Technical Appendix 8

Biological Resources – Fish and Other Aquatic Species

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Acronyms and Abbreviations

Acronym or Abbreviation	Full Phrase
AZGFD	Arizona Game and Fish Department
BLM	Bureau of Land Management
CCS	Continued Current Strategy
CI	confidence interval
CRSS	Colorado River Simulation System
DMDU	Decision Making Under Deep Uncertainty
EIS	Environmental Impact Statement
ESA	Endangered Species Act
LB Priority	Lower Basin Priority
LB Pro Rata	Lower Basin Pro Rata
LCR MSCP	Lower Colorado River Multi-Species Conservation Program
Lower Basin	Lower Colorado River Basin
LTEMP	Long-Term Experimental and Management Plan
maf	million acre-feet
mg/L	milligrams per liter
mm	millimeter
NDOW	Nevada Department of Wildlife
NPS	National Park Service
PIT	Passive Integrated Transponder
Reclamation	Bureau of Reclamation
RM	river mile
SIB	Southerly International Boundary
FWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey

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TA 8. Biological Resources – Fish and Other Aquatic Species

TA 8.1 Affected Environment

The affected environment includes a diverse range of aquatic organisms, including foundational elements of the food web that support native and nonnative fish populations. Native fish species analyzed in this section are special status species, including federally threatened and endangered, range-wide conservation species, Bureau of Land Management sensitive species, and Lower Colorado River Multi-Species Conservation Program (LCR MSCP) covered species. Nonnative fish species analyzed in this section include cold-water and warm-water species (see **TA 8 Attachment 1** for detailed life histories and distributions), including state-regulated sportfish. Variations in river flow can impact these aquatic organisms and their habitats, affecting areas within the channel, along the shorelines, in backwaters, and at the mouths of tributaries.

TA 8.1.1 Aquatic Food Base

The basic components of the aquatic food base are invertebrates, algae, rooted plants, and organic matter (Gloss et al. 2005). Although most of this food base is produced within the aquatic system, terrestrial inputs of organic matter and invertebrates also contribute. In turn, instream production of algae and invertebrates helps support terrestrial consumers, such as grasshoppers and spiders, insectivorous birds and bats, reptiles, amphibians, mammals, and waterfowl. Indirect links include peregrine falcons, belted kingfishers, osprey, great blue herons, and bald eagles, which feed on fishes or waterfowl that consume aquatic food base organisms (Bastow et al. 2002; Baxter et al. 2005; Sabo and Power 2002; Shannon et al. 2003; Stevens and Waring 1986; Yard et al. 2011). A description of the aquatic food base prior to and following the construction and operation of Glen Canyon Dam can be found in the 2016 Glen Canyon Dam Long-Term Experimental and Management Plan (LTEMP) Final Environmental Impact Statement (EIS; Reclamation and NPS 2016) and was supplemented in Glen Canyon Dam LTEMP Final Supplemental EIS (2024 LTEMP Final SEIS; Reclamation 2024). This discussion includes invasive aquatic species that have affected or may affect food base organisms of the Colorado River Ecosystem.

The major groups of the aquatic food base include (1) periphyton (e.g., algae and cyanobacteria that live attached to rocks and other surfaces) and rooted aquatic plants, (2) plankton (very small plants [phytoplankton] and animals [zooplankton] that occur in the water column), and (3) macroinvertebrates (i.e., invertebrates that are visible to the naked eye). Large mainstem dams on the Colorado River have altered the physical, chemical, and biological nature of the river by changing flow, temperature, sediment, and salinity regimes (Cross et al. 2013). Nutrients and sediments have become trapped in reservoirs, such as Lake Powell, depleting downstream reaches of important carbon sources for primary and secondary production and foundational sands and silts for the establishment of rooted vegetation that supports diverse riparian animal communities (Johnson and Carothers 1987). These changes have altered riverine food webs, reduced biodiversity, and can often lead to the extirpation of native species while facilitating invasion by nonnative species (Miller 1961).

After Glen Canyon Dam was constructed, the combination of altered flows and temperatures, reduced organic inputs from areas upstream, decreased turbidity, and altered the thermal regime led to a shift in the aquatic food base of the Colorado River downstream of the dam (Benenati et al. 2002; Blinn et al. 1995; Kennedy and Gloss 2005). In general, aquatic invertebrate diversity declined while density and biomass increased (Kennedy and Gloss 2005). The influence of Glen Canyon Dam, coupled with sediment inputs from tributary streams, resulted in a stair-step decrease in food base biomass in the Colorado River due to clear and cold hypolimnetic dam releases and to downstream increases in turbidity from tributary sediment inputs. In the post-dam period, the 16-mile reach of Glen Canyon from the dam to Lees Ferry, accounted for 69 percent of the algal and 50 percent of the macroinvertebrate mass collected throughout a 224-mile section of river downstream of the dam (Melis 2011). High sediment inputs from tributary streams downstream of the Paria River have resulted in an aquatic food base with lower biomass and species diversity. The suspended sediments increase turbidity, and the deposited sediments alter substrate characteristics (Shannon et al. 1994; Shannon et al. 2001; Melis 2011). Thus, the aquatic food bases of the tailwater section (between the dam and the Paria River [Lees Ferry sub-reach]) differ from the rest of the mainstem (between the Paria River and Diamond Creek). In the clear tailwaters of the Lees Ferry sub-reach, aquatic productivity is driven primarily by photosynthetic production of algae, diatoms, and macroinvertebrates. The Colorado River downstream of the Paria River is seasonally influenced by tributary sediment and organic matter inputs, making the river more like pre-dam conditions, particularly as the distance from the dam increases (Rosi-Marshall et al. 2010) and water temperatures equilibrate with air temperature (i.e., warm in summer and cool in winter).

Most aquatic insects have complex life cycles with a winged adult stage that is terrestrial, while egg, larval, and pupal stages are aquatic. Approximately 80 percent of adult aquatic insects (Kennedy et al. 2016) use river edge habitats for egg laying, whereby eggs are cemented onto rocks or vegetation along the river edge and just under the water surface. Brief desiccation of insect eggs (on a scale of hours), as is typical of eggs laid in the varial zone (water-shoreline interface that is wet and dry daily due to dam operations), leads to high mortality (Kennedy et al. 2016). Evidence for this egg mortality effect can be seen throughout the Grand Canyon, with aquatic insects being more abundant in locations where the timing of daily low flows coincides with peak egg-laying activity (i.e., late afternoon), because insect eggs laid in these locations are never subjected to desiccation-induced mortality. Additional evidence of this egg mortality effect is seen by comparing insect diversity downstream of dams throughout the Western United States, with insect diversity being strongly and negatively correlated to the degree of hydropower production (Kennedy et al. 2016). Thus, the varial zone downstream of hydropower dams, such as Glen Canyon, reduces the quality and availability of river edge habitats that aquatic insects use for their egg-laying; this constrains both the diversity and abundance of aquatic insects that are present in these ecosystems and affect food sources for sensitive fish species. "Bug flows" were implemented as experimental releases from Glen Canyon Dam to increase the abundance (number of individuals) and the diversity (number of types) of insects in Grand Canyon by improving egg-laying conditions for insects (GCMRC 2022). These experiments were conducted under the 2016 LTEMP compliance and are not associated with this EIS.

The Colorado River system is close to large human population centers and is susceptible to introductions of invasive aquatic species. Recreational boaters and anglers have inadvertently spread quagga mussels (*Dreissena bugensis*), New Zealand mud snails (*Potamopyrgus antipodarum*), and numerous aquatic plants into numerous locations of the Colorado River. Other nonnative and invasive species that have potentially detrimental effects on both the food base and fish communities have been established in the Colorado River. Nonnative and invasive species introductions, as related to the Colorado River food base, are discussed in detail in the 2024 LTEMP Final SEIS (Reclamation 2024).

TA 8.1.2 Fish Species

The Colorado River and its tributaries have supported a unique and highly indigenous native fish assemblage, as well as approximately 67 nonnative species introduced over the last century (Valdez and Muth 2005), primarily to sustain and support recreational sports fisheries. Because of a prolonged period of geologic isolation, particularly in the Upper Basin, the Colorado River had only 35 native fish species that were freshwater or euryhaline (Miller 1955, 1958), including 14 in the Upper Basin (Valdez and Muth 2005). Of the 35 species native to the Colorado River System, 26 (74 percent) were endemic, or found in no other river basin on earth (Miller 1958). Of the 35 species, 3 are extinct and 12 are listed as federally endangered or threatened (Bestgen et al. 2020; Minckley et al. 2003; Reclamation 2024). The fish species analyzed in this section are included in the 35 native species.

Native Fish Species

There are 7 species of native fish within the analysis area. Four of these species—the humpback chub (*Gila cypha*), razorback sucker (*Xyrauchen texanus*), Colorado pikeminnow (*Ptychocheilus lucius*), and bonytail *(Gila elegans)*—are federally listed, and the bluehead sucker (*Pantosteus discobolus*) and flannelmouth sucker (*Catostomus latipinnis*) are conservation species. The speckled dace (*Rhinichthys osculus*) is a sensitive species. The status and distributions of these native species in the analysis area are shown in **Table TA 8-1**. The roundtail chub (*Gila robusta*) was reported downstream of Lava Cliff Rapid (river mile [RM] 246) in the 1940s before the area was inundated by Lake Mead (Miller 1958), but it has not been subsequently observed in the mainstem downstream of Lake Powell. The species persists in the Little Colorado River in reaches outside the analysis area and is not included in the analyses.

Table TA 8-1
Native Fish of the Colorado River in the Analysis Area

Species	Listing Status[1]	Presence in Analysis Area[2] (river miles downstream from Lees Ferry)
Humpback chub (*Gila cypha*)	ESA-T, CH; AZ-SGCN	30-Mile Spring (29.8) to Pearce Ferry (280.7); Little Colorado River (61.8); translocated to Shinumo (109.2), Havasu (157.3), and Bright Angel (88.3) creeks. Some individuals occur downstream of Pearce Ferry and Pearce Ferry Rapid.

Species	Listing Status[1]	Presence in Analysis Area[2] (river miles downstream from Lees Ferry)
Razorback sucker (*Xyrauchen texanus*)	ESA-E, CH; AZ-SGCN	Bright Angel Creek (88.3) confluence to Pearce Ferry (280.7); few upstream of Lava Falls (179.7); much of Lake Powell, Lake Mead, Lake Mohave, and Lake Havasu, as well as the much of the Lower Colorado River proper
Bonytail (*Gila elegans*)	ESA-E, CH; AZ-SGCN	Stocked in Lake Powell and Lower Colorado River reservoirs, the Lower Colorado River proper and its backwaters; recent bonytail larvae in Lake Powell indicate reproduction.
Colorado pikeminnow (*Ptychocheilus lucius*)	ESA-E, CH; AZ-SGCN	Colorado River and San Juan River inflows to Lake Powell as extensions of upstream populations
Flannelmouth sucker (*Catostomus latipinnis*)	NL; CSp; AZ-SGCN	Glen Canyon Dam (-15.8) to Pearce Ferry (280.7), including tributaries, inflow areas of lakes Mead and Powell, as well as the reintroduced population downstream of Davis Dam
Bluehead sucker (*Pantosteus discobolus*)	NL; CSp; AZ-SGCN	Paria River (0.8) to Pearce Ferry (280.7), including tributaries
Speckled dace (R*hinichthys osculus*)	NL; AZ-SGCN	Paria River (0.8) downstream to Pearce Ferry (280.7), including tributaries

Sources: 56 *Federal Register* 54957; AZGFD 2001a, 2001b, 2002a, 2002b, 2003; Albrecht et al. 2014; Andersen et al. 2010; Bezzerides and Bestgen 2002; Coggins and Walters 2009; Francis et al. 2015; GCMRC 2014; Gloss and Coggins 2005; Makinster et al. 2011; Ptacek et al. 2005; Rees et al. 2005; Rinne and Magana 2002; Ward and Persons 2006; Woodbury 1959; FWS 2002; Valdez and Carothers 1998; Brockdorff 2022

[1]ESA = Endangered Species Act; E = endangered, T = threatened; CH = federally designated critical habitat in analysis area; AZ-SGCN = Arizona species of greatest conservation need; NL = not federally listed; CSp = included in the Rangewide Conservation Plan and Agreement and Strategy (UDWR 2006)

[2]Habitat and life history information is presented in species-specific descriptions in this section.

Nonnative Fish Species

Generally, nonnative fish species in the Colorado River system pose risks to native fish species through competition for resources, predation, and habitat alteration. In limited cases, nonnative fish species in the Colorado River Basin can provide beneficial effects (i.e., fishing and providing forage for other sportfish [see **TA 14.1.11**, Sportfishing, in **TA 14**, Recreation]) depending on their location in the system and their recreational value. Reservoirs frequently support large populations of nonnative sportfish, and the tailwater of Glen Canyon Dam supports a high-value trout fishery. When nonnative species move outside of these sportfishing areas, they pose threats to native fish communities. In 2021 the National Park Service (NPS) Expanded Nonnative Aquatic Species Management Plan in Glen Canyon National Recreation Area and Grand Canyon National Park (2018) was updated to better reflect the threat level of nonnative fish in Glen Canyon National Recreation Area and Grand Canyon National Park, below Glen Canyon Dam, based on their potential for predation, competition, or other adverse interactions with native and federally listed species, as well as to the recreational rainbow trout (*Oncorhynchus mykiss*) fishery in the Lees Ferry sub-reach (NPS 2021). Four species were assigned a threat level of "(1) Very High", including smallmouth bass (*Micropterus dolomieu*), walleye (*Sander vitreus*), flathead catfish (*Pylodictis olivaris*), and brown trout (*Salmo trutta*). Five species were assigned a threat level of "(2) High", including rainbow trout (high in Grand Canyon National Park, low in Glen Canyon National Recreation Area), northern pike (*Esox lucius*), green sunfish (*Lepomis Cyanellus*), striped bass (*Morone saxatilis*), white sucker (*Catostomus commersonii*), and burbot (*Lota lota*).

Nonnative, predatory species known to occur within the upper and lower Colorado River are black bullhead (*Ameiurus melas*), black crappie (*Pomoxis nigromaculatus*), white crappie (*Pomoxis annularis*), channel catfish (*Ictalurus punctatus*), common carp (*Cyprinus carpio*), largemouth bass (*Micropterus salmoides*), striped bass, walleye, yellow bullhead (*Ameiurus natalis*), bluegill (*Lepomis macrochirus*), green sunfish, northern pike, and tiger muskie (*Esox lucius* × *Esox masquinongy*) (UDWR 2023).

TA 8.1.3 Reaches

Table TA 8-2 provides a complete list of the native and nonnative fish species within the analysis area, including the Colorado River from Lake Powell downstream to the Southerly International Boundary (SIB). The scientific name and conservation status of each species is presented in the table, and species present within each reach are shown in the table and described in more detail in the respective sections.

