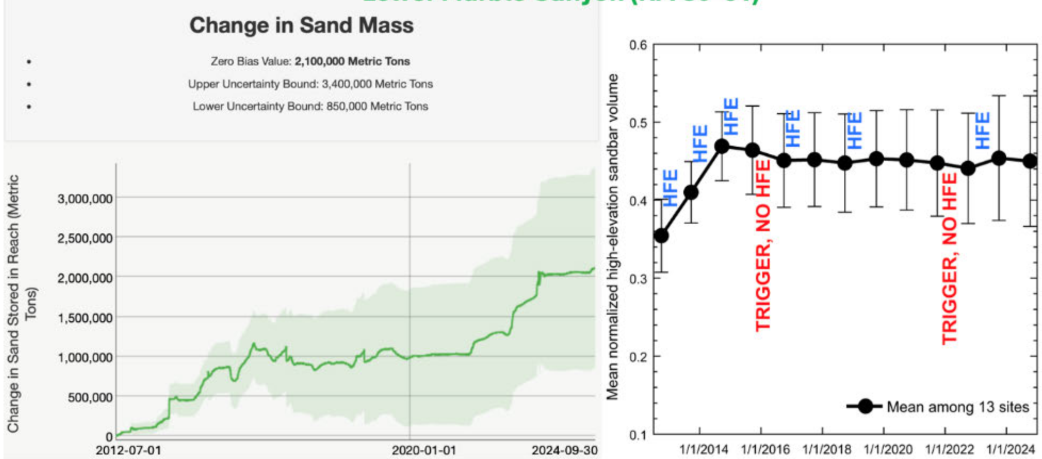
Regardless of the trends in sandbar volume (Metric 7.2.1), a sustained negative trend in Metric 7.1 indicates that LTEMP sand management to rebuild sandbars is not sustainable because the sand bank account supplying sand to these bars is being depleted (Schmidt and Grams, 2011). Figure 7.1 illustrates how the sand supply metric and the sandbar volume metrics are used together to evaluate the performance of dam operations under LTEMP with respect to sand management. For example, a multi-year negative trend in Metric 7.1 may indicate the need to consider a change in dam operations or sediment augmentation (Randle and others, 2007; Topping and others, 2021).

Status and Interpretation of Sand Supply by River Segment

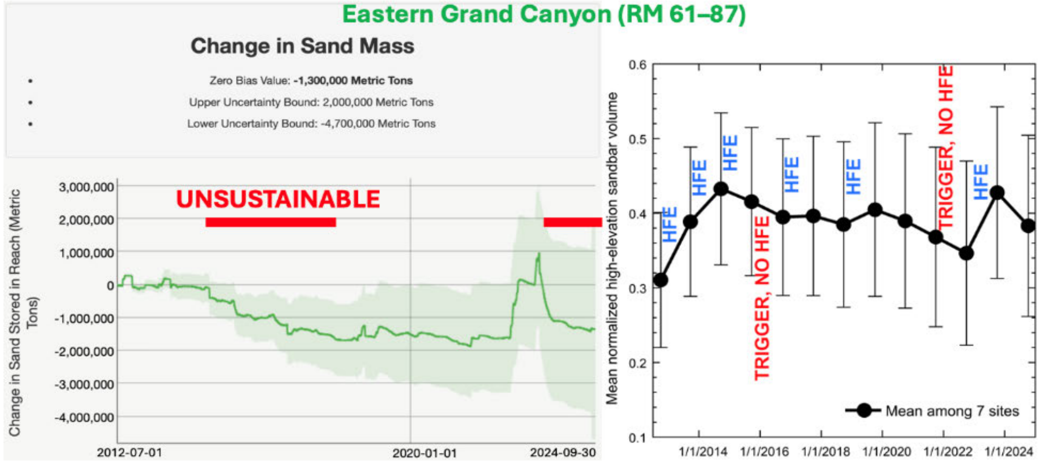
The status of Metric 7.1 paired with the mean normalized high-elevation (above the stage associated with a discharge of 25,000 ft^3/s) sandbar volume in each river segment for July 1, 2012, through August-September 2024 is shown in Figure 7.2. Mean normalized high-elevation sandbar volume was calculated using the data from Metric 7.2.1. The stage associated with a discharge of 25,000 ft^3/s was chosen to be above normal dam operations. Although the trends in Metric 7.1 are generally flat to positive, thereby indicating sustainable sand management, there were periods of unsustainable sand management (negative trends) in three of the five river segments: one multi-year period in Marble Canyon, two multi-year periods in eastern Grand Canyon, and one period in west-central Grand Canyon. Recalculation of the uncertainty envelopes relative to the "zero times" of each of these four periods indicate that each has a demonstrably negative sand budget. Dam operations during these periods of unsustainable sand management were too high relative to the sand supply, thereby depleting the sand back account.



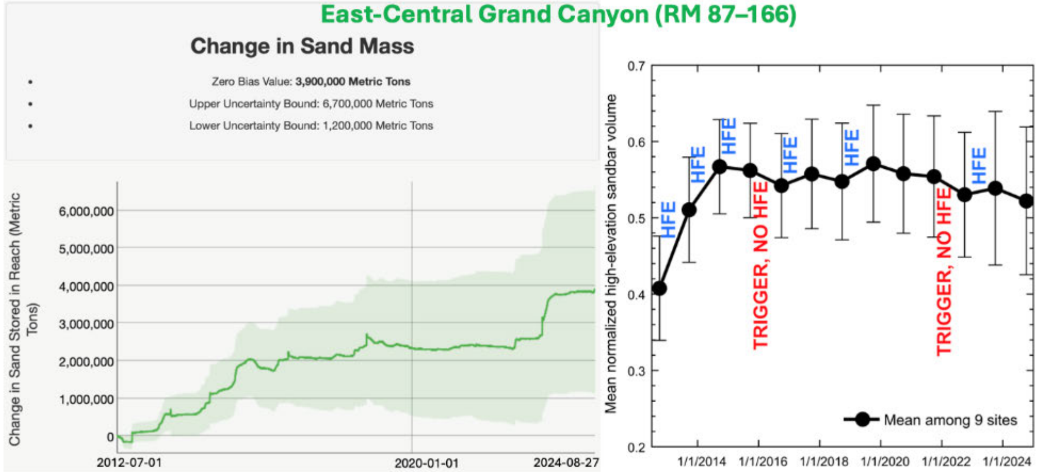
Lower Marble Canyon (RM 30–61)



Eastern Grand Canyon (RM 61–87)



East-Central Grand Canyon (RM 87–166)



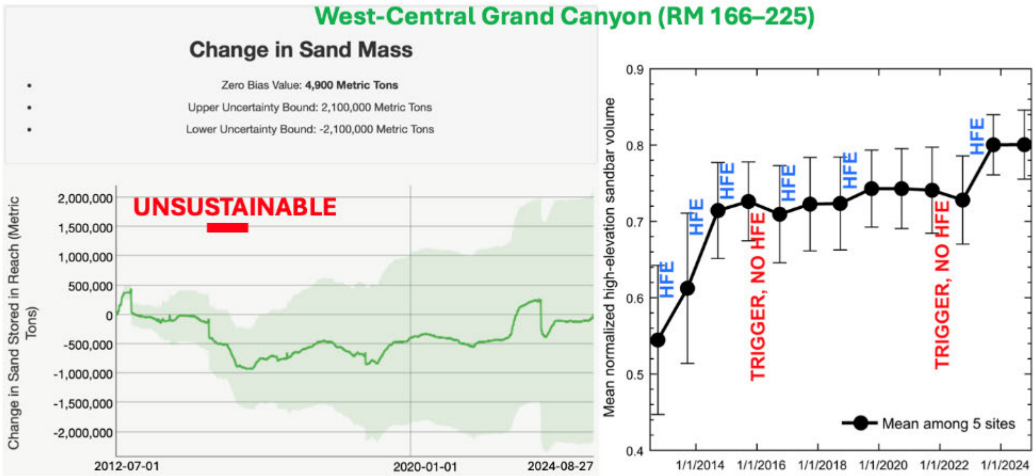


Figure 7.2. (previous pages) Metric 7.1 paired with the mean normalized high-elevation sandbar volume in each river segment. RM indicates river mile. Light green cloud shows the growing uncertainty envelope about the dark green line that depicts the zero-bias sand mass balance in each river segment. Annual or longer periods of unsustainable sand management in each river segment are indicated by the horizontal red lines. Error bars in the paired high-elevation sandbar time series are one standard error. The times of conducted high-flow experiments (HFEs) (blue) and triggered HFEs that were not conducted (red) are indicated in the sandbar time-series plots. In general, Metric 7.1 indicates sustainable sand management associated with multi-year cumulative gains in high-elevation sandbar volume (Metric 7.2.1) when HFEs are conducted every year. Longer gaps between HFEs largely offset any high-elevation sandbar gains resulting from HFEs due to the erosion that occurs during the extended period of intervening dam releases. Plotted data from U.S. Geological Survey (2025a, b).

Goal 7, Performance Metric 7.2: Sandbar Volume

The high-elevation sandbar metric is divided into two components that capture both the long-term changes in high-elevation sandbars (Metric 2.1) and the effectiveness of HFEs in causing sandbar deposition (Metric 2.2).

Goal 7, Performance Metric 7.2.1: Normalized Sandbar Volume

Metric Description and Criteria for Selection

Normalized sandbar volume is a metric for the volume of sand in sandbars above average base flow based on repeat topographic surveys. Measurements are collected annually at 45 long-term monitoring sites in Glen, Marble and Grand Canyons that represent a sample of approximately 5 to 10 percent of sandbars within eddies between Glen Canyon Dam and Diamond Creek (DOI, 2024; Project B.1; Grams and others, 2025). Although there is substantial variability among monitoring sites, response at the monitoring sites has been shown to be generally representative of the larger population of sandbars (Hazel and others, 2022). To verify that the annual measurements continue to be representative for the entire population of sandbars, measurements are also collected at a larger sample of high-elevation sandbars during periodic (every 5 to 10 years) channel mapping surveys (DOI, 2024; Project B.2) of 30- to 60-mile river segments (Grams and others, 2019). The topographic data are normalized to allow comparisons among sites of different size and averaging among different sample sizes (Hazel and others, 2022).

The metric is presented as an average among all sites and as averages among site type, river segment, and elevation zone. These segregations are important, because sandbar response varies significantly by site type and elevation zone (Mueller and others, 2018; Hazel and others, 2022). The most commonly used campsites typically belong to either the “Narrow and Medium Reattachment Bar” category or the “Separation and Undifferentiated Bar” categories of Mueller and others (2018). Sites in the “Wide Reattachment and Upper Pool Bar” category (Mueller and others, 2018) tend to be heavily vegetated and used less frequently as campsites. Because sandbar response also varies by elevation zone, this metric includes both the high-elevation zone (total volume above the 8,000 ft^3/s stage) and the controlled-flood zone (only the volume above the 25,000 ft^3/s stage). Sand in the controlled flood zone is most relevant for campsites because all sand in that zone is above the elevation that is inundated by normal dam operations. Although base flow magnitude is not defined in the LTEMP, this and other sandbar metrics use 8,000 ft^3/s as the base flow discharge (DOI, 2016).

Changes in sandbar volume have been shown to be consistent with changes in campsite area (Hazel and others, 2010; Hadley and others, 2018), but do not show changes caused by vegetation encroachment. This metric is specifically relevant to assessing the performance of the LTEMP with respect to the sediment related goal, however the LTEMP analysis did not use this metric for normalized sandbar volume, because a model for sandbar volume was not available at that time (DOI, 2016). A predictive model for normalized sandbar volume for reattachment bar types is now available (Mueller and others, 2021).

Status and Interpretation of Normalized Sandbar Volume, 1990 – 2023

For each survey of each monitoring site, the volume of sand is calculated in two zones. The fluctuating-flow zone is the zone above the 8,000 ft^3/s stage elevation and above a constant minimum surface up to the elevation inundated by 25,000 ft^3/s, which corresponds to the elevation inundated only by HFEs. The controlled-flood zone is the zone above the 25,000 ft^3/s stage elevation and above a constant minimum surface up to the maximum elevation inundated by HFEs. Volumes are normalized by a constant maximum volume for each site and each zone (Hazel and others, 2022). The normalized volumes are averaged among each site type and for each zone. Error bars are standard error among all sites included in each average. Because the same methods for data collection have been used since 1990 (Hazel and others, 2022), the metric can be computed for this entire period although values prior to 2008 use fewer monitoring sites and have larger uncertainty.

The time series of normalized sandbar volume for the controlled flood zone shows deposition by HFEs and erosion between HFEs (Figure 7.3a). The 2012, 2013, 2014 and 2023 HFEs resulted in deposition for all site types. For the entire period the HFE Protocol has been in place (DOI, 2011), 2012 to 2024, there was a significant net increase in the Narrow and Medium Reattachment Bars and the Wide Reattachment and Upper Pool Bars.

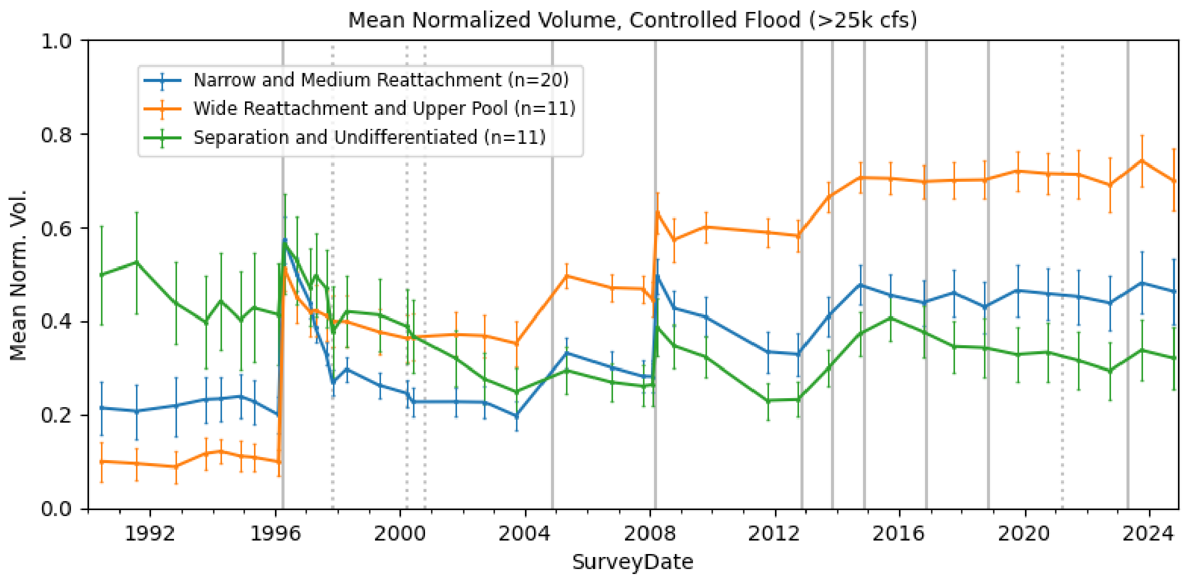


Figure 7.3a. Normalized sandbar volume in the Colorado River ecosystem (CRe) from June 1990 to October 2024, averaged for each site type (Mueller and others, 2018) in the controlled-flood zone, which only includes changes above the 25,000 ft^3/s stage. Methods and normalization procedure are described by Hazel and others (2022). Error bars are standard error. Vertical solid lines show the occurrence of high-flow experiments (HFEs) and the vertical dashed lines show the occurrence of short-duration releases at or below peak powerplant capacity. The "n" values are the number of sites in each category and has been consistent since 2008. Between 1990 and 2008, there were fewer sites in each category. Sites were measured immediately following HFEs only in 1996, 2004, and 2008. Beginning in 2009, measurements were made once annually in October. Data are available in Grams and others (2020) and U.S. Geological Survey (2025b). Cfs, cubic feet per second.

There was an increasing trend in sandbar volume in these bar types from 2016 to 2024, though the increase was not statistically significant. This metric shows that HFEs consistently cause increases in sandbar volume in the zone that is most relevant for campsites and in some of the bar types that are most commonly used as campsites but also underscores the importance of higher HFE frequency in maintained sandbar growth. Separation and Undifferentiated Bars experienced significant net increase in volume from 2012 to 2024. However, that time period is divided by significant increases occurring from 2012 to 2016 when HFEs were larger magnitude and conducted more frequently, and volume decreases from 2016 to 2024 when HFEs still occurred but less frequently.

For the entire monitoring period, 1990 to 2024, there was a significant net increase in the Narrow and Medium Reattachment Bars and the Wide Reattachment and Upper Pool Bars and a significant net decrease in the Separation and Undifferentiated Bars.

Figure 7.3b shows all sandbar changes above base flow, including the fluctuating zone and the controlled flood zone. Although the trends for the HFE Protocol period have been the same as for the controlled flood zone, the increases have been smaller because this zone includes lower elevation sand that has greater rates of erosion during normal fluctuating flows.

There was also a decreasing trend in the Separation and Undifferentiated Bars in the LTEMP period since 2016; the net decrease between 2016 and 2024 was statistically significant (greater than the standard error). In both the controlled flood zone and the entire high-elevation zone, there was a decreasing trend in all bar types from October 2019 to October 2022 and October 2023 to October 2024 when HFEs were not implemented. Considering all site types, zones, and river segments, the mean normalized sandbar volume has been stable or increasing since the start of the HFE Protocol in 2011 (DOI, 2011).

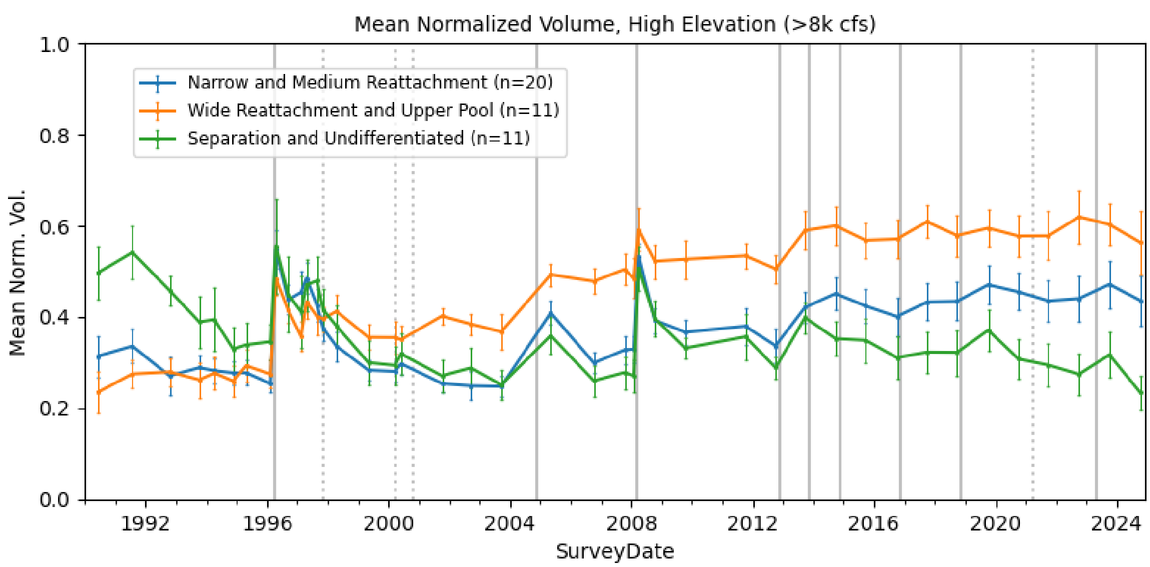


Figure 7.3b. Normalized sandbar volume in the Colorado River ecosystem (CRe) from June 1990 to October 2024, averaged for each site type (Mueller and others, 2018) in the high-elevation zone, which includes all sand volume above the 8,000 ft^3/s stage. Methods and normalization procedure are described by Hazel and others (2022). Error bars are standard error. Vertical solid lines show the occurrence of high-flow experiments (HFEs) and the vertical dashed lines show the occurrence of short-duration releases at or below peak powerplant capacity. The “n” values are the number of sites in each category and has been consistent since 2008. Between 1990 and 2008, there were fewer sites in each category. Sites were measured immediately following HFEs only in 1996, 2004, and 2008. Beginning in 2009, measurements were made once annually in October. Data are available in Grams and others (2020) and U.S. Geological Survey (2025b). Cfs, cubic feet per second.

Metric Limitations

The annual sandbar volume metric is based on 45 long-term monitoring sites. Although comparisons with larger samples have indicated the results are representative (Hazel and others, 2022), the degree to which these sites represent all sandbars in Marble and Grand Canyons could change over time. Therefore, inclusion of the periodic surveys of long river segments (Grams and others, 2019) provides important verification of the representativeness of the annual metric. These data do not consider the effects of vegetation expansion, which may cause portions of the sandbars to be of limited use for recreation.

Goal 7, Performance Metric 7.2.2: Mean Sandbar Response to High-Flow Experiments

Metric Description and Criteria for Selection

The mean sandbar response to HFEs is a metric to indicate the degree to which HFEs continue to result in sandbar deposition. This metric is a component of the HFE Protocol with LTEMP, because decisions on proceeding with the protocol are based, in part, on a positive sandbar response to each HFE (DOI, 2011; DOI, 2016). This metric is based on a visual assessment of 45 long-term monitoring sites using daily images captured by autonomous digital cameras. These observations fill the gap between the annual topographic surveys and support the interpolated sandbar volume derived from the Mueller and Grams (2021) sandbar volume model.

Status and Interpretation of Sandbar Response to High-Flow Experiments

For each HFE, we select images from each site that show the sandbars at approximately the same discharge and in similar lighting conditions and taken within a few days before the beginning of the HFE and within a few days following the completion of the HFE. The pre- and post-HFE photographs are compared and categorized as large or moderate change (deposition or erosion) or negligible change. These evaluations are based on obvious burial or erosion of vegetation, filling or creation of gullies or cutbanks, and burial or exposure of boulders and rocks. This evaluation process is described in detail by Grams and others (2018).

The most recent HFE was conducted in April 2023 and resulted in large deposition at 30 percent of the sites. The sandbar at river mile 30.7 is an example of this (Figure 7.4, top). An additional 33 percent of the sites underwent moderate deposition. Negligible change occurred at 28 percent of the sites, including river mile 22.0 (Figure 7.4, middle). Moderate erosion occurred at 9 percent of the sites, including river mile 65.8 (Figure 7.4, bottom), while zero sites underwent large erosion.

Each of the HFEs conducted under the HFE Protocol has resulted in at least moderate deposition at a majority of the monitoring sites (Table 7.1). The proportion of sites with deposition has been consistent or increased with time, indicating that HFE effectiveness has not decreased.

For each HFE, increases in apparent sandbar size occur at approximately 50-60 percent of the monitoring sites, while negligible change occurs at approximately one-third of the sites, and apparent decreases in sandbar size occur at approximately 10 percent of the sites. The 2023 HFE was unique compared to previous HFEs in that roughly 50 percent of the sites undergoing deposition were classified as "large deposition," (Grams and others, 2018) whereas previously this number was closer to 25 percent. However, owing to erosion between the April HFE and October, the volume of sand measured in the annual monitoring surveys was similar to other measurements following HFEs.

Metric Limitations

Because this metric is a visually based categorical assessment based on comparing images collected immediately before and after each HFE, the values cannot be used to quantify the amount of sand deposition or compare among the HFEs to describe the relative benefits of HFEs of different magnitude or duration. In addition, because it evaluates sandbar response during HFEs over the entire combined length of Marble and Grand Canyons, this metric does not address segment-by-segment changes in high-elevation sandbars let alone distinguish sandbar changes in Grand Canyon from those in Marble Canyon during HFEs.

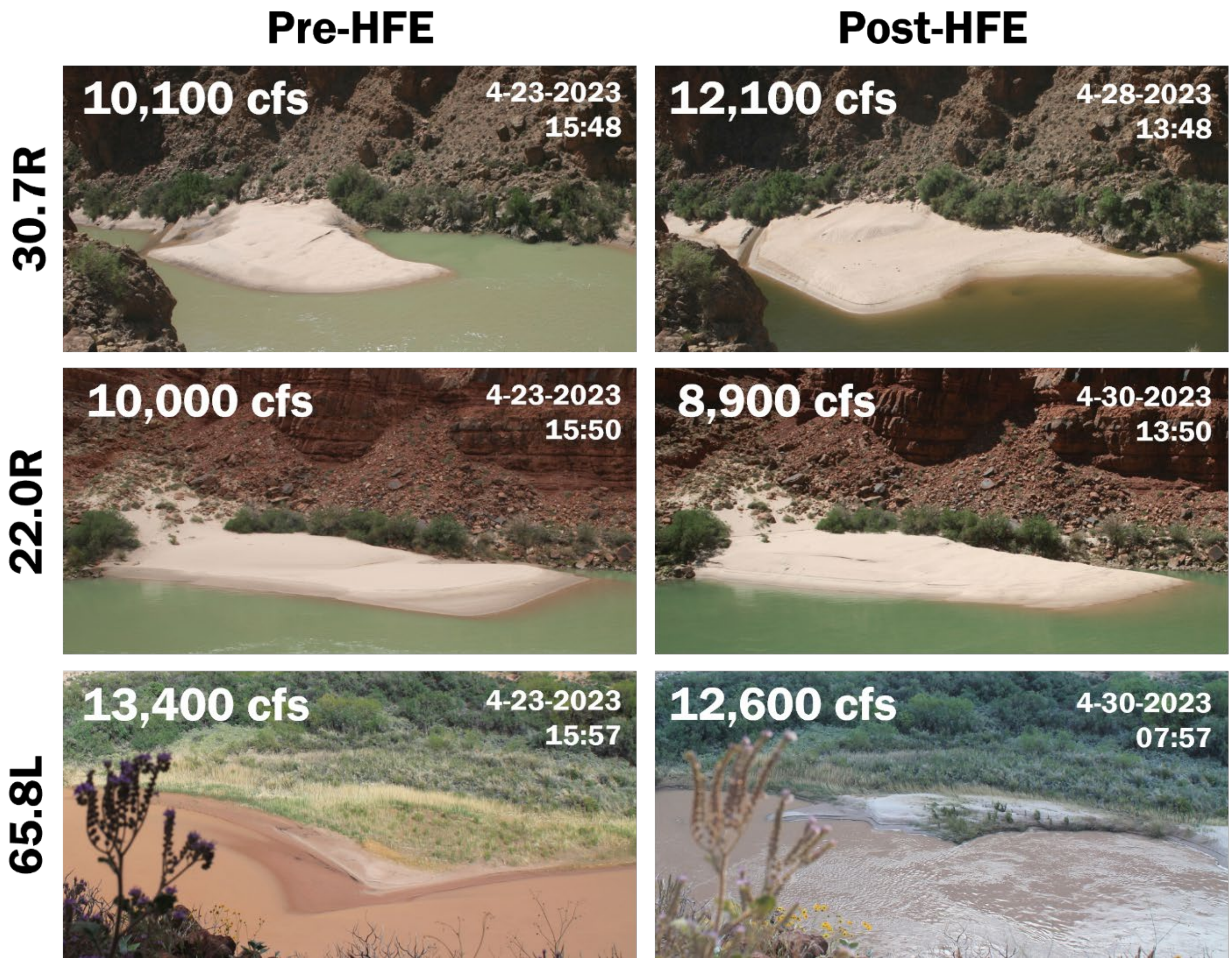


Figure 7.4. Remote camera images (U.S. Geological Survey 2025c) showing sandbar response to the 2023 high-flow experiment (HFE). River mile 30.7 (top) shows large deposition. River mile 22.0 (middle) shows negligible change. River mile 65.8 (bottom) shows moderate erosion. Cfs, cubic feet per second.