Lake Powell

For this analysis, the Lake Powell reach includes approximately 250 mi² covered by the lake (Root and Jones 2022) at its full elevation of 3,700 feet above mean sea level (feet), including the inflows of two major rivers, the Colorado River and the San Juan River. Low levels of Lake Powell, starting with a receding reservoir in the year 2000, resulted in the creation of approximately 35 miles of additional riverine habitat for each of these rivers. As the lake recedes, the rivers carve through the existing sediment deposits, periodically deviating from their historical channels to carve new channels and in some cases, forming waterfalls that block the upstream movement of fish. On the San Juan River, there is an approximately 30-foot waterfall at Piute Farms acting as a major fish barrier (Cathcart et al. 2018; Pennock et al. 2024). The rapid rates of incision in the Colorado River channel near the mouth of the Dirty Devil have eroded lake deposits, but the underlying bedrock has not yet been reached. If, upon further incision, the river encounters bedrock or another resistant layer, it is possible that rapids or waterfalls could form on the order of ~1 meters to ~10 meters (Arens 2023; Grams and Tusso 2023). At full-pool, Lake Powell has a maximum depth of 568 feet. Lake Powell warms seasonally with the warmest surface temperatures of about 30 °C in July and the coldest of about 5.5 °C in January. There is some ice formation along shorelines and sheltered bays in winter, but most of the lake remains ice-free. Lake Powell is a monomictic reservoir with strong thermal stratification through much of the spring, summer, and early fall, with the warmest surface layer (epilimnion) persisting from about May to October, a middle layer (metalimnion) where there is a steep transition between warm and colder water, and the lowest level (hypolimnion) with colder, typically hypoxic, conditions. Inflow from the Colorado River and the San Juan River generally flows into the hypolimnion but may mix with other layers depending on inflow density. At lower reservoir elevations, the metalimnion or epilimnion layers can extend down to the depth of the penstocks (3,470 feet elevation), resulting in warmer release temperatures. Historically, the warmest release temperatures occurred in October and November as the reservoir cooled and mixed (Hueftle and Stevens 2001). In recent years, with lower reservoir elevations, the warmest months are early in the year during September and October.

Table TA 8-2
Native and nonnative fish species in the Colorado River from Lake Powell downstream to the SIB

Common Name	Scientific Name	Status[a]	Lake Powell	Glen Canyon Dam to Pearce Ferry	Lake Mead	Downstream of Hoover Dam to Lake Mohave	Davis Dam to Parker Dam	Parker Dam to Imperial Dam	Imperial Dam to the SIB
					LCR MSCP Reach 1	LCR MSCP Reach 2	LCR MSCP Reach 3	LCR MSCP Reach 4, 5	LCR MSCP Reach 6, 7
Native Species									
bluehead sucker	*Pantosteus discobolus*	RWCS, BLM	Present in small numbers in reservoir and river inflows [1]	Abundant in tributaries and tributary mouths, present in mainstem [2]	Occasionally found at the Colorado River inflow area. [3, 4, 5]	Absent	Absent	Absent	Absent
bonytail	*Gila elegans*	FE, BLM, LCR MSCP	Juveniles (<111 mm total length) stocked by UDWR in 2022; rarely captured [6]	Two individuals captured in Lees Ferry sub-reach in 2023, likely pass through the dam from Lake Powell [6]	Only one record from Las Vegas Bay. [3, 4, 5]	Repatriation efforts; limited numbers. [3, 4]	Topock Gorge and upper Lake Havasu; annual stocking; occasionally found in backwaters downstream of Davis Dam. [3, 4]	Stocked; utilize backwater and riverine habitat. [3, 4]	Absent
Colorado pikeminnow	*Ptychocheilus lucius*	FE	Rare in the reservoir; numerous in Colorado River and San Juan River inflows drifting from upstream [7, 8]	Absent	Absent	Absent	Absent	Absent	Absent

Common Name	Scientific Name	Status[a]	Lake Powell	Glen Canyon Dam to Pearce Ferry	Lake Mead	Downstream of Hoover Dam to Lake Mohave	Davis Dam to Parker Dam	Parker Dam to Imperial Dam	Imperial Dam to the SIB
					LCR MSCP Reach 1	LCR MSCP Reach 2	LCR MSCP Reach 3	LCR MSCP Reach 4, 5	LCR MSCP Reach 6, 7
desert pupfish	*Cyprinodon macularius*	FE	Absent	Absent	Absent	Absent	Present in isolated, constructed refuge ponds [20]	Present in isolated, constructed refuge ponds [20]	Absent
flannelmouth sucker	*Catostomus latipinnis*	RWCS, BLM, LCR MSCP	Present in small numbers in reservoir and river inflows [7]	Common in river and abundant in tributaries during spring spawning [3]	Found within the Colorado River Inflow area and occasionally near the Virgin River inflow area. [3, 4, 5, 9]	Absent	Introduced with numbers declining. [3, 4, 10]	Absent	Absent
humpback chub	*Gila cypha*	FT, LCR MSCP	Absent	Found as aggregations in river; abundant from Havasu Rapid to Pearce Ferry and in and around the Little Colorado River; introduced into Shinumo, Havasu, Bright Angel creeks [3]	Colorado River inflow area within full-pool of Lake Mead. [3, 4, 5]	Absent	Absent	Absent	Absent

Common Name	Scientific Name	Status[a]	Lake Powell	Glen Canyon Dam to Pearce Ferry	Lake Mead	Downstream of Hoover Dam to Lake Mohave	Davis Dam to Parker Dam	Parker Dam to Imperial Dam	Imperial Dam to the SIB
					LCR MSCP Reach 1	LCR MSCP Reach 2	LCR MSCP Reach 3	LCR MSCP Reach 4, 5	LCR MSCP Reach 6, 7
razorback sucker	*Xyrauchen texanus*	FE, BLM, LCR MSCP	Found in small numbers in reservoir; common in river inflows [7]	Rare upstream of Lava Falls Rapid; found in small numbers downstream of Lava Falls; translocated into the Little Colorado River and Havasu Creek by NPS/FWS [23]; move upstream from Lake Mead [3]	Sustained populations at the Colorado River inflow area, Las Vegas Bay, Echo Bay, and Overton Arm. Less abundant in Bonelli Bay. [3, 4, 5, 9]	Repatriated population; annual stocking; genetic refuge population. [3, 4, 11]	Stocked and found downstream of Davis dam to Lake Havasu. [3, 4, 10]	Stocked; utilize backwater and riverine habitat. [3, 4]	Absent
speckled dace	*Rhinichthys osculus*	BLM	Found in small numbers along shoreline and canyons especially near inflow [12]	Numerous along shoreline, in midchannel riffles, debris fans, tributaries and tributary mouths [3]	Numerous and common in Colorado River within the full-pool of Lake Mead. [3, 4]	Absent	Absent	Absent	Absent

Common Name	Scientific Name	Status[a]	Lake Powell	Glen Canyon Dam to Pearce Ferry	Lake Mead	Downstream of Hoover Dam to Lake Mohave	Davis Dam to Parker Dam	Parker Dam to Imperial Dam	Imperial Dam to the SIB
					LCR MSCP Reach 1	LCR MSCP Reach 2	LCR MSCP Reach 3	LCR MSCP Reach 4, 5	LCR MSCP Reach 6, 7
High-risk Nonnative Species									
brown trout	*Salmo trutta*	SRSF	Absent	found in increasing numbers in Lees Ferry with evidence of reproduction; found in tributaries including Bright Angel Creek and in the mainstem but abundance uncertain; considered very high risk nonnative between GCD and Pearce Ferry[24,253]	Absent	Absent	Absent	Absent	Absent
flathead catfish	*Pylodictis olivaris*	SRSF	Absent	Absent	Absent	Absent	Common in Lake Havasu, and below Parker Dam. [22]	Present [13, 14, 15]	Present [14, 15]

Common Name	Scientific Name	Status[a]	Lake Powell	Glen Canyon Dam to Pearce Ferry	Lake Mead	Downstream of Hoover Dam to Lake Mohave	Davis Dam to Parker Dam	Parker Dam to Imperial Dam	Imperial Dam to the SIB
					LCR MSCP Reach 1	LCR MSCP Reach 2	LCR MSCP Reach 3	LCR MSCP Reach 4, 5	LCR MSCP Reach 6, 7
green sunfish	*Lepomis cyanellus*	SRSF	Popular sportfish [7, 17]	Found in increasing numbers in Lees Ferry with evidence of reproduction; found downstream in tributaries and warm tributary mouths; considered dangerous invasive to native fish [3]	Common throughout. [4, 6, 8]	Common [5, 8]	Common [1, 7]	Common [9]	Presence is likely in unknown abundance. [3, 16]
rainbow trout	*Oncorhynchus mykiss*	SRSF	Absent	Popular sportfish in Lees Ferry sub-reach, high risk to native fish in downstream of Glen Canyon Dam to Pearce Ferry [2]	Occasionally found at the Colorado River inflow area. [3, 5]	Stocked and popular sportfish. [3, 18]	Stocked and popular sportfish. [3, 10]	Absent	Absent

Common Name	Scientific Name	Status[a]	Lake Powell	Glen Canyon Dam to Pearce Ferry	Lake Mead	Downstream of Hoover Dam to Lake Mohave	Davis Dam to Parker Dam	Parker Dam to Imperial Dam	Imperial Dam to the SIB
					LCR MSCP Reach 1	LCR MSCP Reach 2	LCR MSCP Reach 3	LCR MSCP Reach 4, 5	LCR MSCP Reach 6, 7
smallmouth bass	*Micropterus dolomieu*	SRSF	Popular sportfish; Introduced by UDWR as 500 fry in 1982. [7, 17]	Found in increasing numbers in Lees Ferry with evidence of reproduction; found downstream but abundance uncertain; considered dangerous invasive to native fish [3]	Popular sportfish. [3, 5, 9, 18]	Popular sportfish. [3, 11, 18]	Popular sportfish. [3, 10, 16]	Popular sportfish. [13, 15]	Unknown
walleye	*Sander vitreus*	SRSF	Popular sportfish [7, 17]	Found in small numbers immediately downstream of the dam and increasing captures through Grand Canyon; considered dangerous invasive to native fish [3]	Occasionally found at the Colorado River inflow area. [9,18]	Absent	Absent	Absent	Absent

Common Name	Scientific Name	Status[a]	Lake Powell	Glen Canyon Dam to Pearce Ferry	Lake Mead	Downstream of Hoover Dam to Lake Mohave	Davis Dam to Parker Dam	Parker Dam to Imperial Dam	Imperial Dam to the SIB
					LCR MSCP Reach 1	LCR MSCP Reach 2	LCR MSCP Reach 3	LCR MSCP Reach 4, 5	LCR MSCP Reach 6, 7
Nonnative, Regulated Sportfish									
black bullhead	*Ameiurus melas*	SRSF	Common along shorelines near rock structure and brush piles [7, 17]	Found in small numbers along shorelines and in warm tributaries and tributary mouths [3]	Common throughout. [3, 5, 9, 18]	Presumed rare. [3, 18]	Absent	Absent	Absent
black crappie	*Pomoxis nigromaculatu s*	SRSF	Popular sportfish [7, 17]	Rare [3]	Popular sportfish. [3, 5, 9, 18]	Presumed rare. [3, 13]	Absent	Present in low abundance. [13, 14, 15]	Presence is likely in unknown abundance. [3, 16]
bluegill	*Lepomis macrochirus*	SRSF	Popular sportfish [7, 17]	Rare [3]	Popular sportfish and common throughout. [3, 5, 9, 18]	Popular sportfish; common. [3, 11, 18]	Common sportfish. [3, 10, 16]	Common [13, 14, 15]	Presence is likely in unknown abundance. [3, 16]
channel catfish	*Ictalurus punctatus*	SRSF	Popular sportfish [7, 17]	Found in small numbers along shorelines and in warm tributaries and tributary mouths [3]	Popular sportfish. [3, 5, 9, 18]	Popular sportfish. [3, 11, 18]	Common sportfish. [3, 10, 16]	Present in low abundance. [14, 15]	Presence is likely in unknown abundance. [3, 16]
fathead minnow	*Pimephales promelas*	FS	Abundant as an introduced forage species	Abundant as an introduced forage species	Abundant as an introduced forage species	Abundant as an introduced forage species	Abundant as an introduced forage species	Abundant as an introduced forage species	Abundant as an introduced forage species

Common Name	Scientific Name	Status[a]	Lake Powell	Glen Canyon Dam to Pearce Ferry	Lake Mead	Downstream of Hoover Dam to Lake Mohave	Davis Dam to Parker Dam	Parker Dam to Imperial Dam	Imperial Dam to the SIB
					LCR MSCP Reach 1	LCR MSCP Reach 2	LCR MSCP Reach 3	LCR MSCP Reach 4, 5	LCR MSCP Reach 6, 7
gizzard shad	*Dorosoma cepedianum*	FS	Abundant	Rare, some entrainment through Glen Canyon Dam [23]	Abundant	Present	Present	Present	Present
golden shiner	*Notemigonus crysoleucas*	FS	Rare, sometimes used as baitfish by anglers	Rarely encountered [21]	Rare or absent	Absent	Absent	Absent	Absent
largemouth bass	*Micropterus salmoides*	SRSF	Popular sportfish [7, 17]	Found below Glen Canyon Dam and in small numbers throughout Grand Canyon 3, [23]in tributaries and warm tributary mouths [3]	Popular sportfish. [3, 5, 9, 18]	Popular sportfish. [3, 11, 18]	Common sportfish. [3, 10, 16]	Popular sportfish. [13, 14, 15]	Presence is likely in unknown abundance. [3, 16]
northern pike	*Esox lucius*	SRSF	Rare [7]	Absent	Absent	Absent	Absent	Absent	Absent
plains killifish	*Fundulus zebrinus*	FS	Abundant as an introduced forage and mosquito control species	Abundant as an introduced forage and mosquito control species	Abundant as an introduced forage and mosquito control species	Abundant as an introduced forage and mosquito control species	Abundant as an introduced forage and mosquito control species	Abundant as an introduced forage and mosquito control species	Abundant as an introduced forage and mosquito control species
red shiner	*Cyprinella lutrensis*	FS	Abundant as an introduced forage species	Abundant as an introduced forage species	Abundant as an introduced forage species	Abundant as an introduced forage species	Abundant as an introduced forage species	Abundant as an introduced forage species	Abundant as an introduced forage species