Table 7.1. Categorized sandbar response to high-flow experiments (HFEs) for long-term sandbar monitoring sites with remote cameras for HFEs conducted in the Colorado River ecosystem (CRe) from 2012 to 2023, based on sites and methods described by Grams and others (2018). Data from Grams and others (2025) using photographs from U.S. Geological Survey (2025c).

Observed response	2012 HFE (n = 33)	2013 HFE (n = 42)	2014 HFE (n = 42)	2016 HFE (n = 43)	2018 HFE (n = 43)	2023 HFE (n=43)	Average
	Immediately following HFE						
Large deposition	12%	19%	14%	14%	12%	30%	17%
Moderate deposition	39%	33%	43%	42%	54%	33%	41%
Negligible change	39%	36%	31%	33%	22%	28%	31%
Moderate erosion	9%	10%	10%	7%	10%	9%	9%
Large erosion	0%	2%	2%	5%	2%	0%	2%

Goal 7, Performance Metric 7.3: Silt and Clay Retention by River Segment

Metric Description and Criteria for Selection

The metric chosen to track the amount of silt and clay retained in the CRe for ecological purposes is the percent of time a turbidity level of 30 formazin nephelometric units (FNUs) is equaled or exceeded each sediment year (DOI, 2016). Although a threshold value of ~30 FNUs is reasonable for ecological purposes in the modern CRe, the turbidity was always greater than ~100-200 FNUs in the pre-dam CRe (Voichick and Topping, 2014). This metric is applied on a segment-by-segment basis owing to the large average difference in turbidity between different river segments (Voichick and Topping, 2014), and is calculated using 15-minute turbidity data collected at the USGS streamgage stations located at the downstream ends of the following five river segments: upper Marble Canyon, lower Marble Canyon, eastern Grand Canyon, east-central Grand Canyon, and west-central Grand Canyon (refer to list and map of gaging stations and segments in Topping and others, 2021, or at https://www.gcmrc.gov/discharge_qw_sediment/). Turbidity was chosen over a direct measure of silt and clay concentration or silt and clay mass in storage for four reasons. First, this turbidity-duration metric links silt and clay to an ecological purpose; a turbidity level of 30 FNUs provides a reasonable single-value approximation of the threshold above which trout feeding efficiency becomes disadvantaged through the reactive distance and trout growth becomes impeded (Barrett and others, 1992; Dzul and others, 2016; Korman and others, 2021). Second, because of the inverse nonlinear relation between particle size and turbidity, turbidity is much more affected by changes in silt-and-clay concentration than by changes in suspended-sand concentration (Davies-Colley and others, 1993); for this reason, turbidity is generally well-correlated with silt-and-clay concentration (Voichick and Topping, 2014).

Third, because silt and clay are transported largely as washload in the CRe and only a tiny amount of silt and clay on the bed is required to support a large amount of silt and clay in suspension, silt-and-clay concentration is a highly sensitive indicator of how much silt and clay is present on the bed and banks of the Colorado River (Topping and others, 2021). Fourth, because the annual retention of silt and clay in the CRe is typically less than ~10 percent of the tributary supply (Topping and others, 2021), it is difficult to construct continuous mass-balance silt-and-clay budgets similar to the continuous mass-balance sand budgets used to develop Metric 7.1 because the uncertainties associated with the measurements of silt-and-clay concentration used to construct such budgets are relatively large compared to the magnitude of the within-a-year silt-and-clay retention in the various river segments of the CRe. Lower values of Metric 7.3 indicate a change in management as described in Figure 7.5 if managers desire a larger amount of silt and clay on the perimeter and banks of the Colorado River.

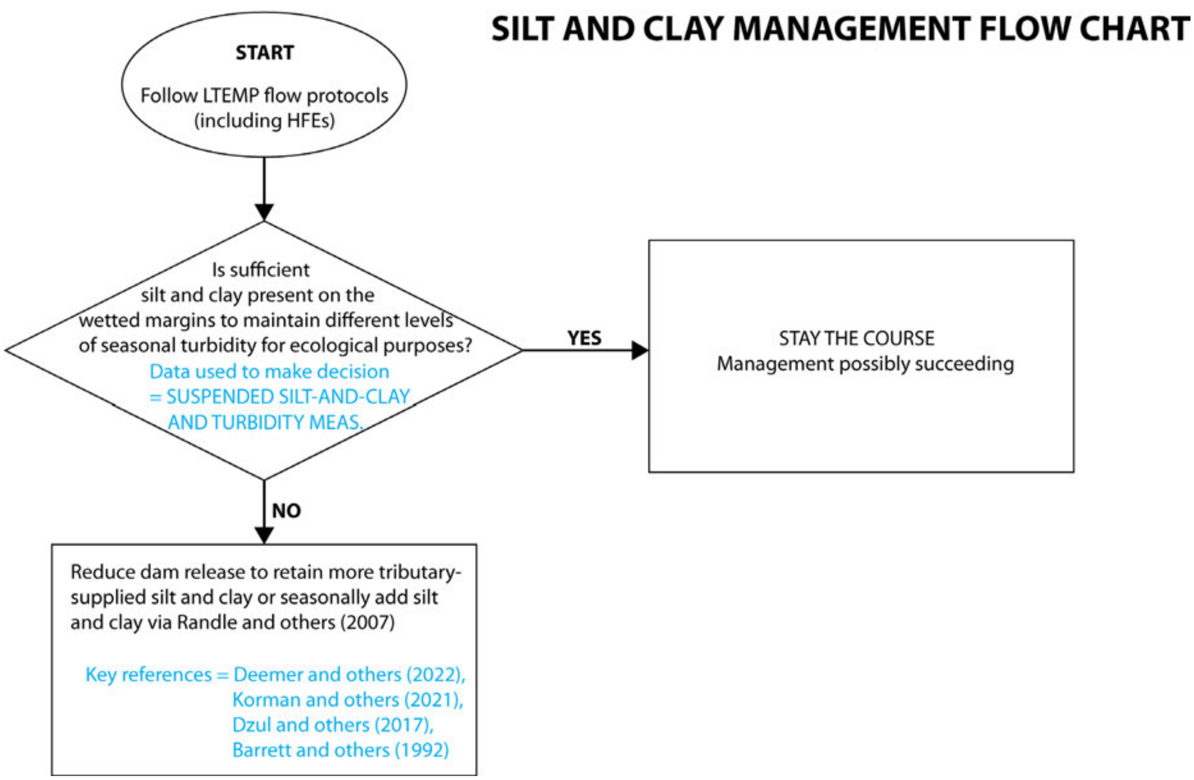


Figure 7.5. Silt and clay management flow chart option for the Colorado River ecosystem (CRe) showing the decision point and management options associated with Metric 7.3.

Status and Interpretation of Metric

The status of Metric 7.3 at three gaging stations is shown in Figure 7.6. Higher values of this metric generally indicate greater persistence of more-turbid conditions from the summer tributary-flood season into the winter months, thus indicating greater retention of silt and clay over more of each sediment year in the river segments upstream from these gaging stations (lower Marble Canyon for Figure 7.6a, eastern Grand Canyon for Figure 7.6b, and west-central Grand Canyon for Figure 7.6c). The exceptionally low values of Metric 7.3 at river mile 61 during sediment years 2018, 2020, and 2021 indicate dam releases that were too high relative to the minor tributary resupply of silt and clay to retain much silt and clay in lower Marble Canyon during these years. Similarly, the exceptionally low values of Metric 7.3 at river miles 87 and 225 during sediment years 2018 and 2021 indicate dam releases that were too high relative to the minor tributary resupply of silt and clay to retain much silt and clay in Grand Canyon during these two years. Sediment years 2018 and 2021 were the two years of clearest water throughout Marble and Grand Canyons. These two years had far less silt and clay on the perimeter and banks of the Colorado River compared to any other year since monitoring began in the mid-2000s.

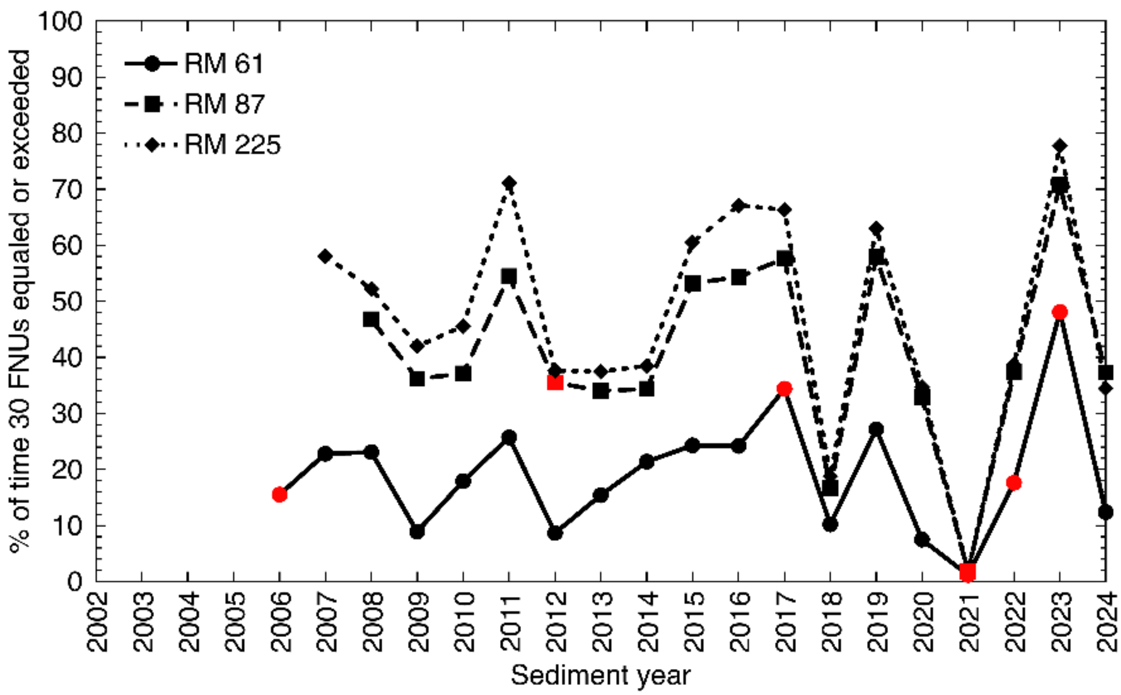


Figure 7.6. Plots of Metric 7.3, the percent of time a turbidity level of 30 formazin nephelometric units (FNUs) is equaled or exceeded during each sediment year (July–June) at river mile (RM) 61 (U.S. Geological Survey Colorado River above Little Colorado River near Desert View, AZ, 09383100 gaging station), RM 87 (Colorado River near Grand Canyon, AZ, 09402500 gaging station), and RM 225 (Colorado River above Diamond Creek near Peach Springs, AZ, 09404200 gaging station). The values in these plots were extracted from the duration-curve plots of turbidity that can be constructed for each Colorado river gaging station at https://www.gcmrc.gov/discharge_qw_sediment/stations/GCDAMP. Red symbols indicate the years where a gap in the consecutive 15-minute turbidity data exceeds 30 days, leading to greater uncertainty in the value of this metric. Plotted data from U.S. Geological Survey (2025a).

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LTEMP Resource Goal 8. Tribal Values and Resources

"Maintain the diverse values and resources of traditionally associated Tribes along the Colorado River corridor through Glen, Marble and Grand Canyons" (U.S. Department of the Interior [DOI], 2016).

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Background

The Long-Term Experimental and Management Plan (LTEMP) goal (Goal 8) for tribal values and resources is broadly worded to encompass all values and resources along the Colorado River that are of interest to multiple tribes, including but not limited to Hopi, Hualapai, Navajo, multiple southern Paiute Tribes, and the Pueblo of Zuni (DOI, 2016). At the inception of the metrics definition project, the five Tribal representatives who actively participate in the Glen Canyon Dam Adaptive Management Program (GCDAMP) indicated issues with the prospect of defining quantifiable metrics for Goal 8 (DOI, 2016). For example, there may be issues with trying to capture diverse Tribal values in any single metric or suite of metrics and conveying values through a suite of quantifiable measurements. This is a result of the uniqueness of each Tribe, and each Tribe may value the Grand Canyon and its diverse resources in accordance with their unique cultures, histories, and traditional knowledge.

Since a main purpose of the metrics is to provide a basis for assessing how well the GCDAMP is achieving the stated goals of LTEMP, providing an annual assessment of the program based on each Tribe's unique perspectives and traditional knowledge may facilitate inclusion of Tribal values. Through further conversations involving four of the five Glen Canyon Dam Adaptive Management Program (GCDAMP) Tribes (with the exception of the Pueblo of Zuni), Bureau of Reclamation, and GCMRC, it was determined that in lieu of quantified metrics, each Tribe could provide an annual assessment of the status of LTEMP resources and the GCDAMP framed around three questions:

1. Is the Colorado River ecosystem healthy? Please explain why or why not.
2. How well is LTEMP doing at meeting its goals? Please explain the basis for your assessment.
3. Overall, is the Grand Canyon doing better, worse, or about the same as last year? Please explain the basis for your assessment.

To minimize additional impacts on the Tribes, it was also discussed that the Tribes could provide answers to these questions as part of their annual monitoring reports, which are normally submitted to the Bureau of Reclamation in late winter/early spring. Therefore, in this report, we have not included Tribal responses to the three questions.

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LTEMP Resource Goal 9. Rainbow Trout Fishery

"Achieve a healthy high-quality recreational Rainbow Trout fishery in GCNRA and reduce or eliminate downstream trout migration consistent with NPS fish management and ESA compliance" (U.S. Department of the Interior [DOI], 2016).

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Background

Lees Ferry is a tailwater fishery in the Colorado River below Glen Canyon Dam, Arizona, and is managed as a Rainbow Trout (*Oncorhynchus mykiss)* fishery by the Arizona Game and Fish Department (AGFD), Glen Canyon National Recreation Area (GCNRA), and the Glen Canyon Dam Adaptive Management Program (GCDAMP; U.S. Department of the Interior [DOI], 2016). The Long-Term Experimental and Management Plan (LTEMP) goals for the Rainbow Trout fishery are "*Achieve a healthy high-quality recreational Rainbow Trout fishery in GCNRA and reduce or eliminate downstream trout migration consistent with NPS fish management and ESA compliance*" (DOI, 2016). Two metrics have been developed in response to these goals: 1) Angler catch rate for Rainbow Trout and 2) Rainbow Trout abundance in Glen Canyon.