Common Name	Scientific Name	Status[a]	Lake Powell	Glen Canyon Dam to Pearce Ferry	Lake Mead	Downstream of Hoover Dam to Lake Mohave	Davis Dam to Parker Dam	Parker Dam to Imperial Dam	Imperial Dam to the SIB
					LCR MSCP Reach 1	LCR MSCP Reach 2	LCR MSCP Reach 3	LCR MSCP Reach 4, 5	LCR MSCP Reach 6, 7
striped bass	*Morone saxatilis*	SRSF	Popular sportfish [7, 17]	Found in small numbers that likely pass through the dam; upstream migration from Lake Mead 3	Popular sportfish. [3, 5, 9, 18]	Popular sportfish. [3, 11, 18]	Common sportfish. [3, 10, 16]	Present in low abundance. [14, 15]	Presence is likely in unknown abundance. [3, 16]
redear sunfish	*Lepomis microlophus*	SRSF	Absent	Absent	Absent	Absent	Common sportfish. [3, 10, 16]	Common. [13, 14, 15]	Presence is likely in unknown abundance. [3, 16]
threadfin shad	*Dorosoma petenense*	FS	Abundant as an introduced forage species for primarily striped bass	Generally absent, but found occasionally as escaped through Glen Canyon Dam	Present as an introduced forage species	Present	Present	Present	Unknown
white crappie	*Pomoxis annularis*	SRSF	Popular sportfish [7]	Rare	Absent	Absent	Absent	Absent	Absent
yellow bullhead	*Ameiurus natalis*	SRSF	Common along shorelines near rock structure and brush piles [7, 17]	Rare in the mainstem; few records in the Little Colorado River [19]	Less common than black bullhead. [3, 5]	Present in low relative abundance. [3, 11]	Present in Lake Havasu. [22]	Absent	Absent

aRWCS = Rangewide Conservation Species; FE = Federally endangered; FT = Federally threatened; SRSF = State regulated sportfish; BLM = Bureau of Land Management special status species; LCR MSCP = Lower Colorado River Multi-Species Conservation Program covered species; FS = forage species; mm = millimeters.
[1]UDWR 2006, [2]Kegerries et al. 2020, [3]Pacey and Marsh 1998, [4]Bestgen et al. 2020, [5]Pennock et al. 2023, [6]FWS 2024, [7]Mueller et al. 2001, [8]FWS 2022, [9]NDOW 2023, [10]NDOW 2020, [11]NDOW 2022, [12]Pennock et al. 2021, [13]Ham 2023a, [14]Ham 2023b, [15]Ramey and Ham 2024, [16]Mueller and Marsh 2002, [17]UDWR 2023, [18]Renner and Day 2022, [19]Marsh and Douglas 1997, [20] FWS 2012, [21]Valdez and Ryel 1995, [22]FWS 2018, [23]Emily Omana-Smith, NPS, personal communication, 2025, [24]NPS 2018, [25]NPS 2021

Preliminary analyses of long-term trends in limnological data reveal a directional change in several parameters in Lake Powell in recent decades (Deemer et al. 2023). These initial data analyses suggest an increase in surface phytoplankton biovolume (algal biomass) that may be paired with changes in community composition and could have cascading effects on ecosystem function and water quality management. For example, cyanobacteria genera across all sites have appeared to shift considerably from the beginning of monitoring (1993–1997) compared to recent years (2017–2021). These community shifts could translate into changes in algal toxin formation and/or altered edibility for aquatic grazers. With growing occurrences of harmful algal blooms, these could mean critical changes to Lake Powell water quality with respect to human use and aquatic life.

Altogether, there are six species of native fish analyzed in Lake Powell, including bonytail, bluehead sucker, Colorado pikeminnow, flannelmouth sucker, razorback sucker, and speckled dace (**Table TA 8-2**; Mueller et al. 2001). Bonytail were intentionally introduced into Lake Powell in 2022 by the Utah Division of Wildlife Resources (Smith 2022), and fish stocked upstream in the Upper Colorado and Green rivers have moved into the reservoir (FWS 2024). Colorado pikeminnow were found in the lake as it was reaching maximum pool elevation in 1980; these fish were likely migrating upstream to spawning sites on the Colorado River (Persons et al. 1982; Valdez et al. 1982). Today, Colorado pikeminnow in the lake are found mostly in the Colorado River and San Juan River inflows (Pennock et al. 2024; FWS 2022). Bluehead sucker and flannelmouth sucker are present in small numbers in the reservoir and river inflows, and razorback sucker are found in small numbers in the reservoir, but are more common in the river inflows (Schleicher et al. 2024). Durst and Francis (2016) reported four razorback suckers moving from the San Juan Arm across Lake Powell and up the Colorado River, indicating that there is movement across the Basin. The speckled dace is found in small numbers along the shoreline and canyons of the lake. Humpback chub were found in Lake Powell as the reservoir was filling and inundating the lower Cataract Canyon population (Valdez et al. 1982), but the species is not a lake species and has not persisted in the reservoir (Valdez 1990; Mueller et al. 2001).

There are at least 17 nonnative fish species in Lake Powell, including 12 state-regulated sportfish and 5 forage fish species (**Table TA 8-2**). Lake Powell is one of the more popular warm-water recreational lake fisheries in the southwestern United States. The most common and popular sportfish caught by anglers are largemouth bass, black crappie, channel catfish, smallmouth bass, striped bass, walleye, bluegill, green sunfish, and bullheads (UDWR 2023).

Two species of nonnative bivalves have been discovered in Lake Powell: the Asian clam (*Corbicula fluminea*) and the quagga mussel. The Asian clam was first found in Lake Powell in 1991 (USGS 2024), and the quagga mussel was found in 2012 (Verde 2017). Both species are filter feeders that can alter the composition of plankton communities in lakes and reservoirs. Zebra mussels (*Dreissena polymorpha*), a close relative of the quagga mussel, have been implicated in promoting *Microcystis* in noneutrophic waters (Verde 2017). *Microcystis* is a genus of freshwater cyanobacteria that includes the harmful algal bloom-forming *Microcystis aeruginosa*. Many members of a *Microcystis* community can produce neurotoxins and hepatotoxins, such as microcystin and cyanopeptolin, that can impose a human health risk when consumed with untreated water.

Quagga mussels can disrupt the pelagic arm of the food web by interfering with the link between phytoplankton and herbivorous zooplankton (Verde 2017). This will likely have negative impacts on pelagic fish, such as striped bass and their main food source, threadfin shad (*Dorosoma petenense*), which rely on midwater plankton as a food source. Quagga mussels may also boost benthic productivity in the littoral zone by diverting nutrients from the water column to the benthos following filtration and fecal output. This may have positive impacts on littoral fishes, such as largemouth bass, smallmouth bass, bluegill, and green sunfish.

Glen Canyon Dam to Pearce Ferry at Lake Mead

The Colorado River from Glen Canyon Dam to Pearce Ferry at Lake Mead spans approximately 295 miles through Arizona (USGS 2023). As the Colorado River exits Glen Canyon Dam, it flows through a series of rugged and steep canyons, including Glen Canyon (dam to Paria River), Marble Canyon (Paria River to Little Colorado River), and Grand Canyon (Little Colorado River to Grand Wash Cliffs, RM 276). Pearce Ferry (RM 280) is 4 miles downstream of Grand Wash Cliffs. The river corridor in this reach is included in or borders Glen Canyon National Recreation Area, Grand Canyon National Park, the Hualapai Indian Reservation, the Navajo Nation, and Lake Mead National Recreation Area. This reach of the Colorado River has several major tributaries, including the Paria River, Little Colorado River, Clear Creek, Bright Angel Creek, Shinumo Creek, Tapeats Creek, Deer Creek, Kanab Creek, Havasu Creek, Diamond Creek, and Spencer Creek, along with numerous ephemeral tributaries.

Since the closure of Glen Canyon Dam in 1963, the Grand Canyon reach of the Colorado River has supported few native fish species due to extreme seasonal changes in flow and temperature, with substantial ecological shifts occurring in this reach(see **TA 8 Attachment 1** for details). The fish community is influenced by the river's complex flow patterns, varying water temperatures, and a diversity of habitats reflective of shoreline geology (Valdez and Ryel 1995). Today, five of the originally common species are known to persist in the Grand Canyon reach (**Table TA 8-2**). Native fish species analyzed in this reach include the endangered razorback sucker, the threatened humpback chub, as well as bluehead sucker, flannelmouth sucker, and speckled dace (NPS 2023a) (**Table TA 8-2**). Colorado pikeminnow, roundtail chub, and bonytail are extirpated from this reach. However, two bonytail have recently been detected in the reach and most likely moved from Lake Powell through Glen Canyon Dam. A proposal was also developed to evaluate the feasibility of stocking Colorado pikeminnow in the Grand Canyon, aiming to boost the recovery and resilience of this endangered species. An expert science panel reviewed current river conditions and concluded that the Grand Canyon now offers suitable habitat for adult and subadult pikeminnow, though limited juvenile survival may constrain recruitment. The panel recommended that experimental stocking, monitoring, and further research could help determine the viability of reintroduction in this changing environment (Dibble et al. 2023).

Nonnative fish species include rainbow trout, brown trout, largemouth bass, green sunfish, channel catfish, fathead minnow (*Pimephales promelas*), smallmouth bass, and other less common species (NPS 2023a; Shollenberger et al. 2025) (**Table TA 8-2**). Results from 2024 sampling (Shollenberger et al. 2025) indicate that smallmouth bass detections were highest between Glen Canyon Dam and the Paria River, with capture rates decreasing substantially downstream; however, the majority of sampling efforts were concentrated within the first 24 river miles downstream of Glen Canyon Dam.

Electrofishing and netting efforts confirmed that smallmouth bass were rare in the lower reaches, with the most downstream detection at RM 16.4 in 2023 and 15.27 in 2024. The furthest downstream eDNA detection was at RM 37.5. Juvenile smallmouth bass were captured in 2022 and 2023 in the Lees Ferry sub-reach, which indicated reproduction (Reclamation 2025). In 2024, experimental low temperature dam releases (cool mix flows) were initiated to prevent the establishment of a smallmouth bass population below Glen Canyon Dam (Reclamation 2024). No smallmouth bass nests were observed, no young-of-year were captured, and no evidence of recruitment was detected in 2024 following that experiment (Reclamation 2025). A significant increase in green sunfish captures was observed during the fall of 2023 compared to previous years, suggesting a strong recruitment and/or entrainment event. However, cooler conditions in 2024 likely prevented a similar influx of juvenile green sunfish, as suggested by lower capture rates in fall months. Green sunfish also exhibited reduced recruitment and/or entrainment and higher average sizes in 2024, suggesting minimal reproduction (Reclamation 2025). Entrainment modeling (Eppehimer et al. 2025) shows that the risk of smallmouth bass invasion below Glen Canyon Dam increases as Lake Powell elevations drop below 3,530 feet, and is especially hard to control below 3,510 feet with penstock releases. Keeping reservoir elevations above 3,570 feet can substantially reduce this risk. Lower lake levels also result in warmer downstream water, which favors nonnative fish that threaten native and Endangered Species Act (ESA)-listed species. If smallmouth bass establish downstream, removal would be difficult and costly (Eppehimer et al. 2025).

Flannelmouth sucker and bluehead sucker are "conservation species" that are included in a range-wide conservation agreement among six states (UDWR 2006). These species are found as self-sustaining populations and are locally common in the Colorado River from Glen Canyon Dam to Lake Mead (Kegerries et al. 2020). They are seasonally abundant during spring spawning runs into tributaries of the Colorado River, such as the Paria River, Havasu Creek, Kanab Creek, and Tapeats Creek. Both species have adjusted to changing riverine conditions following the construction of Glen Canyon Dam (Paukert and Rogers 2004; Valdez and Carothers 1998). Speckled dace are common to abundant within tributaries and near tributary inflows, as well as on rocky debris fans formed by debris flows from side canyons (Valdez and Ryel 1995; Kegerries et al. 2020). Flannelmouth sucker and bluehead sucker, abundance typically increases with distance downstream of the Little Colorado River and is generally high near major tributary confluences, such as the Little Colorado River, Paria River, Kanab Creek, and Bright Angel Creek (Johnstone et al. 2003; Johnstone and Lauretta 2004; Rogowski et al. 2017).

The humpback chub population expanded in the western Grand Canyon in 2016. This is likely due to the warmer water and the lack of predators in the lower Grand Canyon (Dzul et al. 2023; Van Haverbeke et al. 2017; Rogowski et al. 2018; Kegerries et al. 2020; Rogers et al. 2023). As of 2022, a growing population of 50,000-84,000 adults between Havasu Rapids and Pearce Ferry was reported (Van Haverbeke et al. 2023) and considered a single genetic population (Dzul et al. 2025). Since 2000, larvae and adult razorback suckers have also been found in the Colorado River inflow at the lower end of the Grand Canyon (and within Lake Mead proper), including sonic-tagged adults moving from one of the three Lake Mead populations (Kegerries et al. 2017a, 2017b). A confirmed spawning site was discovered in 2010 about 10 miles downstream of Pearce Ferry (Albrecht et al. 2010; Valdez et al. 2012; Albrecht et al. 2017). Although annual surveys for larval and small-bodied razorback suckers have been conducted in the Grand Canyon since 2014, no small-bodied fish have been detected, and larval fish have not been detected since 2019. The razorback sucker in the Grand Canyon is found primarily downstream of Lava Falls Rapid, with individuals rarely found upstream (Kegerries et al. 2020; Rogers et al. 2023). In 2024, 790 passive integrated transponder (PIT) -tagged razorback sucker were stocked into the Little Colorado River, with 70 being detected near the confluence with the Colorado River and another 6 being recaptured in the Little Colorado River and the mainstem (Ward and Wood 2025).

A total of 18 nonnative fish species have been reported between Glen Canyon Dam and Lake Mead from 1957 through 2023 (Valdez and Ryel 1995; Trammell et al. 2002; Fonken et al. 2023; Fennell et al. 2024). Native fish continue to dominate abundance, as total fish numbers, in the Grand Canyon fish community (Kegerries et al. 2020; Dzul et al. 2023). Small-bodied, nonnative fish like the fathead minnow, red shiner (*Cyprinella lutrensis*), and plains killifish *(Fundulus zebrinus*) are predominantly found downstream of the Little Colorado River confluence. Nonnative fish, including the golden shiner (*Notemigonus crysoleucas*), redside shiner, striped bass, and threadfin shad, infrequently occur in this reach. Evidence strongly indicates that this tributary is the primary source of these species in the Colorado River ecosystem (Johnstone et al. 2003), although additional sources include entrainment through Glen Canyon Dam, upstream movement from Lake Mead, and other tributaries (Rogowski et al. 2017).

The primary sportfish in the Colorado River between Glen Canyon Dam and Lake Mead inflow is the rainbow trout, which is maintained by the NPS and Arizona Game and Fish Department (AZGFD) in Lees Ferry as a recreational fishery. Natural reproduction of rainbow trout that move downstream in the Grand Canyon is dependent on cool water temperatures, access to tributaries for spawning, and continued availability of suitable mainstem habitat. These variables are directly affected by patterns and temperatures of water released from Lake Powell. McKinney and Speas (2001) found that *Gammarus*, chironomids, and Cladophora constituted about 90 percent of rainbow trout's food by volume during cold releases. With more recent warm releases, the food of rainbow trout has shifted to a greater variety of invertebrate and algal species, with midges and blackflies as a large component of the diet (Cross et al. 2013).

Humpback chub rely on *Gammarus* and chironomids, but also consume larval simuliids (blackflies), which become more common downstream of the Paria River (Gloss et al. 2005). *Cladophora*, *Oscillatoria* spp., and terrestrial organic matter serve as key energy sources for aquatic invertebrates between Glen Canyon Dam and Lake Mead. *Cladophora* and *Oscillatoria* are also consumed by fish, largely because of the dense populations of lipid-rich diatoms (Gloss et al. 2005).