Metric 9.1: Rainbow Trout Angler Catch Rate

Angler catch rate metrics have been established to assess the quality of the Rainbow Trout fishery, and the AGFD has employed standardized angler surveys (creel) to evaluate angler success. A mean annual catch rate of one Rainbow Trout per hour of angling has been established as the AGFD management goal for the fishery (National Park Service [NPS] 2013a, b; Rogers, 2015).

Methods

The AGFD has been conducting angler surveys at Lees Ferry since 1977. Stratified random sampling (by month and weekday/weekend) is used to select two weekday and four weekend days per month to conduct angler surveys. Anglers are classified into two categories, those going upriver in boats and those who access the fishery on foot (walk-in anglers). We use mean annual catch per unit effort (CPUE) in terms of Rainbow Trout captured per hour of angling as a standardized metric to assess catch rates for both angling groups.

Results

A total of 699 boat anglers and 329 walk-in anglers were interviewed in 2024 by AGFD. Rainbow Trout CPUE dropped significantly for both groups in 2024 and has remained below established management goals since 2015 (Figure 9.1). Rainbow trout CPUE for boat anglers was 0.44 [95 percent confidence interval, 0.39,-0.49] Rainbow Trout/hour in 2024, compared to 0.64 [0.58, 0.70] in 2023. Rainbow trout CPUE for walk-in anglers was 0.13 [0.09, 0.17] Rainbow Trout/hour in 2024, compared to 0.40 [0.27, 0.53] in 2023.

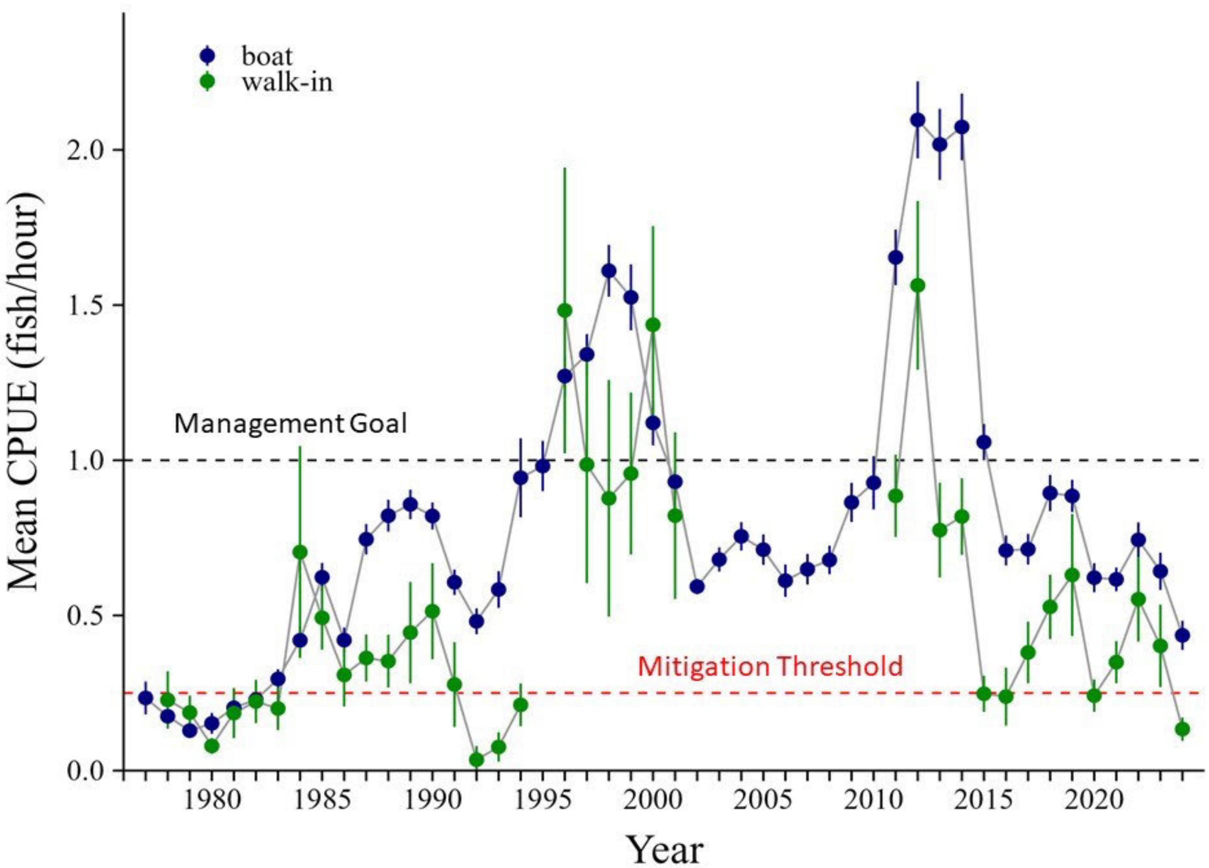

Figure 9.1. Mean angler catch per unit effort (CPUE) from 1977-2024 for boat and walk-in anglers at Lees Ferry, Arizona. Data are available upon request to the Arizona Game and Fish Department.

Discussion

Rainbow Trout CPUE for boat and walk-in anglers at Lees Ferry continued a downward trend in 2024 and remained below management goals for both groups (Figure 9.1; management goal provided in Rogers, 2015). Potential mechanisms for declines in Rainbow Trout abundance are addressed in Metric 9.2, which provides quantitative analyses designed to detect changes in Rainbow Trout abundance and recruitment.

Angler CPUE is likely determined by abundance of Rainbow Trout, which has declined in recent years (Figure 9.2; Metric 9.2). Similarly low angler CPUE was observed in the early 1980s followed by an upward trend in subsequent years (Figure 9.1). A substantial increase in Rainbow Trout abundance is likely needed to improve angler catch rates at Lees Ferry following the recent declines. Angler surveys are ongoing and can assess angler CPUE in future years.

Metric 9.2: Rainbow Trout Abundance in Glen Canyon (Age 1+)

Rainbow Trout abundance reflects underlying demographic processes (i.e., reproduction, recruitment and survival rates) linked to environmental conditions downstream of Glen Canyon Dam and is an important indicator of the state of the fishery. The abundance state may also indicate or be a precursor to downstream emigration events. Downstream movements of Rainbow Trout are episodic and largely density-dependent (linked to high abundance), and in particular, are associated with large recruitment events in Glen Canyon (Korman and others, 2016). Minimizing downstream trout immigration to Grand Canyon and reducing the likelihood of negative population level effects to threatened Humpback Chub (e.g., Yackulic and others, 2018) through competition with and predation by Rainbow Trout is an important management goal for the GCDAMP and fisheries management agencies (DOI, 2016).

Mark-recapture monitoring is conducted in two reaches within Glen Canyon to monitor both demographic processes as well as the state of the fishery through abundance estimation (Korman and others, 2021). Sampling events are conducted several times per year (4-5), through two boat mounted electrofishing passes per trip in each reach. This sampling regime, conducted in at least one reach beginning in 2012, allows for abundance estimation using a multistate robust design Jolly-Seber open population mark-recapture model (Korman and others, 2021). Sampling two reaches located in the upper and lower reaches of Glen Canyon provides some spatial resolution in trout population dynamics and reflects potential longitudinal differences in environmental variables. Here, we present results from a single reach (1C) with the longest continuous record of sampling to reflect trends in abundance (April 2012 through June 2024). We present both interval-specific (seasonal) abundance estimates for all sizes and annual abundance estimates for age-1+ fish in April (75 mm fork length and greater) to display trends in the Rainbow Trout fishery. The seasonal size-structured abundance estimates provide information on whether large recruitment events occurred that may trigger emigration.

Following the largest recruitment event observed in our time series (i.e., high abundance of smallest size class in April 2012; Figure 9.1), driven by high, phosphorus-rich inflows to Lake Powell and releases from GCD in 2011 (Korman and others, 2021; Yard and others, 2023), age 1+ abundance was elevated through 2014.

Abundance declined by ~73 percent by winter of 2015 due to a die-off associated with high densities of Rainbow Trout, low prey availability, and poor water quality due to high temperatures and low dissolved oxygen (DO) in the fall of 2014 (Yard and others, 2023; Figure 9.2). A smaller but substantive recruitment event occurred in 2017 when inflows and phosphorus loads to Lake Powell were higher. Age 1+ abundance has been low from 2019 through 2024 owing to poor recruitment (Figure 9.3). Recent low recruitment, which is an important cause of low abundance, has likely been driven by low food availability (Yard and others, 2023), predation by or competition with Brown Trout (*Salmo trutta*; Runge and others, 2018), and poor water quality from increased temperatures and low dissolved oxygen (USGS, 2023, 2025). Warmwater nonnative fishes (e.g., Green Sunfish *Lepomis cyanellus*, Smallmouth Bass *Micropterus dolomieu*) are also hypothesized to compete with or prey upon salmonids, but the population-level effects may depend on water quality and abundance of introduced predators, or other factors.

In summary, the Rainbow Trout fishery has not met management goals (DOI, 2016) for angler catch rates in recent years, and a declining trend in abundance has occurred since 2019.

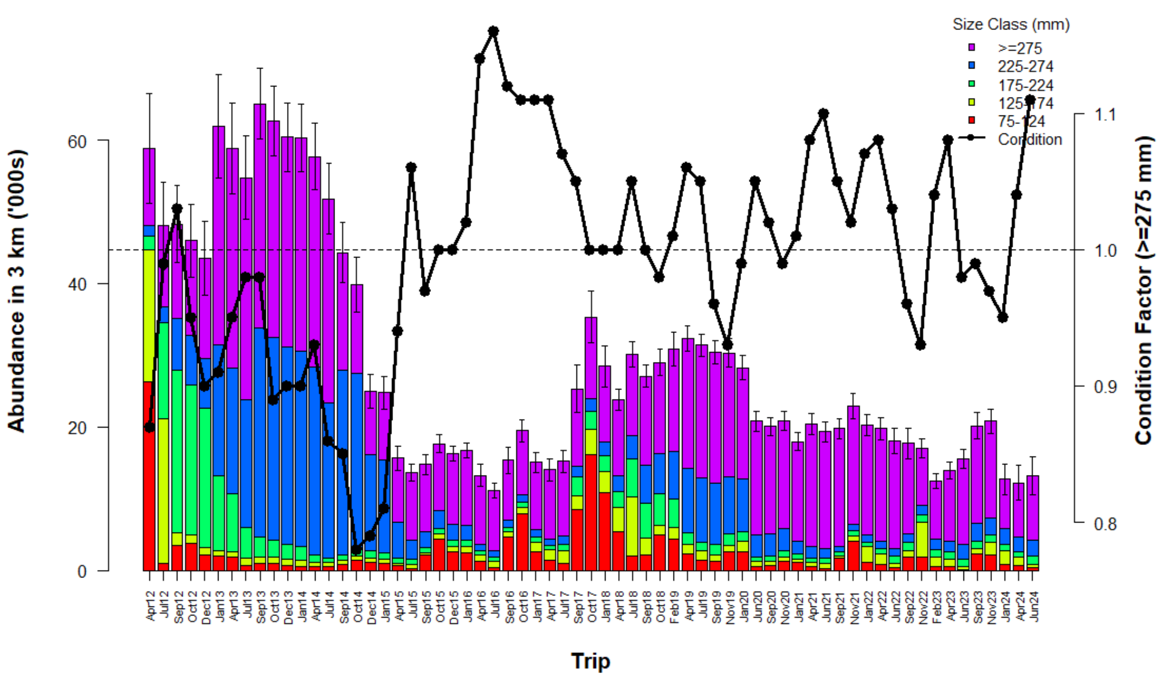

Figure 9.2. Rainbow Trout abundance by monitoring trip and size class estimated using a robust design mark-recapture model, for reach 1C of the Colorado River in Glen Canyon, Arizona, USA. Note that x-axis labels are 3-letter month abbreviation + last two digits of year. Provisional data, subject to change.

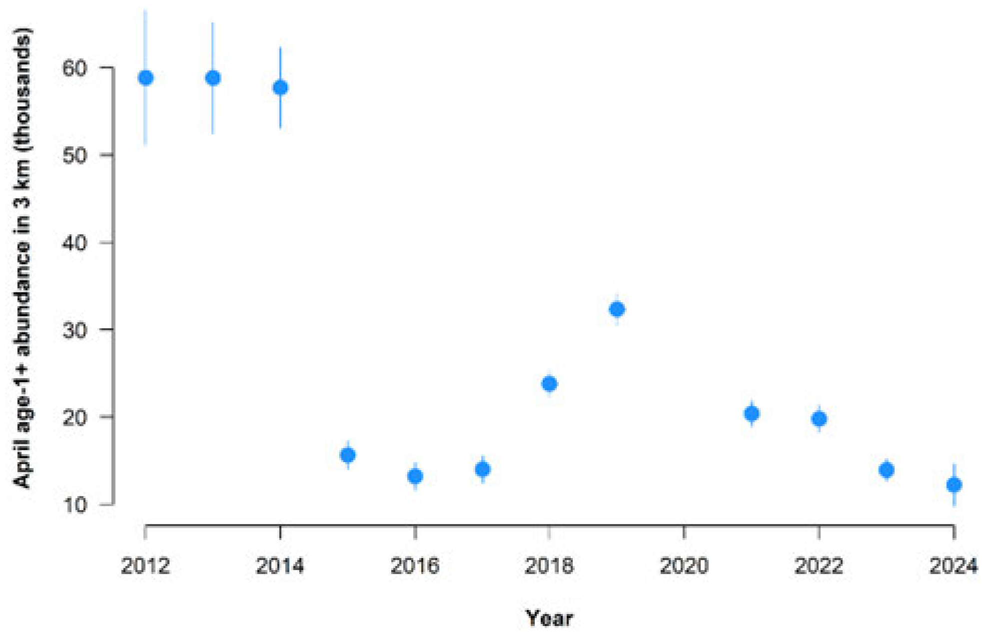

Figure 9.3. Annual April total Rainbow Trout abundance for age-1+ fish estimated using the robust design mark-recapture model, for reach 1C of the Colorado River in Glen Canyon, Arizona. Error bars indicate 1 standard deviation in the abundance estimates. Provisional data, subject to change.