In 2012, the quagga mussel was found in Lake Powell (Verde 2017) and was identified in sampling locations between Glen Canyon Dam and Lees Ferry in November 2014. Mussels continue to be found in the river downstream of the dam. Their distribution is patchy and highly influenced by fluctuating water levels and location-specific flow regimes (NPS 2023b). Kennedy (2007) reported that the risk of quagga mussels establishing within the Grand Canyon is generally low, except for the Lees Ferry tailwater reach, where the risk is high. Conditions within the Lees Ferry tailwater generally appear suitable for quagga mussel establishment, except for the high average water velocity. High suspended sediment, high ratios of suspended inorganic to organic material, and high-water velocities limit the quagga mussel's ability to establish at high densities within the Grand Canyon or its tributaries. Additionally, the rapids in the Grand Canyon contribute to high mortality rates for larval quagga mussels, limiting their ability to disperse and colonize downstream reaches.

The New Zealand mud snail has rapidly spread across the western United States and was first detected in the tailwaters downstream of Glen Canyon Dam in March 2002. Further investigations in the Grand Canyon revealed its presence more than 225 miles downstream from the dam (Cross et al. 2013), suggesting that it had been introduced several years prior. In suitable habitats, mud snail populations can exceed densities of 100,000 individuals per square meter. The species is known to cause ecological changes, including alterations in primary production and reductions in native invertebrate populations in the rivers it invades. New Zealand mud snail populations can increase quickly with suitable temperatures and riverine conditions and are of concern due to their potential impacts on native species, fisheries, and food web dynamics within the Grand Canyon (Cross et al. 2013; NPS 2015).

LCR MSCP Reaches

The LCR MSCP planning area comprises areas up to and including the full-pool elevations of lakes Mead (1,229 feet), Mohave (647 feet), and Havasu (450 feet), and the historical floodplain of the Colorado River from Lake Mead to the SIB.

For purposes of the Fish and Other Aquatic Species section of this document, we utilize the following reaches, as defined within LCR MSCP (2004):

Reach 1—from Separation Canyon (Grand Canyon RM 240) in the lower end of the Grand Canyon to Hoover Dam (Colorado RM 342.2), including Lake Mead up to full-pool elevation.

Reach 2—from Hoover Dam to Davis Dam (RM 276), including Lake Mohave up to full-pool elevation.

Reach 3—from Davis Dam (RM 276) to Parker Dam (RM 192.3), including Lake Havasu up to full-pool elevation.

Reach 4—from Parker Dam (RM 192.3) to Adobe Ruin and Bureau of Reclamation (Reclamation) Cibola Gage (RM 87.3) at the lower end of Reclamation's maintenance Cibola Division.

Reach 5—from Reclamation Cibola Gage (RM 87.3) to Imperial Dam (RM 49.2).

Reach 6—from Imperial Dam (RM 49.2) to the Northerly International Boundary (RM 23.1); and

Reach 7—portion of the Lower Colorado River from Northerly International Boundary (RM 23.1) to SIB (RM 0.0) within the United States.

Note that due to the nature of the river downstream of LCR MSCP Reaches 4 and 5, particularly the scarcity of native and ESA-listed fishes and their survival, along with the current allocation of LCR MSCP fish sampling efforts primarily in Reaches 4–5 (J. Stolberg, LCR MSCP, personal communication; LCR MSCP 2024), we will present information from Reaches 4–5 as the lowest-most combined reach routinely sampled (from Parker Dam [RM 192.3] to Imperial Dam [RM 49.2]). We will not present data for Reaches 6–7 of the LCR MSCP.

LCR MSCP Reach 1- Lake Mead

Lake Mead has four major basins that tend to be separated by narrow connective canyons. In order from upstream to downstream, these include Gregg Basin, Virgin Basin, Boulder Canyon, and Boulder Basin, and can be classified as deep, subtropical, monomictic basins that become thermally stratified at times and typically for a brief period in July. The Muddy (Moapa) River and the Virgin River both discharge into the Overton Arm of the Virgin Basin, and Las Vegas Wash, which carries treated wastewater from the Las Vegas area, discharges into Las Vegas Bay of the Boulder Basin. These are considered the only perennial tributaries of Lake Mead (Baker et al. 1977).

The full-pool footprint of Lake Mead extends up to just upstream of Separation Canyon (RM 240 within the Grand Canyon) and encompasses the inflow area of the Colorado River into Lake Mead. For this document, given the current conditions at Pearce Ferry Rapid, we use Pearce Ferry Rapid to separate the Grand Canyon from Lake Mead. We also note that Lake Mead, when full, encompasses various portions of the inflow areas of the Muddy River, the Virgin River, and Las Vegas Wash, all of which are perennial tributaries to the reservoir. We note that these inflow areas are highly dynamic in nature, sometimes are riverine and at other times are lake-like and provide diverse and important habitats to both native and nonnative fishes, resulting in a robust fish community at Lake Mead inflow areas (Albrecht et al. 2017).

Lake Mead has a diverse fish community (**Table TA 8-2**) (Renner and Day 2022), as would be expected from the largest reservoir by volume within the United States (Rosen et al. 2012). Among all the ESA-listed fishes native to the Colorado River, the razorback sucker is the most prominent species in Lake Mead. This reservoir hosts a unique population of wild, naturally recruiting razorback suckers. At the time of writing, Lake Mead is the only known location supporting a recruiting population of this species (Albrecht et al. 2010, 2020; Rogers et al. 2025).

This document analyzes species routinely and recently documented by the Nevada Department of Wildlife (NDOW) and AZGFD. To that end, **Table TA 8-2** provides a summary of fish species likely to be routinely present within the analysis area since Hoover Dam closed. Following the closure of Hoover Dam, NDOW stocked Lake Mead with largemouth bass and sunfishes (*Lepomis* spp.), establishing a popular bass fishery that began to decline by the 1940s. To improve sportfish conditions, threadfin shad were introduced in 1954 as a forage species.

The construction of upstream impoundments, including Lake Powell in 1963, altered flow regimes and water storage patterns, resulting in lower spring flows and higher winter flows, which negatively affected largemouth bass spawning and affected temperature and nutrient dynamics in Lake Mead (NDOW 2023). In response to the declining bass fishery, coldwater fish species such as rainbow trout, cutthroat trout (*Oncorhynchus clarkii)*, hybrid bowcutt trout (*Oncorhynchus mykiss x Oncorhynchus clarkii)*, silver salmon (*Oncorhynchus kisutch)*, and striped bass were introduced between 1969 and 1972, with striped bass establishing a self-sustaining population. Since surveys began in 1991, nonnative species have been regularly captured in Lake Mead, including striped bass, channel catfish, and common carp. NDOW documented the first walleye in 2001 and gizzard shad (*Dorosoma cepedianum*) in 2009. Red shiners continue to be routinely observed by researchers (Pennock et al. 2023; Rogers et al. 2025). Smallmouth bass, first detected in 1999, have since become as abundant as largemouth bass despite not being officially stocked. Today, Lake Mead's sport fishery includes striped bass, largemouth bass, smallmouth bass, channel catfish, and a small population of black crappie (NDOW 2023).

The Muddy River and Virgin River inflows are not only used to varying degrees by razorback suckers but also have the potential to contain Virgin River chub (*Gila seminuda*), woundfin (*Plagopterus argentissimus*) (Virgin River only at present), flannelmouth sucker (Virgin River only at present), desert sucker (*Catostomus clarkii*) (Virgin River only at present), as well as speckled dace (Virgin River only at present), a likely extirpated subspecies of speckled dace, the Moapa speckled dace (*Rhinichthys osculus moapae*) (Muddy River), Moapa dace (*Moapa coriacea*) (Muddy River), populations of Moapa White River springfish (*Crenichthys baileyi moapae*) (Muddy River), and various headwater associated thermal spring associated endemic macroinvertebrates (Muddy River). Most of these species are documented in reaches higher up within these two river systems, as are Virgin spinedace (L*epidomeda mollispinis*), which is primarily a tributary-oriented species found within the tributaries to the Virgin River mainstem (e.g., Holden et al. 2005; Kegerries et al. 2022; Handtke et al. 2024 for details on the Muddy River fish community, Albrecht et al. 2020 and Rogers et al. 2025 for details on the lower Virgin River fish Community). In recent years, only Virgin River chub have been captured from within Lake Mead proper, a single specimen originating from within the Overton Arm (Rogers et al. 2025).

Rogers et al. (2024 and previous annual reports) discuss Lake Mead razorback sucker history and trends since before dam construction. A study initiated in 1996 has focused on long-term monitoring of the Lake Mead razorback sucker population (Rogers et al. 2025). This effort has continued through 2024, providing a consistent database and expanding sampling to new areas. By using data gathered from sonic-tagged fish in conjunction with long-term trammel-netting and larval-sampling data, information regarding primary spawning sites has been obtained for three long-term monitoring study areas (Las Vegas Bay, Echo Bay, and the Virgin River/Muddy River inflow area), and research sites (the Colorado River inflow area and Bonelli Bay) within Lake Mead. Along with primary spawning site information, sonic-tagged fish have confirmed reservoir-wide and seasonal movement patterns within the long-term monitoring and research study areas.

Razorback suckers continue to spawn and recruit in Lake Mead (e.g., Rogers et al. 2025). Multiple age classes of razorback suckers, including larvae, juveniles, and adults, are documented annually. The Lake Mead razorback sucker population consists of individuals ranging from 2–30+ years old, demonstrating relatively high growth rates indicative of a young, recruiting population.

It is important to note that since 1966, at least some razorback sucker recruitment has occurred nearly annually in Lake Mead. This suggests that lake elevations observed to date have not prevented this population from successfully recruiting. However, it is unclear whether a lower end threshold exists, either defined bathymetrically by suitable spawning and recruitment habitat availability, or because of other recruitment-limiting factors at lake elevations lower than observed to date.

Recently, both larvae and adult razorback suckers have been discovered utilizing both the lower Grand Canyon and the inflows to Lake Mead; razorback suckers have been documented to be spawning at least 100 river miles from Lake Mead consistently between 2014 and 2019 (Albrecht et al. 2014; Kegerries et al. 2017a, 2017b, 2020). Recent data from the Colorado River inflow (Rogers et al. 2023) suggests that Lake Mead razorback suckers move throughout not only Lake Mead, but also up into the Las Vegas Wash, and into the Grand Canyon when conditions allow. Major findings for this and previous studies to date include (1) multiple age-classes of unmarked, wild razorback suckers (including juvenile fish) occupy the Colorado River inflow, and adults spawn there; (2) razorback suckers spawn within Grand Canyon or its associated tributaries; (3) sonic-tagged razorback sucker (stocked and wild) utilize both the Colorado River inflow and Grand Canyon; and (4) native fish dominate the fish community in Grand Canyon despite the presence of numerous nonnative fishes in Lake Mead and Lake Powell. Within the study period (2014–2023), as described in Rogers et al. (2023), razorback sucker use of the Grand Canyon and Colorado River inflow was documented. Overall, study results indicate an apparent interconnected, recruiting population of razorback suckers in Lake Mead that may demonstrate plasticity sufficient to allow for lentic and lotic habitat use in Lake Mead and Grand Canyon.

In addition to razorback suckers, several other native fish occur in Lake Mead, including flannelmouth suckers, which were first captured in 2001, hybrid suckers (razorback x flannelmouth) documented in 2010, bluehead suckers in 2011, bonytail in 2020, and both humpback chub and Virgin River chub in 2021. Notably, humpback chub are relatively common upstream and downstream of Pearce Ferry Rapid when riverine habitats are present (Hedden et al. 2022; Pennock et al. 2023; Rogers et al. 2024).

Dibble et al. (2021) describe geomorphic changes in the lower Grand Canyon resulting from recent declines in Lake Mead's elevation, which have extended the Colorado River by more than 62 river miles and shifted habitat conditions in favor of native fish species. When the elevation of Lake Mead dropped below 1,135 feet, the river downcut through the reservoir delta outside of its historic channel, forming Pearce Ferry Rapid, a bedrock ledge that acts as a barrier to the upstream movement of nonnative fishes from Lake Mead. This physical feature has significant implications for fish populations. If water management prioritizes Lake Powell storage, Pearce Ferry Rapid is likely to persist, blocking or deterring nonnative fish movement upstream. Conversely, prioritizing Lake Mead storage could potentially inundate the rapid and lower Lake Powell elevations, which would warm the river, improving conditions for native fish, but increasing the risk of nonnative fish invasions. An intermediate approach could maintain the current balance of barriers, deterrents, and thermal conditions, supporting native fish while limiting nonnative species (Dibble et al. 2021; Bruckerhoff et al. 2022).

Fish capture and movement data of razorback sucker, flannelmouth sucker, striped bass, and common carp collected or detected between 2012 and 2023 (Kegerries and Albrecht 2013a, 2013b; Albrecht et al. 2014; Kegerries et al. 2015; Rogowski et al. 2025), validate that Pearce Ferry Rapid likely functions as a barrier to fish movement, depending on lake elevation. Upstream movement was observed when mean monthly Lake Mead elevations exceeded approximately 1,090 feet. Rogowski et al. (2025) conclude that the rapid likely effectively restricts the upstream movement of nonnative fish species, thus protecting native habitats, although it may also impede the movement of some native species, potentially creating downstream population sinks. While some native species, such as humpback chub, appear to benefit from reduced predation and competition, the rapid also appears to limit connectivity for other native fish populations (e.g., razorback sucker and flannelmouth sucker). The barrier function of Pearce Ferry Rapid is dynamic and may shift over time due to geomorphic and environmental factors, such as reservoir elevation, channel migration, or possible downcutting. Overall, the formation and persistence of Pearce Ferry Rapid demonstrates how a decline in reservoir elevation can create complex barriers with both positive and negative consequences for native fish conservation in the Grand Canyon (Dibble et al. 2021; Rogowski et al. 2025).

Quagga mussels were found in Lake Mead in January 2007, with the initial colonization probably occurring 3–4 years earlier, and are considered a biological threat to Lake Mead (NPS 2023b). Significant impacts on water quality, lake biology, and infrastructure were expected based on experiences with zebra mussels in other parts of the country, but to date, no significant impacts on either water quality or lake biota have been published that can be directly attributed to quagga mussels (Renner and Day 2022; NPS 2023b). Another invasive bivalve in Lake Mead, the Asian clam, has also been observed during razorback sucker field monitoring and research (Rogers et al. 2025), as have red swamp crayfish (*Procambarus clarkii*) (Renner and Day 2022).

Asian clams are an invasive species native to Asia that have spread widely in freshwater systems across North America and other parts of the world (Renner and Day 2022). Known for their rapid reproduction and ability to form dense populations, Asian clams can outcompete native species, disrupt local ecosystems, and clog water intake pipes, leading to substantial ecological and economic impacts (Renner and Day 2022). They were first detected in 1948 in Lake Mead (Renner and Day 2022). The New Zealand mud snail is a small, invasive freshwater snail originally from New Zealand that has spread to various parts of the world, including North America and Europe (Renner and Day 2022). These snails reproduce rapidly and can form dense colonies, outcompeting native species for food and habitat, and disrupting local ecosystems by altering nutrient cycles and food webs. They were first reported north of Lake Mead in 2007, but continue to threaten downstream habitats (Renner and Day 2022). Another invasive snail (Malaysian trumpet snail [*Melanoides tuberculata*]) has been detected in Lake Mead, as well as other locations downstream (USGS 2025).

LCR MSCP Reach 2- Downstream of Hoover Dam to Lake Mohave

The Hoover Dam Reach extends from the downstream end of Lake Mead, just downstream of Hoover Dam, and continues to Davis Dam, which impounds Lake Mohave. The extent of analysis for this reach is consistent with LCR MSCP Reach 2 and includes approximately 67 miles of riverine habitat (LCR MSCP 2004).

Lake Mohave is managed as a sport fishery by the states of Arizona and Nevada for largemouth bass, smallmouth bass, striped bass, rainbow trout, bluegill, green sunfish, channel catfish, and others (FWS 2018). Lake Mohave is operated by Reclamation as a reregulation reservoir. It fluctuates annually within a 15-foot vertical range, filling by mid-May and lowering to an annual minimum in October. Wave action redistributes sediment deposits from desert washes and shapes these deposits into sandbars or natural berms. In some areas, these sandbars isolate the lower portions of the desert washes from the lake proper, and when the lake is at full-pool, lakeside ponds form. Reclamation and its partners in the Lake Mohave Native Fish Work Group have been using these lakeside ponds since 1992 to rear razorback suckers and bonytail (LCR MSCP 2024). As such, Lake Mohave serves as a fisheries backbone for razorback sucker and bonytail management in the Lower Colorado River.

Lake Mohave supports a diverse fishery, including striped bass, stocked rainbow trout, largemouth bass, smallmouth bass, bluegill, green sunfish, channel catfish, yellow bullhead, threadfin shad, gizzard shad, and common carp (**Table TA 8-2**). The native species present include bonytail and razorback sucker (**Table TA 8-2**). The lake was historically stocked year-round with rainbow trout until hatchery issues caused a pause from October 2013 to February 2017. Striped bass, introduced in the early 1980s, have become a major part of the sport fishery, preying on the stocked rainbow trout and other forage species, including shad, whose populations have declined due to predation (NDOW 2022).

Lake Mohave is home to the most genetically diverse population of razorback sucker left in the world, and as such is an important focus of the LCR MSCP (Marsh et al. 2015; LCR MSCP 2024). Because of the importance of preserving genetic diversity, and despite persistent challenges with the survival of repatriated razorback suckers, comprehensive genetic monitoring is conducted on wild-captured larvae, individuals reared in lakeside backwaters, and the repatriate adult population. These genetic monitoring efforts are ongoing (Bestgen et al. 2020; LCR MSCP 2024). The Lake Mohave population of razorback sucker spawns annually, but recruitment has not been documented since the 1950s (Marsh and Langhorst 1988; Mueller 2006; Karam and Marsh 2010). Striped bass, as well as sunfish, may play a role in population declines throughout Lake Mohave (Karam and Marsh 2010; Kesner et al. 2012; Ehlo et al. 2017); predation of a 500 millimeters (mm) razorback sucker by striped bass was documented when an angler found an acoustic transmitter in a striped bass stomach. Sunfish have been implicated in the lack of young razorback sucker survival (FWS 2018). Renner and Day (2022) provide a historical account of all aquatic species detected in Lake Mead and Lake Mohave, but for purposes of this document, we plan to analyze species routinely and recently documented by NDOW and AZGFD.

To combat the overall lack of razorback sucker recruitment in Lake Mohave and based on work done by the Lake Mohave Native Fish Work Group (Marsh et al. 2015), the LCR MSCP has maintained a stocking/augmentation program, providing for the stocking of up to 660,000 subadult razorback suckers and up to 620,000 subadult bonytail into the Lower Colorado River over a 50-year term. Since 2005, 137,030 bonytail have been stocked into Reaches 2 through 5. In addition, 157,990 razorback suckers have been stocked into Reach 2 during this period in support of maintaining the population in Lake Mohave as a genetic refuge for the species. This rate of stocking is expected to meet LCR MSCP Fish Augmentation Program goals (Bestgen et al. 2020; LCR MSCP 2024).

To obtain sufficient numbers of young fishes for grow-out and eventual stocking, an adult broodstock for each species is maintained by the LCR MSCP. Because the adult razorback sucker population in Lake Mohave is the most genetically diverse among razorback sucker populations, it is used as the primary broodstock for this species. Under the LCR MSCP, razorback sucker larvae are captured from this stock and reared at hatcheries in Arizona and Nevada before being stocked into Lake Mohave or other places within the Lower Colorado River. A second broodstock, developed from Lake Mohave offspring, is maintained at the Southwestern Native Aquatic Resources and Recovery Center in New Mexico, with additional rearing facilities in Arizona and Nevada (LCR MSCP 2024). The Southwestern Native Aquatic Resources and Recovery Center also maintains the world's only bonytail broodstock for species propagation, supported by a genetic management plan and funding for maintenance, augmentation, and distribution. A second bonytail broodstock is held at the Mora National Fish Hatchery as a refuge population to safeguard against catastrophic events. While stocked native fishes persist in some reaches of the Lower Colorado River, low post-stocking survival rates mean augmentation efforts remain focused on improving survival outcomes (LCR MSCP 2024).

Based on the captures reported in 2023, the repatriated population of razorback sucker in Lake Mohave appears stable to slightly increasing. It is thought that a combination of higher post-stocking and adult survival accounts for recent increases in population size. Post-stocking survival of smaller (less than 400 millimeters [mm] total length) fish continues to be low, but an increase in fish released at greater than 400 mm total length in recent years may contribute to higher-than-average post-stocking survival (Reap et al. 2023; LCR MSCP 2024).

The razorback sucker population in Lake Mohave was recently estimated from two data sources: (1) trammel net capture data obtained during the annual, multi-agency March roundup and (2) remote PIT scanning data collected during the sample year. Based on trammel net capture data, the repatriate population estimate for Lake Mohave was 1,720 individuals (95 percent confidence interval [CI] from 950 to 3,440). Based on 2022–2023 remote PIT scanning, the lake-wide Lake Mohave repatriate population was estimated at 5,761 individuals (95 percent CI from 5,590 to 5,931). Subpopulation estimates using zone-specific scanning were also calculated and estimated the basin (reservoir miles 13–29) population at 3,295 individuals (95 percent CI from 3,173 to 3,416) and the river (reservoir miles 43–63) population at 3,603 individuals (95 percent CI from 3,379 to 3,826) (LCR MSCP 2024).

Reclamation and its partners in the Lake Mohave Native Fish Work Group have been using lakeside ponds since 1992 to rear razorback suckers and bonytail. The ponds are stocked with juvenile fishes each year as the reservoir fills (typically stocked in late January), and the LCR MSCP monitors and manages the ponds throughout the growing season. This work includes periodic monitoring of plankton production; removal of weeds and debris; population monitoring using remote sensing technologies; and routine monitoring of physical, chemical, and biological parameters. The ponds are normally harvested in late spring and again in the fall as the lake elevation declines. Fishes from these ponds are then released back into Lake Mohave or elsewhere within the Lower Colorado River. While annual pond harvests can be variable, lake-wide monitoring has consistently shown higher long-term contact rates for pond-reared versus hatchery-reared razorback suckers in Lake Mohave. Pond rearing provides an opportunity for fish to attain larger sizes prior to release into the lake, and contact data suggest that this larger size may improve post-stocking survival, increasing the likelihood that these fish will contribute to the adult broodstock (LCR MSCP 2024).

Bonytail, the other ESA-listed fish species within Reach 2 (Lake Mohave), were not stocked in 2024. In fact, within the past 20 years, approximately 3,500 bonytail have been stocked into Lake Mohave. This species is monitored through ongoing razorback sucker netting and PIT scanning efforts with a few capture records over time (J. Stolberg, LCR MSCP, personal communication). No recent population estimates have been calculated for bonytail in Lake Mohave (or within any other reach of the Lower Colorado River) due to limited contacts and the overall rarity in terms of survival of this species in the Lower Basin (LCR MSCP 2024).

Currently, all reservoirs, lakes, and watersheds receiving raw Colorado River water have been exposed to quagga mussels (Renner and Day 2022; CDFW 2024). Asian clams and the New Zealand mud snail continue to threaten downstream habitats (Renner and Day 2022). Renner and Day (2022) also indicate that freshwater shrimp (*Gammarus fasciatus*) and big-ear radix (*Radix auricularia)* have established populations in Lake Mohave. The invasive Malaysian trumpet snail is also known from locations downstream of Lake Mead (USGS 2025).

LCR MSCP Reach 3- Davis Dam to Parker Dam

The LCR MSCP's Reach 3 extends from Davis Dam (RM 276) to Parker Dam (RM 192.3), including Lake Havasu up to full-pool elevation. This reach is composed of a riverine component and a reservoir portion (Lake Havasu). The Bill Williams River is a notable tributary inflow draining into Lake Havasu, providing diversity in terms of inflow habitats, including potentially important aquatic habitats, areas of vegetative cover, and, at times, turbidity near the inflow area. In 2008, multiple agencies, including Reclamation, NDOW, and the Southern Nevada Water Authority, collaborated to secure the Boy Scout Camp property, incorporating it into the LCR MSCP as the Big Bend Conservation Area (LCR MSCP 2024). This effort resulted in the creation of 15 acres of backwater habitat beneficial to native fish species like razorback suckers, bonytail, and flannelmouth suckers in Reach 3 of the LCR MSCP. The area, located approximately 30 miles south of Laughlin, Nevada, is crucial for sustaining the flannelmouth sucker population. These represent some of the areas of interest of the LCR MSCP and typical habitats created by this program for the benefit of ESA-listed fishes.

Reach 3 contains ESA-listed fishes, including the razorback sucker and bonytail. An important population of flannelmouth sucker (a LCR MSCP covered species and an Arizona species of special concern species of concern) can also be found within this reach of the Lower Colorado River (**Table TA 8-2**) (LCR MSCP 2004). The Colorado River downstream of Davis Dam, to the inflow area of Lake Havasu, supports one of the largest populations of razorback sucker in the Lower Colorado River and is monitored by ongoing LCR MSCP and partner agency efforts (LCR MSCP 2024). This same area is also home to flannelmouth sucker, reintroduced in 1976 by AZGFD, and for years represented one of the most successful reintroductions of a native fish within the Lower Colorado River (Mueller and Wydoski 2004; Best and Lantow 2012). Reach 3 has also received bonytail stockings by the LCR MSCP (LCR MSCP 2024), although these bonytail stockings have produced little evidence of long-term survival, reproduction, or recruitment except for within isolated, predator-free environments (Marsh et al. 2024).

Reach 3 razorback sucker population in 2023 was estimated at 7,224 individuals (95 percent CI from 6,931 to 7,517). This is the largest population estimate for razorback suckers in Reach 3 since program implementation began in 2005. Due to the limited number of bonytail and flannelmouth suckers detected, no recent population estimates have been generated for those species (LCR MSCP 2024). Recent surveys have suggested that flannelmouth sucker captures are unfortunately becoming quite rare in the Lower Colorado River, and it remains a species of concern (J. Stolberg, LCR MSCP, personal communication; LCR MSCP 2024). A total of 2,764 razorback suckers and 2,070 bonytail were stocked into Reach 3 during 2023 (LCR MSCP 2024).

The LCR MSCP currently collects larval razorback sucker from Lake Mohave to rear at various hatcheries and lakeside ponds on Lake Mohave. The result of these efforts, as well as larvae reared at the Southwest Native Aquatic Resources and Recovery Center hatchery, provides razorback sucker of sufficient size and quantity to stock this species into Reach 3. Similarly, bonytail, from hatchery sources supported by the LCR MSCP, are also stocked into Reach 3. Capture and contact data for Reach 3 are acquired through multiple projects, ongoing multi-agency native fish roundups, and from other annual surveys conducted by LCR MSCP partners. Fall and spring netting surveys are conducted throughout Topock Gorge and upper Lake Havasu. PIT scanners are used, and active sampling within the Mohave Valley Conservation Area, a 63-acre backwater created by the LCR MSCP, is also conducted to support understanding of the fish community in Reach 3. Backwater habitats created in Reach 3 by the LCR MSCP will continue to be connected to the mainstem river to address the life history requirements of flannelmouth suckers, as well as other native fishes. Lastly, there are numerous projects dedicated to habitat creation and conservation efforts, for and on behalf of the LCR MSCP fish species in this reach, as further outlined within LCR MSCP (2024). The Colorado River downstream of Davis Dam is the lowermost reach of the river managed by the AZGFD Region III Aquatics Program. This portion of the Colorado River from Davis Dam [RM 276.0] to the Interstate 40 crossing [RM 233.9] at Topock, Arizona, corresponds with the LCR MSCP's Reach 3. Lake Havasu is managed as a sport fishery by the state of Arizona, which supports populations of largemouth bass, smallmouth bass, striped bass, bluegill, and redear sunfish (*Lepomis microlophus*). Other nonnatives present in Lake Havasu, but which are not actively managed, include green sunfish, yellow bullhead, flathead catfish, and channel catfish (**Table TA 8-2**) (FWS 2018). More specifically, during routine electrofishing trend surveys in 2024, AZGFD captured redear sunfish, bluegill, gizzard shad, largemouth bass, green sunfish, common carp, sunfish hybrids (*Lepomis* spp.), striped bass, smallmouth bass, flathead catfish, black crappie, goldfish (*Carassius auratus*), threadfin shad, yellow bullhead, and channel catfish (**Table TA 8-2**) (AZGFD 2024a). In 2021, AZGFD conducted a creel census in the upper portions of this reach from Davis Dam (RM 276) to Avi Bridge crossing (RM 257). This survey reports the fishery to primarily consist of rainbow trout, striped bass, and channel catfish. Other species included smallmouth bass, largemouth bass, sunfishes (*Lepomis* spp.), common carp, razorback sucker, and the introduced population of flannelmouth sucker. Rainbow trout is the only sportfish species currently stocked in this reach, originating from Willow Beach National Fish Hatchery, and routinely being stocked into this location (**Table TA 8-2**) (AZGFD 2023).

Sampling in the 12 miles downstream of Davis Dam indicates that the sport fishery in this area is mainly supported by striped bass, largemouth bass, smallmouth bass, redear sunfish, and regularly stocked rainbow trout (NDOW 2020). Additionally, efforts to repatriate and monitor native species, such as the razorback sucker, bonytail, and flannelmouth sucker, are ongoing in the river and its backwaters. Although flannelmouth suckers are native to the Colorado River, they are believed to have been introduced to this specific section (Mueller and Wydoski 2004). Bonytail, while present in Lake Havasu, are not commonly found in this part of the river but occasionally appear in backwaters

(NDOW 2020). Other species present include common carp, channel catfish, bluegill, and gizzard shad (**Table TA 8-2**) (Ozborn et al. 2021).