All animal procedures were reviewed and approved by the U.S. Geological Survey's Southwest Biological Science Center Institutional Animal Care and Use Committee (Project ID USGS-SBSC-2024-02).

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LTEMP Resource Goal 10. Nonnative Invasive Species

> *"Minimize or reduce the presence and expansion of aquatic nonnative invasive species" (U.S. Department of the Interior [DOI], 2016).*

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Background

Aquatic invasive species, in particular introduced nonnative freshwater fishes, have been known to exert significant ecological, evolutionary, and economic impacts on their new ecosystems and may cause rapid disappearance of native fishes (Cucherousset and Olden, 2011; Britton, 2023). In both the upper and lower Colorado River Basin, warmwater invasive predatory fishes are largely incompatible with native fishes, often preying upon juveniles among other factors, and have been implicated in lack of recruitment and subsequent population declines of threatened and endangered species (Marsh and Pacey, 2005; Mueller, 2005; Minckley and Marsh, 2009; Martinez and others, 2014).

Over the past few decades, construction and operation of Glen Canyon Dam have greatly altered the fish community within the Colorado River in Grand Canyon, and those changes are not static (Van Haverbeke and others, 2017; Rogowski and others, 2018). One way that operations of Glen Canyon Dam affect fish resources downstream is through effects of lake elevation and monthly releases on water temperature. Persistent cold water released from deep within Lake Powell over the last 30+ years has historically led to low levels of invasive warmwater fishes present in the Colorado River and its tributaries in Glen Canyon and Grand Canyon. However, reservoir levels have declined in Lake Powell in recent years because of drought and water storage decisions, increasing the risk of entrainment of nonnative warmwater fishes and the thermal suitability of the Colorado River ecosystem for establishing spawning populations (Dibble and others, 2021; Bruckerhoff and others, 2022; Eppehimer and others, 2025).

Brown Trout (*Salmo trutta*) and Rainbow Trout (*Oncorhynchus mykiss)* were historically the primary species of concern downstream from Lees Ferry, Arizona and have been targeted for suppression in the mainstem (Coggins and others, 2011) and in Bright Angel Creek (Healy and others, 2020). In the Lees Ferry reach, Rainbow Trout are actively managed by the Arizona Game and Fish Department (AGFD) as a Blue-Ribbon Trout Fishery, while increasing populations of Brown Trout have been removed including via incentivized harvest. Rapidly changing conditions in the Colorado River downstream from Glen Canyon Dam since 2022, and the immediate threat of Smallmouth Bass (*Micropterus dolomieu*) and other predatory nonnative fishes to the Colorado River ecosystem, have necessitated a shift in sampling and management efforts and additional funding from outside the Glen Canyon Dam Adaptive Management Program (GCDAMP) to remove Smallmouth Bass and slow or stop growth and reproduction through experimental flow releases (Eppehimer and others, 2025).

There are eight proposed metrics for Long-Term Experimental and Management Plan (LTEMP) Resource Goal 10 that were designed to measure progress in minimizing the presence and expansion of aquatic nonnative invasive species within the Colorado River and its tributaries in Grand Canyon over the course of LTEMP (DOI, 2016). The first set of four metrics are based on the presence/absence of problematic invasive fish and crayfish species in different risk categories (low, medium, high, very-high) throughout the Colorado River in Grand Canyon on an annual basis (Table 10.1). Threat levels were initially developed by the National Park Service in collaboration with state and federal cooperators and stakeholders and were delineated based on the species' potential for predation, competition, and other negative/adverse interactions with listed species, native fishes, and the Rainbow Trout fishery (National Park Service [NPS], 2018). Risk levels were updated in 2021 based on the best available science (NPS, 2021) and can be re-evaluated in future years (E. Omana Smith, National Park Service, oral communication, 2024).

These metrics were developed to identify river segments where warmwater nonnative invasive species are present. Grouping species by risk categories aids in recognizing that not all aquatic nonnative species are of equivalent risk. The second set of four metrics in Goal 10 measure evidence of reproduction based on the presence of early life stages for each species of concern (Table 10.1). Reproduction is determined by examining length-frequency histograms for each fish species by month and location of capture. River segments with evidence of recruitment for aquatic nonnative species are indicative of where they are currently reproducing and potentially expanding into new reaches.

The data used to calculate these metrics are collected by the Grand Canyon Monitoring and Research Center (GCMRC), cooperating agencies (AGFD, NPS, U.S. Fish and Wildlife Service [USFWS]), and contracted consulting firms (Ecometric, Bio-West, American Southwest Ichthyological Researchers [ASIR]), using a wide variety of gear types across seasons.

These sampling efforts are used to catch, document, and sometimes remove or suppress invasive aquatic species (e.g., trout removal in Bright Angel Creek (BAC), Smallmouth Bass removals in Glen Canyon National Recreation Area [GCNRA]), even if they are not the specific focus of monitoring or research efforts.

The invasive aquatic species which are currently present or that have high potential to occur in the Colorado River have been grouped into risk categories in the National Park Service Expanded Nonnative Aquatic Species Management Plan (NNASMP) Environmental Assessment (EA) (NPS, 2018, 2021). These risk categories were used to develop the metrics for calendar year 2024, as reported below.

Table 10.1. National Park Service (NPS) Risk Levels for nonnative aquatic species captured in the Colorado River ecosystem and its tributaries, as specified by the Expanded Nonnative Aquatic Species Management Plan (NNASMP; NPS, 2018, 2021). Categories for 'medium-low' and 'medium-high' were grouped into medium and high-risk categories, respectively. We grouped 'Crayfish spp.' into the medium risk category because species identifications were not collected in the field, and medium represents the median risk category for the species that are either present in or have the potential to invade Grand Canyon (NPS, 2018, 2021). Species in black font were captured in 2024, those in blue italics were not captured in 2024.

Low	Medium	High	Very High
Fathead Minnow (*Pimephales promelas*)	Black Bullhead (*Ameiurus melas*)	Green Sunfish (*Lepomis cyanellus*)	Brown Trout (*Salmo trutta*)
Gizzard Shad (*Dorosoma cepedianum*)	Bluegill (*Lepomis macrochirus*)	Largemouth Bass (*Micropterus salmoides*)	Smallmouth Bass (*Micropterus dolomieu*)
Mosquitofish (*Gambusia affinis*)	Black Crappie (*Pomoxis nigromaculatus*)	Rainbow Trout (GRCA) (*Oncorhynchus mykiss*)	Walleye (*Sander vitreus*)
Plains Killifish (*Fundulus zebrinus*)	Channel Catfish (*Ictalurus punctatus*)	Striped Bass (*Morone saxatilis*)	*Flathead Catfish (Pylodictis olivaris)*
Rainbow Trout (GCNRA) (*Oncorhynchus mykiss*)	Crayfish spp. (*Orconectus rusticus, Cherax quadricarinatus, Procambarus clarkii, Faxonius virilis, Pacifastacus leniusculus, Orconectes nais*)	*Northern Pike (Esox lucius)*	

Brook Trout (Salvelinus fontinalis)/other salmonids	Common Carp (*Cyprinus carpio*)	White Sucker (*Catostomus commersonii*)	
Golden Shiner (*Notemigonus crysoleucas*)	Red Shiner (*Cyprinella lutrensis lutrensis*)	Burbot (*Lota lota*)	
Threadfin Shad (*Dorosoma petenense*)	Yellow Bullhead (*Ameiurus natalis*)		
	Blue Tilapia (Oreochromis aureus)/Cichlids		
	Grass Carp (*Ctenopharyngodon idella*)		
	Asian Carps (silver [Hypophthalmichthys molitrix]/bighead [Hypophthalmichthys nobilis])		
	Redear Sunfish (*Lepomis microlophus*)		

Metric 10.1-10.4: Detection of Invasive Aquatic Species in Each Risk Level

***Metric 10.1** – Average number of low-risk species present per habitat segment in the Grand Canyon ecosystem.*

***Metric 10.2** – Average number of medium-risk species present per habitat segment in the Grand Canyon ecosystem.*

***Metric 10.3** – Average number of high-risk species present per habitat segment in the Grand Canyon ecosystem.*

***Metric 10.4** – Average number of very high-risk species present per habitat segment in the Grand Canyon ecosystem.*

Metrics 10.1-10.4 report the total number of low, medium, high, and very high-risk invasive aquatic species within each habitat segment, and the mean number of species within each risk level for the mainstem Colorado River in Glen Canyon and Grand Canyon, and its sampled tributaries, on an annual basis (DOI, 2016). In 2024, the mainstem Colorado River was divided into 39 habitat segments measuring 12-13 river kilometers (rkm) long, which commenced at Glen Canyon Dam and ended at Pearce Ferry.

The length of these habitat segments was determined through conversations with fish biologists from agencies and partners that collect monitoring data. Different data types are typically collected at differing, but finer resolutions (e.g., electrofishing data usually come from 250-m segments, while hoop net locations are at a finer spatial resolution). Data are often summarized by individual projects at differing, coarser resolutions (e.g., Trout Recruitment and Growth Dynamics [TRGD], Juvenile Chub Monitoring [JCM] and Aggregations often summarize results for 2-3 km river segments, while AGFD has traditionally sampled randomly in larger river segments of tens of kilometers and summarized within these river segments; USGS, 2023). The resolution of 12-13 rkm was chosen as a compromise among these coarser resolutions that allowed different data to be summarized while also recognizing the scale at which environmental conditions and fish community composition might meaningfully differ (e.g., there are clear differences in the relative abundance of warmwater nonnative fishes between the upper and lower Lees Ferry segments). The habitat segments assign an upper (first 13 rkm) and lower (last 12 rkm) section of Lees Ferry, and the remaining river miles in Grand Canyon National Park (GRCA) are broadly focused on the boundaries of tributary segments (e.g., the Little Colorado River [LCR] aggregation is split into parts above and below the LCR). GCMRC and its cooperating partners and contractors collected data in each habitat segment in 2024 (refer to Table 10.2).

Each habitat segment received a score based on the number of species detected in a given risk level. For example, in the first segment downstream from Glen Canyon Dam (0-13 rkm), we captured two low-risk species in 2024 (Rainbow Trout, Gizzard Shad).

The final metric (for each risk level) was the sum of the scores for all segments divided by the total number of segments from Glen Canyon Dam to Pearce Ferry. Segment-scale and final metrics for each risk level were calculated as separate metrics. The number of species in each risk level detected in each habitat segment of the Colorado River and its sampled tributaries are reported in Tables 10.2-10.3. Over several years of metric calculation (after 2024), the average for each year can be displayed in a corresponding figure as a time series.

Table 10.2. Number of Aquatic Invasive Species by risk level in each habitat segment of the Colorado River. Provisional data, subject to change.

Habitat Segment (rkm from dam)	Metric10.1 # Low-Risk Species	Metric 10.2 # Medium-Risk Species	Metric 10.3 # High-Risk Species	Metric 10.4 # Very High-Risk Species
0-13	2	6	2	3
13-25	1	3	1	2
25-37	0	3	2	2
37-49	0	2	2	2
49-61	0	3	2	2
61-73	0	0	2	2
73-85	0	0	1	1
85-97	1	2	2	1
97-109	0	2	2	1
109-121	1	0	2	0
121-133	2	7	3	2
133-145	2	2	3	1
145-157	1	1	1	0
157-169	1	0	2	1
169-181	1	1	1	0
181-193	2	0	1	0
193-205	1	1	1	1
205-217	2	3	1	0
217-229	2	1	2	0
229-241	1	1	2	2
241-252	2	3	2	0
252-265	2	1	1	0
265-277	1	0	1	0
277-289	1	1	2	0
289-301	2	2	1	0
301-313	2	2	1	0
313-325	1	2	1	0
325-337	2	2	0	0
337-349	2	1	0	0
349-361	1	1	1	0
361-385	2	3	3	2
385-397	1	1	0	0
397-409	1	0	0	0
409-421	1	3	1	0
421-433	3	2	1	0
433-445	1	1	0	1
445-457	3	2	0	0
457-469	2	1	0	0
469-481	3	1	0	0
Mean Score	**1.4**	**1.7**	**1.3**	**0.7**

Table 10.3. Number of Aquatic Invasive Species by risk level in each habitat segment of sampled tributaries. Provisional data, subject to change.

River	Habitat Segment (rkm from confluence)	Metric 10.1 # Low-Risk Species	Metric 10.2 # Medium-Risk Species	Metric 10.3 # High-Risk Species	Metric 10.4 # Very High-Risk Species
Little Colorado River	0-6.9	2	4	2	0
Little Colorado River	7-13.6	2	4	0	1
Little Colorado River	13.7-17.8	1	4	0	0
	Mean Score	**1.7**	**4.0**	**0.7**	**0.3**
Kanab Creek	0-1	2	3	0	0
	Mean Score	**2.0**	**3.0**	**0.0**	**0.0**
Shinumo Creek	0-1	1	0	0	0
	Mean Score	**1.0**	**0.0**	**0.0**	**0.0**
Havasu Creek	0-1	0	0	0	0
	Mean Score	**0.0**	**0.0**	**0.0**	**0.0**
Bright Angel Creek	0-2.9	0	0	1	1
Bright Angel Creek	2.9-7.2	0	0	2	1
Bright Angel Creek	7.2-10.1	0	0	1	1
Bright Angel Creek	10.1-12.4	0	0	1	1
Bright Angel Creek	12.4-15.5	0	0	1	1
	Mean Score	**0.0**	**0.0**	**1.2**	**1.0**

Metric 10.1 (# low-risk species): In 2024, the average number of low-risk species per habitat segment in the Colorado River was 1.4 species, with a range of 0-3 (out of 8 species identified by NPS in this risk level). No habitat segment included all five low-risk species captured in the Colorado River in 2024. The highest number of low-risk species per habitat segment occurred in western Grand Canyon and near Pearce Ferry between rkm 421-481. Among tributary streams, Kanab Creek had the highest number of low-risk species, followed by the LCR. We captured Fathead Minnow, Gizzard Shad, Mosquitofish, Plains Killifish, and Rainbow Trout (GCNRA, low-risk species) but did not capture Brook Trout/other salmonids, Golden Shiner, or Threadfin Shad in 2024 (Tables 10.1-10.3).