Anglers primarily target striped bass and rainbow trout. During the spring and early summer, striped bass migrate upstream from Lake Havasu to spawn, concentrating near the tailwater of Davis Dam, where their migration is halted (NDOW 2020). Catch rates for game fish in the surveyed section of the Colorado River typically increase with rising temperatures in the spring (Ozborn et al. 2021). Species diversity is higher in the backwaters, which tend to harbor large bass spp., and redear sunfish; while large striped bass are occasionally caught in the main channel of Laughlin Lagoon, but most are found in the river's main channel (Ozborn et al. 2021).

In 2021, largemouth bass in both the backwater and riverine areas were in very good condition; in contrast, striped bass appeared in poor to fair condition (Ozborn et al. 2021). Although anecdotal due to the small sample sizes, a representative sample would require at least 250 individuals per species. Sampling methods and management activities from downstream of Davis Dam to Parker Dam include trammel-netting, electrofishing, angler use, and harvest surveys (NDOW 2020).

The fish community of Topock Marsh includes self-sustaining populations of largemouth bass, channel catfish, black crappie, striped bass, bluegill, green sunfish, and redear sunfish. Gizzard shad were first caught in 2012 and are now the most abundant species sampled (**Table TA 8-2**). It should be noted that razorback suckers have been stocked five times since 2010 into Topock Marsh and have been caught every year since the first stocking, except 2015 (AZGFD 2022).

A population of desert pupfish (*Cyprinodon macularius*) occurs at Bill Williams National Wildlife Refuge, in a constructed and maintained pond which was established in 2006. In all cases of pupfish presence in the Lower Basin, ponds were established with the intent to provide outreach and educational opportunities, as well as to provide replicates for the El Doctor/Santa Clara Slough lineage of desert pupfish and a source for stockings and augmentations of additional sites. In the construct of the 1993 pupfish recovery plan (FWS 1993) and the 2019 amendment to the recovery plan (FWS 2019a), these highly managed artificial ponds are considered to be tier 3 replicates that contribute to the required number of replicates (by pupfish lineage) necessary for desert pupfish to be downlisted or delisted. As such, these ponds are an important conservation tool that contributes to recovery (FWS 2012).

Flathead catfish are of particular concern as they are voracious predators and pose a substantial threat to native fishes of the Lower Colorado River. This species appears to be expanding its use of habitats within Reach 3 (potentially due to temperature) and is routinely monitored by U.S. Fish and Wildlife Service (FWS) personnel. Through 2022, this species had been captured nearly as far upstream as the area between Needles and Fort Mohave in the Lower Colorado River (near Boundary Cone Road crossing) (Rasset and Love-Chezem 2024).

Since quagga mussels were discovered in Lake Mead in Nevada in 2007, subsequent surveys have found smaller numbers of quagga mussels in Lake Havasu and in the Colorado River Aqueduct System, which serves Southern California. Presently, all reservoirs, lakes, and watersheds receiving raw Colorado River water have been exposed to quagga mussels at this point (Renner and Day 2022; CDFW 2024). Crayfish (various species) are invasive freshwater crustaceans originally introduced as fishing bait and biological control for aquatic vegetation (Renner and Day 2022; AZGFD 2024b) and assumed to be present within Reach 3. The Asian clam was first detected in 1948 in Lake Mead and threatens downstream ecosystems (Renner and Day 2022); thus, present within Reach 3. As with upstream reaches, the New Zealand mudsnail and Malaysian trumpet snail continue to be a threat (Renner and Day 2022; USGS 2025).

LCR MSCP Reaches 4 and 5- Parker Dam to Imperial Dam

The Colorado River within the LCR MSCP Reaches 4–5, extending from Parker Dam (RM 192.3) to Imperial Dam (RM 49.2), represents the lowest combined reach regularly sampled by the LCR MSCP. This section historically was characterized as a wide, turbid river abundant in sediment, featuring large woody debris and devoid of large boulders (Mueller and Marsh 2002; Mueller et al. 2005). It operated as a dynamic ecosystem influenced by extreme water temperature fluctuations (0–35°C), as well as fluctuations between flooding and drought that shaped the life histories of native Colorado River fishes (Reclamation 2004; Mueller 2006).

Today, this reach of the river extends roughly 100 miles south to Yuma, AZ, and is heavily utilized for recreation, particularly boating and other activities (Mueller and Marsh 2002). It has been notably altered, becoming a channelized waterway often spanning up to 10 miles across, predominantly used for agricultural purposes (Mueller and Marsh 2002). The original riparian landscape, characterized by large willows and cottonwood trees, has been replaced by tamarisk due to altered flood management practices, leading to salt crusts on the ground due to inadequate drainage (Mueller and Marsh 2002).

The introduction of nonnative fish species has substantially displaced native fauna throughout much of the area (Mueller 2006). However, some native fish populations persisted in isolated habitats such as the Cibola High Levee Pond, a 5-acre oxbow pond semi-connected to the mainstem river. This pond provided a refuge for native fish species, allowing them to survive despite the challenges posed by nonnative species (Mueller and Marsh 2002; Mueller 2006). Additionally, Imperial Ponds Conservation Area, an 80-acre pond complex created by the LCR MSCP, also provides predator-free habitats for razorback sucker and bonytail (Kesner et al. 2012). Similar to Cibola High Levee Pond, these ponds have seen successful spawning and recruitment of native fishes; however, nonnative fish invasions are still a threat (Kesner et al. 2012). Such environments also serve as crucial research sites for biologists studying the dynamics of native fish populations in altered river ecosystems (Mueller 2006).

Currently, this reach contains ESA-listed fishes, including the razorback sucker and bonytail (**Table TA 8-2**); each primarily sustained by stocking efforts. Although stocked native fish have been found to persist in this reach, research and monitoring have shown that post-stocking survival rates are low. Therefore, native fish augmentation efforts will continue to focus on improving post-stocking survival (LCR MSCP 2024). From 2007–2023, a total of 152,515 razorback suckers and 65,303 bonytail were stocked in reaches 4–5 (LCR MSCP 2024). Many of the bonytail stocked appeared to disperse downstream, while razorback sucker seemed to disperse upstream of the stocking location; both species often utilize backwater habitats; however, razorback sucker utilized riverine habitats as well (Kelley et al. 2022). Bonytail post-stocking survival has been low and often immeasurable (Heishman et al. 2023), and bonytail mortality is likely high immediately after stocking (Kelley et al 2022). On the other hand, razorback sucker stockings have seemed more successful with population growth and relatively high post-stocking survival (0.44–0.85) in this location (Heishman et al. 2023). The population estimate was 2,208 in 2023 (95 percent CI: 2,062–2,355), the largest population estimate in this reach since the stocking program's inception in 2005 (LCR MSCP 2024). Native fish management in these reaches has primarily focused on off-channel, predator-free habitats and stocking efforts.

The LCR MSCP currently collects larval razorback sucker from Lake Mohave to rear at various hatcheries. These efforts, as well as larvae reared at Southwest Native Aquatic Resources and Recovery Center, provide razorback sucker of sufficient size and quantity to stock this species into Reach 4–5. Similarly, bonytail, from hatchery sources supported by the LCR MSCP, are also stocked into Reach 4–5. Backwater habitats created in this reach by the LCR MSCP will continue to be disconnected from the mainstem river to address the life history requirements of bonytail and razorback suckers. There are several projects, such as those at Cibola High Levee Pond and the Imperial Ponds Conservation Area, dedicated to habitat creation, backwater disconnection, manipulation, and conservation efforts, for and on behalf of the LCR MSCP fish species in this reach (LCR MSCP 2024).

The AZGFD Region IV Aquatics Program is charged with the management of the area that corresponds with the LCR MSCP Reach 4–5. The nonnative fishery primarily consists of striped bass, channel catfish, smallmouth bass, largemouth bass, sunfishes (*Lepomis* spp.), common carp, red shiner, fathead minnow, crappie (*Pomoxis* spp.), shad (*Dorosoma* spp.), goldfish, bullhead (*Ameiurus* spp.), and blue tilapia (*Oreochromis aureus*) (Renner and Day 2022; Ham 2023a, 2023b; Ramey and Ham 2024). Flathead catfish have been found in the Lower Basin since at least the 1960s, and the most recent surveys reported finding flathead catfish, a particular concern to the native fishes due to their predatory nature (**Table TA 8-2**) (FWS 2019b; Rasset and Love-Chezem 2024).

A population of desert pupfish occurs at Cibola (Reach 4) and Imperial National Wildlife Refuges (Reach 5) in constructed and maintained ponds that were established in 1999 and 2000, respectively. Currently, all fish from the pond at the Imperial refuge have been moved to Cibola National Wildlife Refuge (D. Williams, personal communication, 2025). In all cases of pupfish presence in the Lower Basin, these ponds were established with the intent to provide outreach and educational opportunities, as well as to provide replicates for the El Doctor/Santa Clara Slough lineage of desert pupfish and a source for stockings and augmentations of additional sites. In the construct of the 1993 pupfish recovery plan (FWS 1993) and the 2019 amendment to the recovery plan (FWS 2019a), these highly managed artificial ponds are considered to be tier 3 replicates that contribute to the required number of replicates (by pupfish lineage) necessary for desert pupfish to be downlisted or delisted. As such, these ponds are an important conservation tool that contributes to recovery (FWS 2012).

Crayfish are invasive freshwater crustaceans originally introduced as fishing bait and biological control for aquatic vegetation (Renner and Day 2022; AZFG 2024b). Mueller et al. (2005) observed red swamp crayfish (*Procambarus clarkii*)predating on bonytail and razorback sucker eggs in Cibola High Levee Ponds. Additionally, quagga mussels and Malaysian trumpet snail, which are known from locations within and downstream of Lake Mead, are also found in Reaches 4–5 (Renner and Day 2022; CDFW 2024; USGS 2025).

LCR MSCP Reaches 6 and 7– Imperial Dam to the SIB

Due to the overall low numbers of native and ESA-listed fishes and their low survival, coupled with current LCR MSCP fish sampling efforts being allocated primarily down through Reaches 4–5 (J. Stolberg, LCR MSCP, personal communication 2025; LCR MSCP 2024), we present information from Reaches 4–5 as the lowest-most combined reach routinely sampled (from Parker Dam [RM 192.3] to Imperial Dam [RM 49.2). We will not present data for Reaches 6–7 of the LCR MSCP. We note that historical literature provides at least some semblance of species present in this reach (**Table TA 8-2**).

TA 8.2 Environmental Consequences

TA 8.2.1 Methodology

Decision Making Under Deep Uncertainty Modeling

To assess future alternatives and management strategies, a modeling analysis known as Decision Making Under Deep Uncertainty (DMDU) (see **Chapter 3**, **Section 3.2.6**, Decision Making Under Deep Uncertainty for additional details) was applied to systematically evaluate potential system responses across a wide range of plausible futures. The analysis incorporated five alternative scenarios (No Action, Basic Coordination, Enhanced Coordination, Maximum Operational Flexibility, and Supply Driven Alternatives) representing various flow conditions and the continuation of the current flow management strategies from Lake Powell (Continued Current Strategies [CCS] Comparative Baseline). These modeled scenarios are designed to span a broad spectrum of uncertainty, allowing examination of the impacts on aquatic habitats and resources under several alternatives and baseline conditions. By evaluating outcomes across these alternative futures, DMDU offers a rigorous approach to quantifying changes in habitat extent for both lake and riverine environments. This method enables detailed interpretation of data, such as lake elevations and flow conditions under each scenario, to assess the impacts on spawning and habitat availability relative to historic conditions. If an alternative achieves a robustness score of 90 percent or higher, it can be considered truly robust with respect to a particular resource. When the difference in robustness between alternatives exceeds 10 percent, one alternative can be considered more robust than another. If the difference is less than 10 percent, the alternatives are considered similarly robust. Models were considered based on a multiagency cooperation of resource impacts.

DMDU figures are presented to provide comprehensive and reliable information about potential system outcomes under each alternative, regardless of future uncertainties. By intentionally disconnecting the analysis from probabilistic interpretation, these figures focus attention on key resource concerns and improve our understanding of how each alternative performs across a range of hydrologic conditions, and are presented to provide comprehensive and reliable information about potential system outcomes under each alternative, regardless of future uncertainties. By intentionally disconnecting the analysis from probabilistic interpretation, these figures focus attention on key resource concerns and improve our understanding of how each alternative performs across a range of hydrologic conditions.

Conditional Box Plots

The conditional boxplots serve as a visual tool to summarize the range of potential resource impacts for each alternative under varying hydrologic conditions, categorized by the 3-year average Lees Ferry natural flow preceding each simulated year. This averaging period balances capturing both persistent moderate hydrology and annual extremes, with five flow categories chosen to reflect historic conditions and stakeholder perspectives. The boxplots illustrate how each alternative responds to these hydrologic scenarios, especially under challenging conditions like drought, enabling direct comparison of resource impacts. By highlighting the variability and distribution of responses, the figures allow readers to infer connections between system history and outcomes. Because resource data points within each category are determined by hydrology rather than modeling outcomes, readers can compare relationships between different resources across conditional boxplots, and, where relevant, multiple variables may be displayed together to further illuminate system responses under different alternatives.

Robustness Heat Maps

Robustness heat maps evaluate how each alternative performs across a wide range of future scenarios over extended modeling periods, such as decades or the entire simulation horizon (2027–2060). Unlike conditional boxplots, which assess each year independently, heatmaps aggregate results according to resource-specific definitions of "acceptability," using thresholds and frequencies to classify scenarios as successful or not. Each alternative is assigned a robustness score, indicating the percentage of futures where performance criteria are met, with higher scores reflecting greater robustness. The heatmaps display multiple levels of performance, from the most challenging criteria at the top to less stringent criteria downstream of, and use a highlighted row to emphasize key acceptability thresholds or significant comparison points. This color-coded format distills complex modeling results into an accessible, comparative framework, enabling readers to quickly compare alternatives, understand their relative robustness, and make informed decisions.

Vulnerability Bar Plots

Vulnerability bar plots display, for each alternative, the hydrologic conditions, based on a key Lees Ferry natural flow statistic, under which threshold outcomes are classified as acceptable (blue) or unacceptable (red), such as during the worst 10-year drought. This visual division highlights the specific scenarios that lead to vulnerability, with larger blue regions indicating greater robustness. Accompanying boxplots provide context by relating these vulnerability thresholds to recent observations and a wide range of plausible future scenarios. The primary purpose of the vulnerability bar plot is to clarify the conditions under which an alternative is likely to fail and to determine whether those conditions fall within the range of what can reasonably be anticipated, thus informing decision-makers about each alternative's limits and resilience.

Lake Powell

To assess the impacts of reservoir elevations in Lake Powell on fish populations and aquatic habitats—including critical habitats for the Colorado pikeminnow and razorback sucker—we drew from multiple data sources using a combination of monitoring, reporting, and spatial analysis methods. Annual catch data for sportfish, collected from various monitoring programs, were analyzed alongside other datasets to track changes in lake habitat quality and quantity as water levels declined. Monitoring and distribution data for endangered Colorado pikeminnow and razorback sucker, including abundance and expansion into newly formed riverine inflows, are sourced from the Utah Division of Wildlife Resources and the Upper Colorado River Basin and San Juan River Basin Recovery Implementation Programs. These programs also provide information on spawning habitat locations and the abundance of razorback sucker larvae, which are monitored to detect shifts in spawning sites and potential bathymetric tipping points that could limit habitat availability. Stage-to-habitat descriptions and relationships derived from the scientific literature were used to evaluate habitat and fish species abundance. Stage-to-habitat descriptions are ways of linking stage height (e.g., the elevation of the water surface) to the amount and type of habitat available for fish. Surface area and longitudinal analyses were used to quantify changes in habitat extent for both lake and riverine environments, with data interpretation (e.g., lake elevation by alternative) using DMDU analysis (for additional details see **TA 3**). Additionally, the smallmouth bass model (Eppehimer et al. 2025; Eppehimer and Yackulic 2026) was used to determine entrainment of smallmouth bass at Glen Canyon Dam and the probability of entrainment based on Lake Powell elevation. The smallmouth bass model was developed by the U.S. Geological Survey (USGS) in collaboration with partners, including the Reclamation and AZGFD, as part of the Glen Canyon Dam Adaptive Management Program. The model was developed to evaluate how variables, such as water elevation, temperature, and velocity, affect smallmouth bass population dynamics. Entrainment refers to the fish being pulled into and carried through water management structures like dam intakes, pumps, or turbines, often resulting in injury, death, or unintended movement into new habitats. Together, these methodologies enable a comprehensive assessment of how reduced reservoir elevations impact sportfish, endangered native fish, and their critical habitats in Lake Powell and its inflows.

Glen Canyon Dam to Lake Mead

To assess the impacts of modified releases from Glen Canyon Dam down the Colorado River to Lake Mead, a multi-faceted methodology was employed. Habitat availability and species abundance for threatened and endangered native fishes, including humpback chub and razorback sucker, as well as other native species (e.g., flannelmouth sucker, bluehead sucker) and nonnative species (rainbow trout, brown trout, smallmouth bass) were evaluated using stage-to-habitat descriptions and relationships derived from the scientific literature. These descriptions are used to describe how changes in flow or water level (stage) create or reduce habitats, including backwaters, side channels, riffles, or shorelines that different species need for spawning, rearing, or feeding. Monitoring data and reports from agencies such as the Grand Canyon Monitoring and Research Center, the FWS, AZGFD, and the NPS provided empirical evidence on changes in habitat and fish abundance by life stage, particularly in the Lees Ferry sub-reach. Predictive modeling tools included the smallmouth bass model, applied to predict escapement from Lake Powell, entrainment through Glen Canyon Dam, and subsequent survival downstream Eppehimer et al. 2025; Eppehimer and Yackulic 2026). Additionally, the conservation measures for humpback chub outlined in LTEMP Biological Opinion, including the specific triggers that require management action, were incorporated to help guide management decisions and evaluate whether mitigation efforts are effective. Together, these methods and the DMDU approach described in **Chapter 3**, **Section 3.2.6**, Decision Making under Deep Uncertainty, enabled a comprehensive assessment of how flow modifications impact fish communities and habitats from Glen Canyon Dam to Lake Mead.

Lake Mead to SIB

To assess the impacts of reservoir elevations from Lake Mead downstream to the SIB on fish, aquatic species, and their habitats, a comprehensive methodology was employed that integrated agency monitoring, habitat analysis, and species-specific evaluations. Annual and seasonal reports, along with monitoring data from agencies, including Reclamation, the LCR MSCP, AZGFD, NDOW, and Grand Canyon Monitoring and Research Center, were used to track changes in habitat quantity and quality for lake, wetland, and riverine environments. These data sources provided information on the distribution and abundance of sportfish, ESA-listed, and other native species (humpback chub, razorback sucker, bonytail, flannelmouth sucker, and desert pupfish) across newly expanded inflow zones and affected sub-reaches. The methodology included the analysis of Colorado River Simulation System (CRSS) data to identify changes in Lake Mead shoreline area (spawning and nursery habitats) and longitudinal habitat assessments to quantify habitat expansion or contraction throughout the Lower Colorado River system. CRSS is a long-term planning model, developed and maintained by Reclamation, and is used to simulate how water moves through the Colorado River system under different inflow, demand, and management scenarios. Particular focus was placed on known spawning sites and critical habitats, as well as how changes in water quantity, release timing, temperature, and quality from Hoover Dam affect downstream riverine habitats.

The alternative analysis approach for Lower Basin fishes was based on the observation that native and nonnative fish populations have persisted through the interim guideline operational period (2008 to present), with ongoing multi-year drought, declining reservoir storage, and revised Colorado River management strategies (Reclamation 2007). Flow and water quality data from this period were thoroughly documented and analyzed (see **TA 3** and **TA 6**). The guiding principle for fisheries analysis assumed that populations capable of surviving under these historic conditions would likely persist under similar future scenarios. Using the DMDU approach, the analysis evaluated whether the Draft EIS alternatives could produce future conditions, such as flows, lake elevations, and habitat availability, outside the historical range observed from 2008 to 2025. All relevant data were assessed using established DMDU procedures and CRSS modeling (see **TA 3** for details). This integrated, historically informed approach enabled a robust evaluation of how changing reservoir conditions could affect fish populations and aquatic habitats across the analysis area.

Impact Analysis Area

The impact analysis area includes the Colorado River corridor from the full-pool elevation of Lake Powell to the SIB as described in **Sections 3.2** and **TA 10.1**. The issues analyzed under this section are specific to defined boundaries within the larger study area, as delineated in **Table TA 8-2**.

Assumptions

- There will be modifications in quantity, timing, temperature, and quality of water released from Glen Canyon Dam and Hoover Dam through the Lower Colorado River. Fish habitats, distributions, and abundances are likely influenced by changes in flows, with reduced flows and variations in dam releases potentially altering available habitat and population numbers.
- Backwaters and riverine native fish habitats are likely affected by reduced flows.

- Known spawning sites of razorback sucker may be affected by reduced flows. Analyses involving models (CRSS and smallmouth bass) will be conducted in collaboration with model developers and users from various cooperating agencies.
- LCR MSCP reaches may be combined for analysis due to the lack of special status species, habitat, operations, and management.
- Ongoing fisheries management practices conducted by the various agencies in the Colorado River Basin would be similar and ongoing into the future.

Impact Indicators

- Abundance and location of razorback sucker larvae throughout the project area.
- Elevation where spawning habitat may become limited (if available) throughout the project area.
- Changes in annual monitoring of catches of sportfish throughout the project area.
- Changes in available habitat and abundances of native fish species by life stage throughout the project area.
- Changes in backwater areas at flow stages in the Lower Colorado River Basin (Lower Basin).
- Changes in distribution and abundance of Colorado pikeminnow and razorback sucker with expanded inflows in Lake Powell.
- Changes in distribution and abundance of humpback chub with expanded inflow habitats in Grand Canyon.
- Changes in electrofishing monitoring catch rates and angler catch rates of rainbow trout and brown trout in Lees Ferry sub-reach.
- Changes in fish habitat and/or fish distribution and numbers based on dam releases and flows throughout the project area.
- Changes in razorback sucker spawning site areas and suitability at flow stages in the Lower Basin.
- Reduction in Colorado pikeminnow and razorback sucker critical habitat in the Colorado River and San Juan River inflows to Lake Powell with increased elevation of Lake Powell.
- Changes to razorback sucker critical habitat, which is the full-pool of Lake Mead.
- Increased numbers of smallmouth bass escaping through Glen Canyon Dam and increased distribution and numbers of smallmouth bass downstream.
- Changes in salinity concentration in the Lower Basin.
- Lower reservoir elevation changes to shoreline spawning and nursery habitat for razorback sucker throughout the project area.
- Lake Powell elevation below 3,666.5 feet to sustain Piute Farms waterfall as a nonnative fish barrier.
- Lake Mead elevation below 1,090 feet to sustain Pearce Ferry rapid as a nonnative fish barrier, but above 1,000 feet to protect the power pool.
- Reduced abundance and distribution of desert pupfish in the Lower Basin.
- Reduced abundance and distribution of tilapia and other fish species in the Lower Basin.

- Reduced water quality from nitrates and selenium, resulting in algal blooms and fish die-offs in the Lower Basin.
- Reduced wetland area and river inflow habitat in the Lower Basin.

TA 8.2.2 Issue 1: Lower Lake Powell reservoir elevations have reduced lake habitat and extended the inflows of the Colorado River and San Juan River by approximately 35 miles each. How will changes in flow impact the quantity and quality of lake, wetland, and riverine inflow habitats?

Recent declines in reservoir elevations have dramatically reshaped aquatic habitats in the Lake Powell region, reducing the extent of traditional lake environments while simultaneously lengthening the riverine inflow zones of both the Colorado and San Juan Rivers. Lower reservoir elevations in Lake Powell have expanded riverine inflow habitats that benefit ESA-listed species like Colorado pikeminnow and razorback sucker by exposing greater lengths of riverine critical habitat. These changes raise important questions about how altered flow regimes will affect the quantity and quality of habitats available for lake, wetland, and riverine species. Understanding the ecological consequences of these shifting boundaries is critical for safeguarding endangered native species such as the Colorado pikeminnow and razorback sucker, managing sportfish populations, and maintaining the overall health and resilience of these interconnected aquatic systems.

Of the Impact Indicators listed above for Lake Powell, those of greatest importance as threats to native fish include, in order of importance: (1) Increased numbers of smallmouth bass escaping through Glen Canyon Dam and increased distribution and numbers of smallmouth bass downstream, (2) Lake Powell elevation below 3,666.5 feet to sustain Piute Farms waterfall as a nonnative fish barrier, (3) Reduction in Colorado pikeminnow and razorback sucker critical habitat in the Colorado River and San Juan River inflows to Lake Powell with increased elevation of Lake Powell.

Lake elevations also influence upstream fish passage at the Piute Farms Waterfall on the San Juan River, which serves as a barrier for upstream movement of nonnative fishes but also prevents native fishes from moving upstream into historic riverine habitats. Blocking upstream movement of nonnative fish from Lake Powell into the San Juan River benefits the native fish by reducing predators and competitors in the river. However, the waterfall blocks upstream movement of threatened razorback sucker and Colorado pikeminnow that move or are transported downstream of the waterfall, preventing them from returning to their historic and critical habitat in the river. Lowered lake elevations also increase the risk of smallmouth bass entrainment at Glen Canyon Dam, especially when the elevation is more closely aligned with the penstocks. Lower reservoir elevation also results in a reduction of lake habitat for sportfish, which may negatively impact recreational sportfishing, but could increase predation and competition on native fish. Impact indicators for these changes include variations in the distribution and abundance of Colorado pikeminnow and razorback sucker within expanded inflow areas, alterations in upstream passage at the Piute Farms Waterfall, rates of smallmouth bass entrainment at Glen Canyon Dam, and changes in annual catches of sportfish. Several thresholds are considered, including the lake elevation above 3,490 feet required to maintain Glen Canyon Dam releases and power production, the lake elevation at which critical habitat for Colorado pikeminnow and razorback sucker begins to inundate on the

Colorado River inflow (3,598 feet) and the San Juan River inflow (3,600 feet), the elevation at which Piute Farms Waterfall is inundated (3,666.5; Paul Grams, USGS/GCMRC, personal communication), permitting fish passage, and the elevation enabling smallmouth bass entrainment at Glen Canyon Dam (3570 feet; see smallmouth bass model (Eppehimer et al. 2025; Eppehimer and Yackulic 2026), **Section TA 8.2.3**). These thresholds are crucial for the persistence of ESA-listed fish populations and sportfish in Lake Powell and downstream of Glen Canyon Dam. A relationship was developed linking riverine length of the Colorado River and San Juan River inflows into Lake Powell with lake elevation (**Table TA 8-3**, **Table TA 8-4**). This analysis focuses on the designated critical habitat for Colorado pikeminnow and razorback sucker. Critical habitat for humpback chub and bonytail does not occur in these inflow areas. Critical habitat for Colorado pikeminnow and razorback sucker extends 33.5 miles of the Colorado River inflow from Rapid 25 (RM 201.5, full-pool elevation of Lake Powell at 3,700 feet) downstream to North Wash (RM 168, elevation of 3,598 feet). When Lake Powell rises above 3,598 feet, it begins to inundate designated critical habitat for Colorado pikeminnow and razorback sucker, reducing habitat quality. A similar relationship was developed for critical habitat on the San Juan River inflow that extends 39 miles from upstream of the Clay Hills boat ramp (RM 63.5, full-pool elevation) downstream to Neskahai Canyon (RM 24.5) (**Table TA 8-4**). These species, especially the Colorado pikeminnow, prefer riverine habitats when available (FWS 2018, 2022), and lengthening the riverine inflow exposes a greater length of critical habitat and benefits these species.

Table TA 8-3
Estimated Linear Miles of Colorado Pikeminnow and Razorback Sucker Critical Habitat at Varying Lake Powell Elevations (in feet) in the Colorado River Inflow

Colorado River Inflow	River Mile	Elevation	Critical Habitat Inundated (River Miles)
Rapid 25	201.5	3,700	33.5
Imperial Canyon	200	3,690	32
Gypsum Canyon	196.6	3,680	28.6
Clearwater Canyon	192	3,660	24
Bowdie Canyon	190.6	3,658	22.6
Cove Canyon	186	3,640	18
Dark Canyon	183	3,625	15
Sheep Canyon	177	3,610	9
Narrow Canyon	173	3,604	5
Dirty Devil River	169.5	3,602	1.5
North Wash	168	3,598	0

Note: River miles of critical habitat are computed from the relationship y= 0.3178x - 972.22 (R2 = 0.97), where x = lake elevation, y = river mile. River miles are measured upstream of Lees Ferry.

Table TA 8-4
Estimated Linear Miles of Colorado Pikeminnow and Razorback Sucker Critical Habitat at Varying Lake Powell Elevations (in feet) in the San Juan River Inflow

San Juan River Inflow	River Mile	Elevation	Critical Habitat Inundated (River Miles)
5.5 miles upstream of Clay Hills Boat Ramp	63.5	3,700	39
Clay Hills Boat Ramp	58	3,690	33.5
Piute Falls Waterfall-top	54.5	3,666	30
Piute Falls Waterfall-bottom	53.5	3,638	29
Mikes Canyon	50.5	3,632	26
Copper Canyon	45	3,627	20.5
Nokai Creek	42	3,626	17.5
Great Bend	29.5	3,605	5
Neskahai Wash	24.5	3,600	0

River miles of critical habitat are computed from the relationship y=0.3363x - 1177.8.22 (R2=0.81), where x = lake elevation, y = river mile. River miles are measured upstream from the historic confluence of the San Juan and Colorado rivers.