Metric 10.2 (# medium-risk species): The average number of medium-risk species per habitat segment in the Colorado River was 1.7 species in 2024, with a range of 0-7 (out of 12 species identified by NPS as medium risk). No habitat segment included all 8 of the medium-risk species captured in the Colorado River in 2024. The highest number of medium-risk species per habitat segment occurred just downstream from Glen Canyon Dam (0-13 rkm) and at rkm 121-133 (near the LCR confluence). The LCR and Kanab Creek had the highest number of medium-risk species among tributary streams.

We captured Black Bullhead, Bluegill, Black Crappie, Channel Catfish, Crayfish spp., Common Carp, Red Shiner, and Yellow Bullhead but did not capture Blue Tilapia/cichlids, Grass Carp, and Asian Carp, or Redear Sunfish in 2024 (Tables 10.1-10.3).

Metric 10.3 (# high-risk species): The average number of high-risk species per habitat segment in the Colorado River was 1.3 species, with a range of 0-3 (out of 7 species identified by NPS as high risk). No habitat segment included all four of the high-risk species captured in the Colorado River in 2024. The highest number of high-risk species per habitat segment occurred at and downstream from the LCR confluence (rkm 121-145) and near Fall Canyon (rkm 361-385). Bright Angel Creek had the highest number of high-risk nonnative species due to capture of Rainbow Trout and Green Sunfish, followed by nonnatives in the LCR near its confluence with the Colorado River. These results may be driven by a higher concentration of sampling in those areas of Grand Canyon, which could lead to more detections of nonnative fishes. We captured Green Sunfish, Largemouth Bass, Rainbow Trout (GRCA, high risk), and Striped Bass but not Northern Pike, White Sucker, or Burbot in 2024 (Tables 10.1-10.3).

Metric 10.4 (# very high-risk species): The average number of very high-risk species per habitat segment in the Colorado River was 0.7 species, with a range of 0-3 (out of four species identified by NPS as very high-risk). The river segment downstream from Glen Canyon Dam (0-13 rkm) contained all three very high-risk species found in the Colorado River this year (Smallmouth Bass, Walleye, Brown Trout). Bright Angel Creek had the highest number of high-risk nonnative species due to capture of Brown Trout. We did not catch the fourth very high-risk species (Flathead Catfish) in 2024 (Tables 10.1-10.3).

Metric 10.5-10.8: Detection of Reproduction by Invasive Aquatic Species in each Risk Level

Metric 10.5 *– Average number of low-risk species with evidence of recent recruitment per habitat segment in the Grand Canyon ecosystem.*

Metric 10.6 *– Average number of medium-risk species with evidence of recent recruitment per habitat segment in the Grand Canyon ecosystem.*

Metric 10.7 *– Average number of high-risk species with evidence of recent recruitment per habitat segment in the Grand Canyon ecosystem.*

Metric 10.8 *– Average number of very high-risk species with evidence of recent recruitment per habitat segment in the Grand Canyon ecosystem.*

Metrics 10.5-10.8 report the total number of low, medium, high, and very high-risk invasive aquatic species that show evidence of reproduction (based on presence of early juvenile life stages) within each habitat segment, and the mean number of fish and crayfish species reproducing within each risk level for the mainstem Colorado River and its sampled tributaries. We used the same habitat segments as identified in Metrics 10.1-10.4.

Each habitat segment received a score based on the number of species detected reproducing in a given risk level. The final metric (within each risk level) was the sum of the scores for all segments divided by the total number of segments in each risk level. The number of species in each risk level with recruitment detected in each habitat segment is reported in Tables 10.4-10.5. Over several years of metric calculation (after 2024), the average for each year can be displayed in a corresponding figure as a time series.

Table 10.4. Evidence of reproduction (number [#] of species) of Aquatic Invasive Species by risk level in each habitat segment of the Colorado River. Provisional data, subject to change.

Habitat Segment (rkm from dam)	Metric 10.5 # Low-Risk Species Reproducing	Metric 10.6 # Medium-Risk Species Reproducing	Metric 10.7 # High-Risk Species Reproducing	Metric 10.8 # Very High-Risk Species Reproducing
0-13	1	4	2	1
13-25	1	1	1	1
25-37	0	2	2	1
37-49	0	1	2	1
49-61	0	1	2	1
61-73	0	0	2	0
73-85	0	0	1	0
85-97	1	0	1	0
97-109	0	1	1	0
109-121	0	0	0	0
121-133	2	5	1	0
133-145	2	1	1	0
145-157	0	0	0	0
157-169	0	0	0	0
169-181	0	0	1	0
181-193	2	0	0	0
193-205	1	0	0	0
205-217	2	1	1	0
217-229	2	0	1	0
229-241	1	0	1	0
241-252	2	1	0	0
252-265	2	0	1	0
265-277	1	0	1	0
277-289	1	0	1	0
289-301	2	0	0	0
301-313	2	0	0	0
313-325	1	0	0	0
325-337	2	1	0	0
337-349	2	0	0	0
349-361	1	0	0	0
361-385	2	2	2	0
385-397	0	0	0	0
397-409	0	0	0	0
409-421	1	1	0	0
421-433	3	1	0	0
433-445	1	1	0	0
445-457	3	1	0	0
457-469	2	0	0	0
469-481	1	0	0	0
Mean Score	**1.1**	**0.6**	**0.6**	**0.1**

Table 10.5. Evidence of reproduction (number [#] of species) of Aquatic Invasive Species by risk level in each habitat segment of sampled tributaries. Provisional data, subject to change.

River	Habitat Segment (rkm from confluence)	Metric 10.5 # Low-Risk Species Reproducing	Metric 10.6 # Medium-Risk Species Reproducing	Metric 10.7 # High-Risk Species Reproducing	Metric 10.8 # Very High-Risk Species Reproducing
Little Colorado River	0-6.9	2	2	0	0
Little Colorado River	7-13.6	2	3	0	0
Little Colorado River	13.7-17.8	1	3	0	0
	Mean Score	**1.7**	**2.7**	**0.0**	**0.0**
Kanab Creek	0-1	2	1	0	0
	Mean Score	**2.0**	**1.0**	**0.0**	**0.0**
Shinumo Creek	0-1	1	0	0	0
	Mean Score	**1.0**	**0.0**	**0.0**	**0.0**
Havasu Creek	0-1	0	0	0	0
	Mean Score	**0.0**	**0.0**	**0.0**	**0.0**
Bright Angel Creek	0-2.9	0	0	0	0
Bright Angel Creek	2.9-7.2	0	0	1	0
Bright Angel Creek	7.2-10.1	0	0	1	1
Bright Angel Creek	10.1-12.4	0	0	0	1
Bright Angel Creek	12.4-15.5	0	0	0	1
	Mean Score	**0.0**	**0.0**	**0.4**	**0.6**

Metric 10.5 (# low-risk species with evidence of reproduction): In 2024, the average number of low-risk species with evidence of reproduction in the Colorado River was 1.1 species and ranged from 0-3 reproducing species per habitat segment. Reproduction of low-risk species was highest in western Grand Canyon between rkm 421-433 and 445-457. Kanab Creek and the LCR supported higher numbers of species with evidence of reproduction, including species such as Fathead Minnow and Plains Killifish (Tables 10.4-10.5).

Metric 10.6 (# medium-risk species with evidence of reproduction): The average number of medium-risk species with evidence of reproduction in the Colorado River was 0.6 species per habitat segment and ranged from 0-5 species. Reproduction of medium-risk species was highest near the LCR confluence (rkm 121-133) and just downstream from Glen Canyon Dam (rkm 0-13). The LCR and Kanab Creek supported higher numbers of medium-risk species with evidence of reproduction or support of early life stages, including Channel Catfish, Bluegill, Common Carp, Red Shiner, and Black Bullhead (Tables 10.4-10.5).

Metric 10.7 (# high-risk species with evidence of reproduction): The average number of high-risk species with evidence of reproduction in the Colorado River was 0.6 species per habitat segment and ranged from 0-2 species. Reproduction of high-risk species was highest downstream from Glen Canyon Dam and into Marble Canyon (rkm 0-73) and near Fall Canyon (rkm 361-385), which is a result that could be driven by a higher amount of sampling effort in those areas. Bright Angel Creek had evidence of reproduction for one high-risk species (Tables 10.4-10.5).

Metric 10.8 (# very high-risk species with evidence of reproduction): The average number of very high-risk species with evidence of reproduction in the Colorado River was 0.2 species per habitat segment and ranged from 0-2 species. Reproduction of very high-risk species was highest downstream from Glen Canyon Dam (rkm 0-37). Bright Angel Creek had evidence of reproduction for one very high-risk species in the upper reaches (Tables 10.4-10.5).

Data Considerations and Limitations

This metrics assessment includes data from federal, state, and non-government partners that collected monitoring and research data in the Colorado River and its tributaries from January-December 2024 (with the exception of Bright Angel Creek, where data was used from fall 2023). At the time of publication, data are considered preliminary and are not ready for publication by its collectors (U.S. Geological Survey, U.S. Fish and Wildlife Service, National Park Service, Arizona Game and Fish Department, Bio-West, and the American Southwest Ichthyological Researchers).

In 2024 the Smallmouth Bass rapid response effort generated more data than is typical for the Colorado River from Glen Canyon Dam to Pearce Ferry, particularly from the dam to the Little Colorado River confluence. This can be taken into consideration when analyzing data in future years relative to this baseline metric year, as sampling effort should be fairly consistent across years to make reliable comparisons. Location-specific results may also be driven by higher levels of sampling effort in Glen and upper Marble Canyons, near the LCR and Fall Canyon, and in Bright Angel Creek. In addition, detection of reproduction by some species can be difficult, and non-detection does not necessarily mean recruitment of a species is not occurring. Due to the possibility of entrainment from Lake Powell, detection of young juvenile nonnative fishes does not always equate with reproduction in Glen and Grand Canyons. Coolmix flows in 2024 likely prevented or limited reproduction of warmwater nonnative fish in Lees Ferry downstream to the Little Colorado River but may have also affected capture probabilities due to cold water in comparison to the last few years. Differences in capture probabilities and efforts across time and space may benefit from using occupancy models (which explicitly model detection probabilities) in lieu of detection data and could include a concerted effort to design an annual process for preparing data so that results are available in a timely manner.

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All animal procedures were reviewed and approved by the U.S. Geological Survey's Southwest Biological Science Center Institutional Animal Care and Use Committee (Project ID USGS-SBSC-2024-02, USGS-SBSC-2024-05, USGS-SBSC-2024-08, USGS-SBSC-2025-01, USGS-SBSC-2024-09).

LTEMP Resource Goal 11. Riparian Vegetation

> *"Maintain native vegetation and wildlife habitat, in various stages of maturity, such that they are diverse, healthy, productive, self-sustaining, and ecologically appropriate" (U.S. Department of the Interior [DOI], 2016).*

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Background

Riparian plant communities along the Colorado River between Glen Canyon Dam and Lake Mead ("Colorado River ecosystem" [CRe] hereafter) are key components of the river ecosystem. Riparian plants are culturally and socially important and affect both the physical and biological properties of rivers. The over 300 vascular plant species found within the CRe increase regional biodiversity, support migratory and resident wildlife, contain culturally important species, and provide shade and windbreaks for recreationists (Fairley, 2005; Ralston, 2005; Sabo and others, 2005; Spence, 2006; Jackson-Kelly and Hubbs, 2007). Riparian plants are integrally linked to sediment deposition and erosion, such that complex feedback loops determine how fluvial sediment is deposited and transported to upland ecosystems (Butterfield and others, 2020; Merritt, 2022; Sankey and others, 2023). The quality of plant communities is related to both the amount of that plant community on the landscape and the species that form it. The first aspect of plant communities can be represented by an estimate of how much of the ground is covered by living plants (cover). The second aspect can be represented by how much native (as opposed to nonnative) plants contribute to existing cover and by the number of species that occur in a defined area (composition).

Riparian plant communities are strongly influenced by flow patterns, so dam operations that change flow patterns change the cover and composition of riparian plants (Poff and others, 1997; Sankey and others, 2015a; Durning and others, 2021; Palmquist and others, 2023, 2025). Current policies for Glen Canyon Dam operations result in three longitudinal bands within the riparian area that are flooded at different frequencies (U.S. Department of Interior [DOI], 2016; Palmquist and others, 2018). The band, or hydrologic zone, that is most frequently inundated is referred to here as the "active channel" or "AC." This includes all areas inundated by releases up to 25,000 cubic feet per second (cfs; 707 m^3/s). The "active floodplain" or "AF" is inundated by high-flow experiments and includes areas that are inundated by releases between 25,000 cfs and 45,000 cfs (1,274 m^3/s).

The “inactive floodplain” or “IF” is the area along the river that is inundated by releases over 45,000 cfs, which is not planned under current policies (DOI, 2016). These three zones are expected to have different plant communities and trends over time due to the strong influence of inundation and water availability for plants (Sankey and others, 2015a; Palmquist and others, 2023; Butterfield and Palmquist, 2024).

Three metrics to evaluate whether or not the Long-Term Experimental and Management Plan (LTEMP) Goal 11 is being met are currently proposed – total living plant cover, the proportion of living cover composed of native species, and native species richness (Palmquist and others, 2023). Each is described below. These metrics have not yet been finalized by the Glen Canyon Dam Adaptive Management Program (GCDAMP), so both the metrics themselves and their analyses are preliminary.

General Methods and Limitations

Each metric is calculated for hydrologic zones associated with ranges of inundating discharge and for each year where data are available. Metric estimates evaluating the riparian area can be calculated periodically from vegetation classification maps derived from remote sensing aerial imagery (e.g., Sankey and others, 2015a; Durning and others, 2021). The remote sensing results presented here are calculated from imagery acquired at approximately decadal or shorter timesteps from 1965 to 2021, and for five segments of the river channel riparian area used by Sankey and others (2015a, b).

Annual estimates based on a sample of the riparian area can be assessed through the GCMRC riparian plant community long-term monitoring program (Palmquist and others, 2018). The annual results presented here are calculated using monitoring data collected following the current GCMRC riparian plant community monitoring protocol for randomly selected sample sites (Palmquist and others, 2018). In short, 80-100 sample sites are randomly selected each year. These sites include debris fans, eddy sandbars, and channel margins. At each site, ocular cover estimates of each plant species occurring in 1-m^2 quadrats spanning the hydrological zones (2,000 – 2,600 per year) are recorded, along with an estimate of total living plant cover and associated environmental variables. For the complete protocol refer to Palmquist and others (2018). Data used for analyses are available in Palmquist and DiMartini (2025).

The analyses presented here are preliminary results of these vegetation metrics. Periodic remote sensing-based estimates are generated at the frequency of overflight data collection (Sankey and others, 2015a; Durning and others, 2021). Reports of riparian vegetation trends based on annual ground-based field sampling can be generated periodically (Palmquist and others, 2018, 2023). These reports can include changes to these metrics, plus evaluations of why changes are or are not occurring.

The annual analyses below use field-estimated hydrologic zones rather than categorizing quadrats by elevation, which can be estimated from field data and elevation models (Wiele and Griffin, 1998; Magirl and others, 2008; Palmquist and others, 2025). The current models do not include estimates of observation error (Irvine and others, 2019), the process for which is currently in development (Palmquist and others, 2025).

Metric 11.1: Total Living Plant Cover

In the CRe, the total amount of space used by plants along the river corridor (either native or nonnative) determines, in part, how much camping space is available for recreationists (Hadley and others, 2018), how much habitat is available for wildlife (Spence, 2006), and how much sediment deposition occurs on sandbars (Butterfield and others, 2020) and in higher elevation sand deposits (Sankey and others, 2023). Thus, tracking total plant cover along the river corridor is relevant for assessing the goal for riparian vegetation and the overall state of the CRe.

Vegetation maps from aerial imagery can be and have been used to evaluate how the total area covered by living plants has changed over multi-year time frames (Figure 11.1; Sankey and others, 2015a; Durning and others, 2021). From 1963 through 2021, plant cover increased from less than 10 percent to more than 30 percent in the AC (8,000-25,000 ft^3/s inundation discharge zone; Figure 11.1) and AF (25,000-31,000 and 31,000-45,000 ft^3/s inundation discharge zones; Figure 11.1). From 2013 through 2021, plant cover increased from less than 20 percent to more than 30 percent in the AC (Figure 11.1).

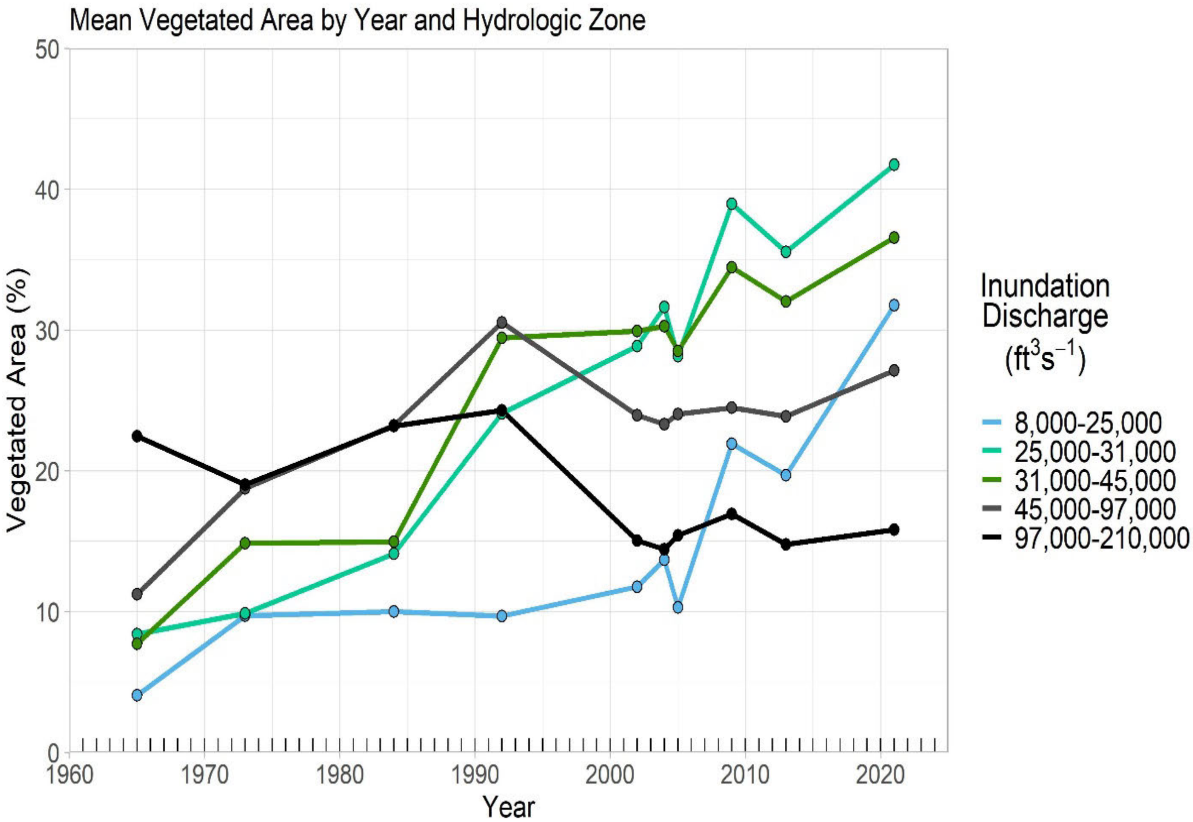

Figure 11.1. Mean percent of the Colorado River riparian area between Glen Canyon Dam and Lake Mead covered by vegetation measured from aerial imagery from nine post-dam acquisitions. Values are provided for five riparian zones defined by range of inundating discharges derived from one-dimensional hydraulic models. Data from Sankey and others (2015b), Durning and others (2017), and Sankey and others (2025).

Recent annual estimates of total plant cover illustrate interannual variability (Figure 11.2). Total plant cover has two related components, plant occurrence (presence or absence) and, when present, how much space is filled by living plants (cover). These metrics are not calculated, rather they are directly recorded data. Plant presence in a quadrat indicates that at least one plant is rooted in the quadrat, and it can have any amount of cover. Plant cover is an ocular cover estimate of how much of the quadrat is covered by living plant material from plants that are rooted in the quadrat. Refer to Palmquist and others (2018) for more details. Generally, the occurrence of any plants at all (of any size) was common in quadrats in all hydrologic zones (Figure 11.2A), though a little less in the AC. Mean cover, when plants are present, is generally higher in the active floodplain and active channel than in the inactive floodplain (Figure 11.2C).

Plant occurrence and cover can be modeled together providing a complete analysis of how plant occurrence and cover are changing over time. Each hydrologic zone was modeled separately but used the same model form.

The model framework, an ordinal, zero-augmented Bayesian regression model, is adapted from Palmquist and others (2025) and Irvine and others (2016). It allows plant presence/absence and cover to be modeled simultaneously, while appropriately modeling ordinal cover classes.

Two submodels are used, one for plant presence/absence (occurrence model) and the other for plant cover (cover model). The occurrence model utilizes a Bernoulli-logit linear regression to model the likelihood of plant presence. The cover model evaluates the probabilities of cover classes given an underlying estimated continuous latent cover, modeled with Beta-logit linear regression. Both linear models included random effects for sample site and fixed effects for sample year, with site nested in year. 'Post-sweeping' reparameterization was used to avoid issues with non-identifiability of effects (Ogle and Barber, 2020). The intercept coefficients were assigned relatively non-informative normal priors (Gelman, 2006). A relatively non-informative gamma prior was assigned to φ, a parameter in the cover model. Zero-centered normal distributions were specified for the site random effects. The standard deviations for the random effects were assigned relatively non-informative uniform priors.

The Bayesian models were implemented in JAGS (Plummer, 2003) using the R (R Core Team, 2022) packages 'jagsUI' (Kellner and others, 2019) and 'coda' (Plummer and others, 2006) on the U.S. Geological Survey (USGS) Hovenweep supercomputer (Falgout and others, 2025). The Raftery and Lewis diagnostic (raftery.diag, coda; Raftery and Lewis, 1995) was calculated based on pilot runs to determine the number of iterations needed for final runs. Three chains were run. History plots were generated with mcmcplot (mcmcplots; Curtis and others, 2015) and the Gelman and Rubin convergence diagnostic (gelman.diag, coda; Gelman and Rubin, 1992) was used to evaluate convergence and mixing. Posterior distributions of model parameters were summarized using posterior means and 95 percent credible intervals (CIs).

The mean and 95 percent CIs of the year effect indicate if there are significant changes to either occurrence or cover from year to year (Figure 11.2B and D), where error bars that do not cross zero indicate significantly higher or lower occurrence or cover. Plant occurrence in the IF has varied continuously, but not linearly. Plant occurrence has been fairly stable in the AF since 2016. The AC had higher plant occurrence in 2018 and 2019 and a large decrease in 2023, after higher steady flows in the summer of 2023 that inundated much of this zone. For plant cover (Figure 11.2D), the IF is again characterized by variable, nonlinear changes in cover. The AF exhibits similar patterns to the IF. The AC shows cover climbing through 2021, then dropping in 2022 and even further in 2023.

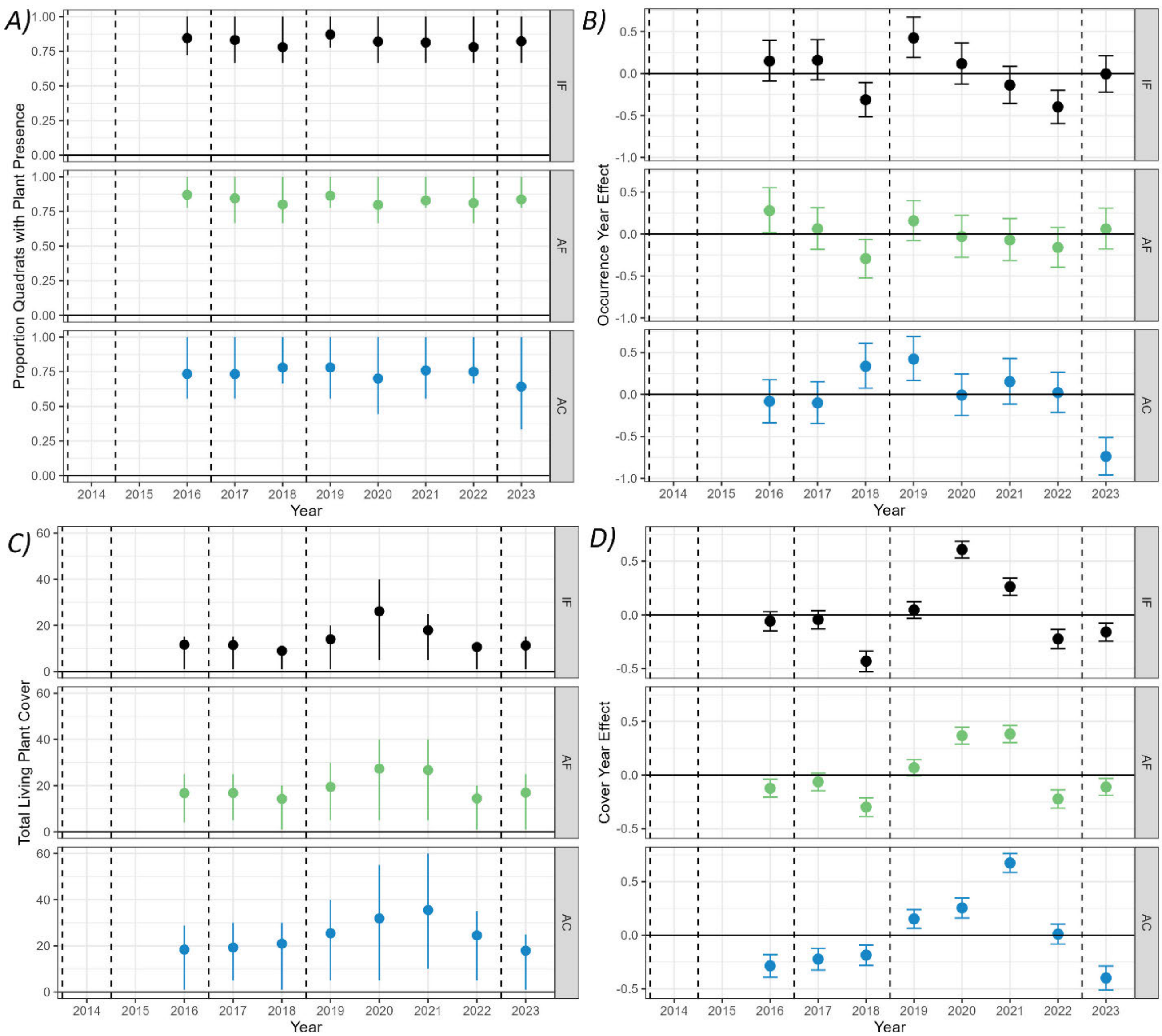

Figure 11.2. Changes in plant occurrence and total living cover in the Colorado River riparian area between Glen Canyon Dam and Lake Mead over time. Metrics are assessed for hydrologic zones defined by ranges of inundating discharge. The active channel (AC, blue) includes areas inundated by discharges up to 25,000 cfs. The active floodplain (AF, green) is inundated by discharges between 25,000 and 45,000 cfs. The inactive floodplain (IF, black) is inundated by discharges above 45,000 cfs. A) Mean proportion of quadrats within a site with plants present (point) and the 25th to 75th quantiles (error bars) for each year. B) Mean and 95 percent credible intervals of year effects for plant occurrence derived from ordinal, zero-augmented Bayesian regression. C) Mean total living cover for quadrats with plants present (point) and the 25th to 75th quantiles (error bars) for each year. D) Mean and 95 percent credible intervals of year effects for total cover derived from ordinal, zero-augmented Bayesian regression. For both B and D, error bars that do not cross zero indicate significantly higher (above zero) or lower (below zero) values in that year. Dotted lines indicate when high-flow experiments occurred in relation to sampling. Data from Palmquist and DiMartini (2025).

Metric 11.2: Proportion of Native Plant Species Cover

The proportion of vegetation cover that is composed of native plant species evaluates if a native plant community is being supported in the CRe. The proportion of native plant species cover is calculated as the sum total of native plant cover divided by the sum total of plant cover (native plus nonnative cover) for a sample quadrat. A value of 0.50 means that half of the cover of a quadrat is native species and the other half is nonnative species. This assessment is only for quadrats that have plant cover. Generally, native plant cover is greater than nonnative plant cover in the CRe, but this varies considerably (Figure 11.3A). The IF tends to have greater native species cover than either the AF or AC, with the AC having slightly lower mean proportion of native species.

A Bayesian beta-logit linear regression model was used to analyze these proportional data. Each hydrologic zone was modeled separately but using the same model form. Quadrats with a proportional cover of 1.0 were converted to 0.9999 to conform to the limitations of the Beta distribution. The model consisted of an intercept, random effects for sample sites, and fixed effects for sample year. Site effects were nested in year and utilized 'post-sweeping' reparameterization to avoid issues with non-identifiability of effects (Ogle and Barber, 2020). The intercept coefficients were assigned relatively non-informative normal priors (Gelman, 2006). A relatively non-informative gamma prior was assigned to φ, a parameter for the Beta distribution. Zero-centered normal distributions were specified for the site random effects. The standard deviations for the random effects were assigned relatively non-informative uniform priors.

These Bayesian models were also implemented in JAGS (Plummer, 2003) using the R (R Core Team, 2022) packages 'jagsUI' (Kellner and others, 2019) and 'coda' (Plummer and others, 2006) on the USGS Hovenweep supercomputer (Falgout and others, 2025). The Raftery and Lewis diagnostic (raftery.diag, coda; Raftery and Lewis, 1995) was calculated based on pilot runs to determine the number of iterations needed for final runs. Three chains were run. History plots were generated with mcmcplot (mcmcplots, Curtis and others, 2015) and the Gelman and Rubin convergence diagnostic (gelman.diag, coda; Gelman and Rubin, 1992) was used to evaluate convergence and mixing. Posterior distributions of model parameters were summarized using posterior means and 95 percent credible intervals (CIs).

The mean and 95 percent CIs for each year indicate if there are significant differences in the proportion of native species cover across years (Figure 11.3B). The IF shows highly variable proportions of native plant cover over time with 2016 and 2022 having particularly high native plant cover and 2020 having particularly low native plant cover. The AF has had higher proportions of native plant cover in recent years with 2021-2023 having significantly higher values. The AC has had fairly steady proportions of native plant cover, but somewhat higher amounts in 2019 and 2022.

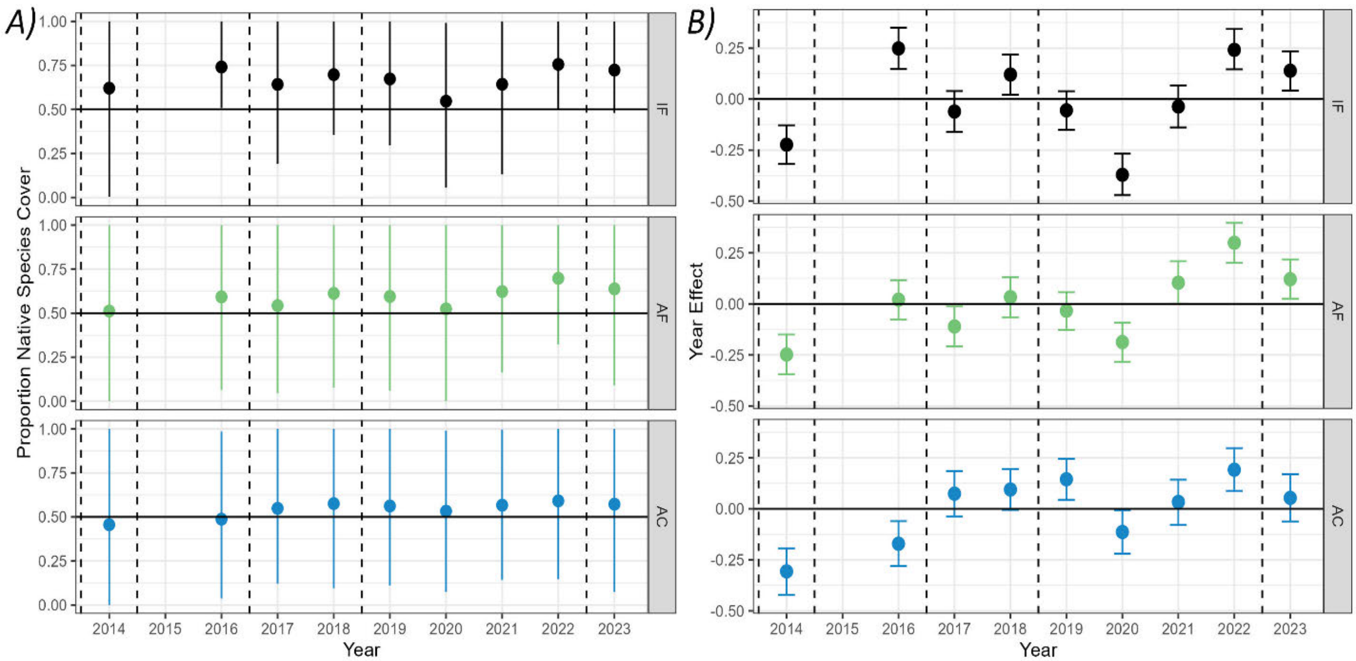

Figure 11.3. Changes in the proportion of cover composed of native plant species in the Colorado River riparian area between Glen Canyon Dam and Lake Mead over time. Metrics are assessed for hydrologic zones defined by ranges of inundating discharge. The active channel (AC, blue) includes areas inundated by discharges up to 25,000 cfs. The active floodplain (AF, green) is inundated by discharges between 25,000 and 45,000 cfs. The inactive floodplain (IF, black) is inundated by discharges above 45,000 cfs. A) Mean proportion of native species cover (point) and the 25th to 75th quantiles (error bars) are shown for each year. The horizontal black line indicates 0.50, where native and nonnative plant cover are equal. B) Mean and 95 percent credible intervals of year effects derived from beta Bayesian regression. Error bars that do not cross zero indicate significantly higher (above zero) or lower (below zero) proportion of native cover in that year. Dotted lines indicate when high-flow experiments occurred in relation to sampling. Data from Palmquist and DiMartini (2025).

Metric 11.3: Native Plant Richness

Native plant richness indicates if the CRe is supporting a wide variety of plant species or if only a few plant species make up the riparian plant communities. Native plant richness is calculated as the total number of native species rooted inside a sample quadrat. At the sample quadrat level, native plant richness ranges from 0 to 11 with a mean of 1.29 and a median of 1 (Figure 11.4A). There is considerable variability in the native plant richness across quadrats within a year.

A Bayesian negative-binomial regression with a log-link was used for these over-dispersed count data. Each hydrologic zone was modeled separately but used the same model form. Similar to the proportion of native plant cover model, this model consisted of an intercept, random effects for sample sites, and fixed effects for sample year.

Site effects were nested in year and utilized 'post-sweeping' reparameterization to avoid issues with non-identifiability of effects (Ogle and Barber, 2020). The intercept coefficients were assigned relatively non-informative normal priors (Gelman, 2006).

A relatively non-informative gamma prior was assigned to the overdispersion parameter, r. Zero-centered normal distributions were specified for the site random effects. The standard deviations for the random effects were assigned relatively non-informative uniform priors.

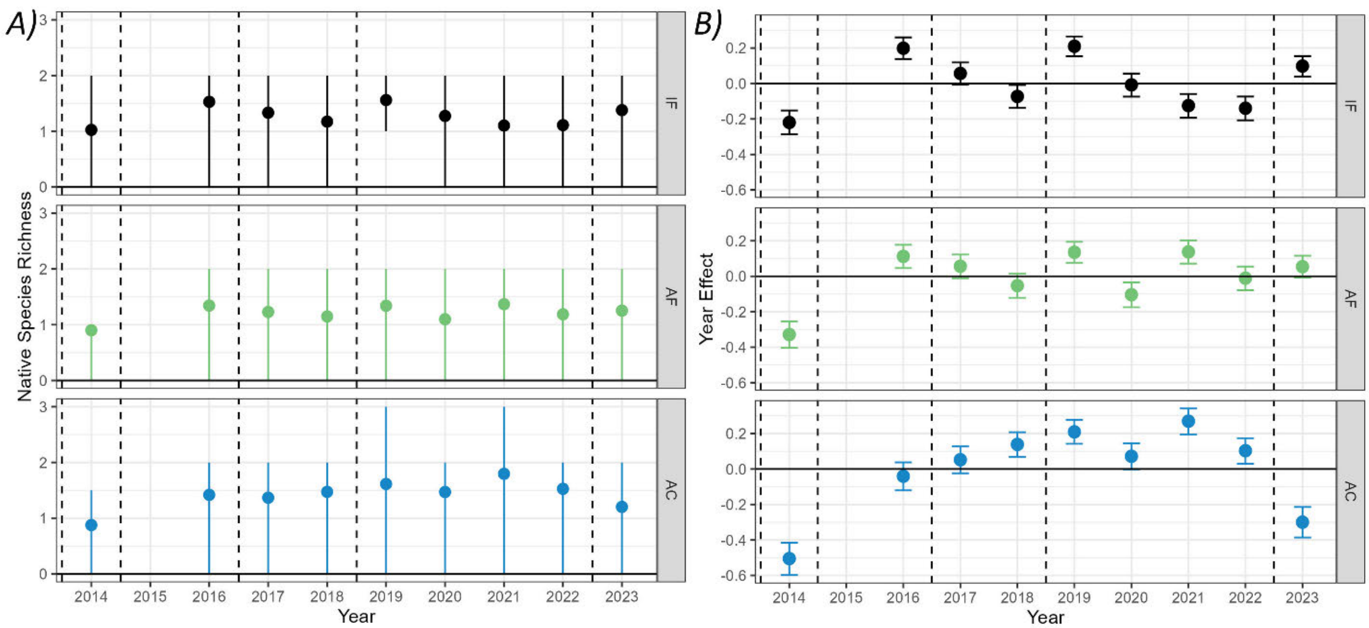

Figure 11.4. Changes in native plant richness in the Colorado River riparian area between Glen Canyon Dam and Lake Mead over time. Metrics are assessed for hydrologic zones defined by ranges of inundating discharge. The active channel (AC, blue) includes areas inundated by discharges up to 25,000 cfs. The active floodplain (AF, green) is inundated by discharges between 25,000 and 45,000 cfs. The inactive floodplain (IF, black) is inundated by discharges above 45,000 cfs. A) Mean richness (point) and the 25th to 75th quantiles (error bars) of richness values for each year. B) Mean and 95 percent credible intervals of year effects derived from negative binomial Bayesian regression. Error bars that do not cross zero indicate significantly higher (above zero) or lower (below zero) richness in that year. Dotted lines indicate when high-flow experiments occurred in relation to sampling. Data from Palmquist and DiMartini (2025).

These Bayesian models were also implemented in JAGS (Plummer, 2003) using the R (R Core Team, 2022) packages 'jagsUI' (Kellner and others, 2019) and 'coda' (Plummer and others, 2006) on the USGS Hovenweep supercomputer (Falgout and others, 2025). The Raftery and Lewis diagnostic (raftery.diag, coda; Raftery and Lewis, 1995) was calculated based on pilot runs to determine the number of iterations needed for final runs. Three chains were run. History plots were generated with mcmcplot (mcmcplots; Curtis and others, 2015) and the Gelman and Rubin convergence diagnostic (gelman.diag, coda; Gelman and Rubin, 1992) was used to evaluate convergence and mixing. Posterior distributions of model parameters were summarized using posterior means and 95 percent credible intervals (CIs).

Values of native plant richness were higher in the AC from 2018 through 2022 and lower in 2014 and 2023 (Figure 11.4B). The AF had higher richness in 2016, 2019, and 2021 and lower richness in 2014 and 2020. The IF had higher native plant richness in two of the three years where sampling occurred within a year of the last high-flow experiment (2019 and 2023, Figure 11.4B). Richness was also higher in the IF in 2016.

Summary and Interpretation

Taking a periodic evaluation of living plant cover over decades together with recent annual evaluations of occurrence, cover, and richness indicates that plant communities are adjusting to changes in discharge. Vegetation cover expanded at higher positions in the riparian area earlier, and are now increasing at lower positions in the riparian area. Higher, steadier releases in 2023 appear to have resulted in a decline in occurrence, cover, and richness in the area flooded by those flows. Periods of lower overall discharge are associated with increasing plant cover in the AC and AF and increasing richness in the AC. High-flow experiments, particularly fall-timed experiments, may increase plant occurrence in the AF and IF and richness in the IF. However, these data can benefit from using measures of multiple flow metrics and precipitation to determine the most likely driver of these patterns. Annual sampling indicates that despite high variability among plots and sites, rapid (annual) changes to plant metrics in response to flows occur and can be detected.

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