In the Average Flow Category (12-14 million acre-feet [maf]) for water year maximums, 90 percent, and 75 percent interquartile ranges for all alternatives are projected to remain above 3,598 feet, which would inundate varying lengths of critical habitat in the Colorado River inflow (see **Table TA 3-6**, Water Year Minimum and End of Water Year (EOWY) Elevations and Storage Volumes of Lake Powell in **TA 3**, Hydrologic Resources). At water year minimums and interquartile ranges of 25 percent and 10 percent, the elevations are projected to remain below 3,598 feet. Projections for 50 percent are variable, above and below 3,598 feet. This analysis shows that most of the time, critical habitat in the Colorado River inflow is expected to be partly inundated. The results are virtually the same for critical habitat in the San Juan River inflow, which is only 2 feet higher at 3,600 feet.

Figure TA 8-1 shows the percent of modeled futures for each alternative in which Lake Powell elevation is below 3,598 feet in the percent of months specified by each row. The highlighted row indicates the preferred minimum performance level, which is for Lake Powell elevation to stay below 3,598 feet in at least 60 percent of months in each modeled future. 60 percent of months is a historical benchmark, as this is approximately the percent of months Lake Powell elevation has been below 3,598 feet during the last 10 years (Lake Powell elevation was below 3,598 feet in 62.5% of months from October 2014 to May 2025). Alternatives that satisfy this minimum preferred performance level in a high percentage of futures are better at protecting critical habitat in the Colorado River inflow for Colorado pikeminnow and razorback sucker, whereas alternatives that satisfy the preferred minimum preferred performance in a low percentage of futures are worse.

This analysis shows that the Enhanced Coordination and Maximum Operational Flexibility Alternatives are the least performing alternatives, satisfying the preferred minimum performance level in only 17 and 20 percent of modeled futures, respectively. The Supply Driven Alternative is the best, meeting the preferred minimum performance level in 58 percent of futures. The Supply

Driven Alternative is the only alternative better than the CCS Comparative Baseline, which meets the preferred minimum performance level in 52 percent of futures.

Figure TA 8-1
Inundation of Critical Habitat of Colorado Pikeminnow and Razorback Sucker by Colorado River Inflow: Robustness.
Percent of futures in which the Lake Powell elevation is below 3,598 feet in the percent of months specified by each row

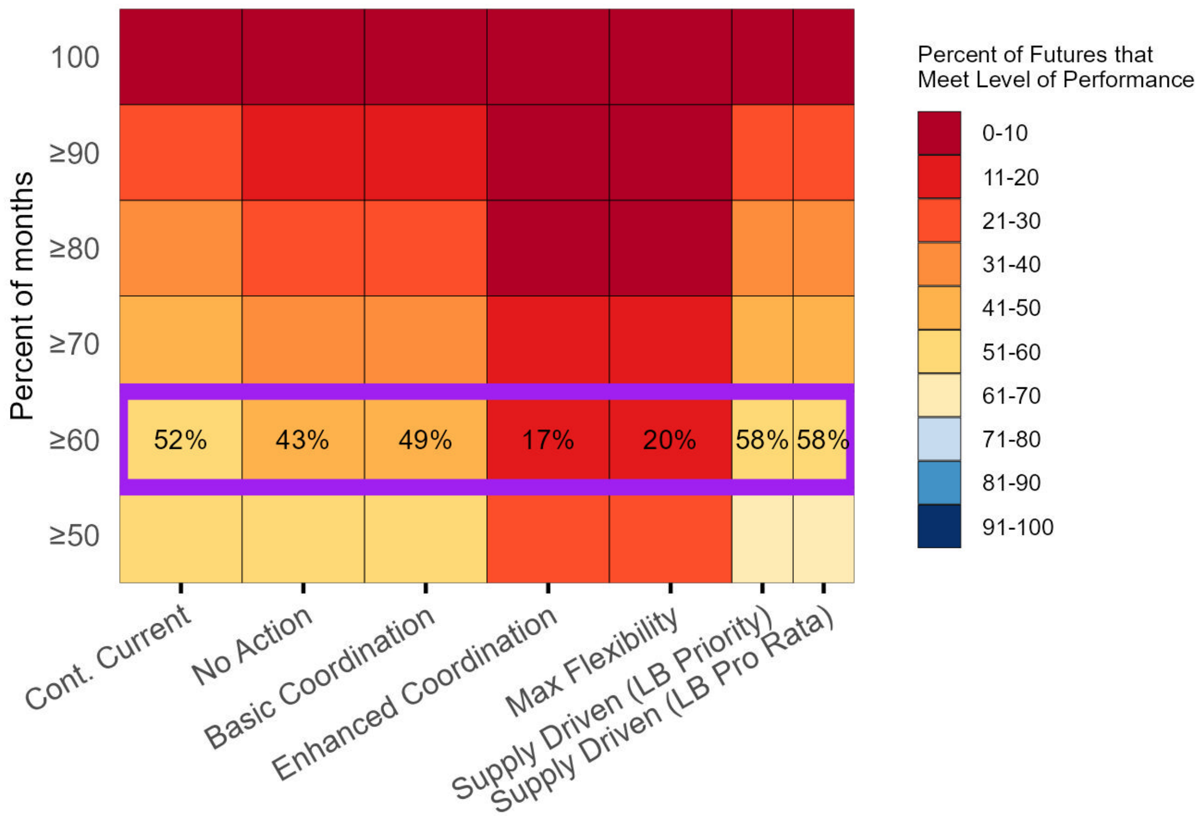


Figure TA 8-2 shows the streamflow conditions likely to cause each alternative to fail the preferred minimum performance level, which means Lake Powell elevation above 3,598 feet in 40 percent or more months. The median 20-year average streamflow in each future, plotted on the vertical axis, was the most skillful streamflow statistic at identifying if a future will satisfy or fail the preferred minimum performance level. The Enhanced Coordination and Maximum Operational Flexibility Alternatives are expected to fail the minimum preferred performance level if the median 20-year average flow is greater than 11.6 and 11.7 maf per year, respectively. Unless future hydrology is significantly drier than the most recent historical 20-year average (13.1 maf, shown by the dotted line in the reference hydrology panel), it can be reasonably expected that these two alternatives will result in Lake Powell elevations above 3,598 feet more than 40 percent of months. The remaining alternatives are vulnerable to streamflow conditions similar to recent history (ranging from 12.8 to 13.3 maf per year). If future hydrology is similar or drier, these alternatives are more likely to maintain critical habitat for Colorado pikeminnow and razorback sucker. Inundated critical habitat reduces the value of habitat for these species and could reduce the numbers of fish in the inflow, as well as survival and reproductive potential.

Figure TA 8-2
Inundation of Critical Habitat of Colorado Pikeminnow and Razorback Sucker by Colorado River Inflow: Vulnerability.
Conditions that could cause Lake Powell to go above 3,598 feet in more than 40% of months

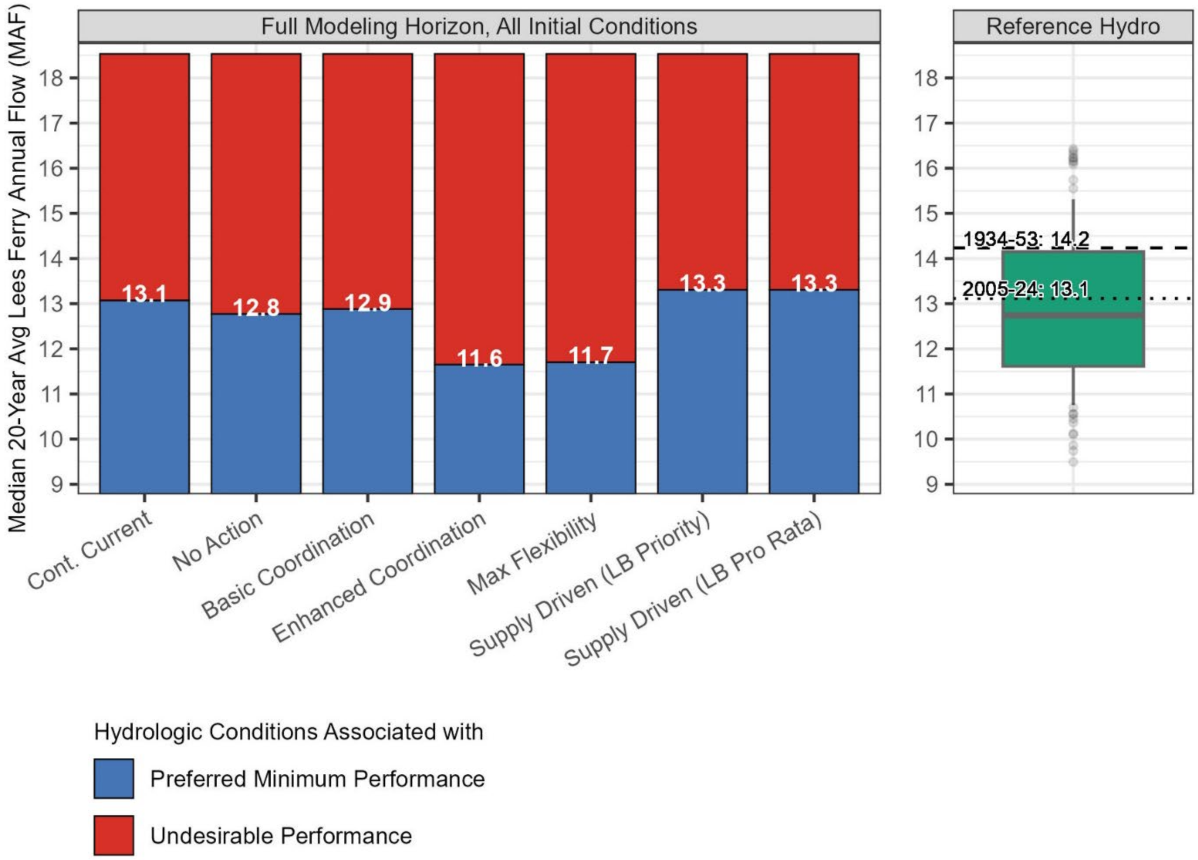


Figure TA 8-3 shows the percent of modeled futures for each alternative in which Lake Powell elevation is below 3,600 feet in the percent of months specified by each row. The highlighted row indicates the preferred minimum performance level, which is for Lake Powell elevation to stay below 3,600 feet in at least 60 percent of months in each modeled future. 60 percent of months is a historical benchmark, as this is approximately the percent of months Lake Powell elevation has been below 3,600 feet during the last 10 years (Lake Powell elevation was below 3,600 feet in 64.1% of months from October 2014 to May 2025). Alternatives that satisfy this minimum preferred performance level in a high percentage of futures are better at protecting critical habitat in the Colorado River inflow for Colorado pikeminnow and razorback sucker, whereas alternatives that satisfy the preferred minimum preferred performance in a low percentage of futures are worse.

This analysis shows that the Enhanced Coordination and Maximum Operational Flexibility Alternatives are the least performing alternatives, satisfying the preferred minimum performance level in only 18 and 21 percent of modeled futures, respectively. The Supply Driven Alternative is the best, meeting the preferred minimum performance level in 59 percent of futures. The Supply Driven Alternative is the only alternative better than the CCS Comparative Baseline, which meets the preferred minimum performance level in 53 percent of futures.

Figure TA 8-3
Inundation of Critical Habitat of Colorado Pikeminnow and Razorback Sucker by San Juan River Inflow: Robustness.
Percent of futures in which the Lake Powell elevation is below 3,600 feet in the percent of months specified by each row

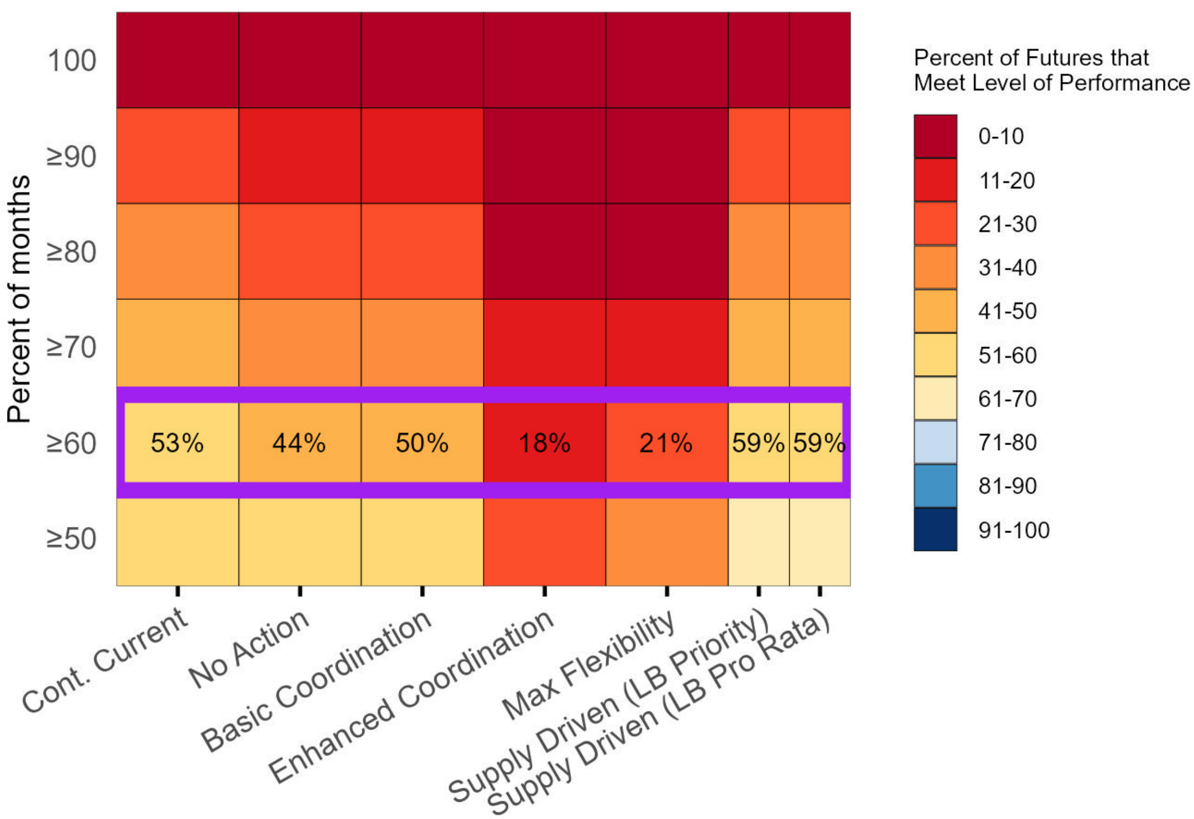


Figure TA 8-4 shows the streamflow conditions in which each alternative is likely to fail the preferred minimum performance level, meaning Lake Powell elevation above 3,598 feet in 40 percent or more months. The median 5-year average streamflow in each future, plotted on the vertical axis, was the most skillful streamflow statistic at identifying if a future will satisfy or fail the preferred minimum performance level. The Enhanced Coordination and Maximum Operational Flexibility Alternatives are expected to fail the minimum preferred performance level if the median 5-year average flow is greater than 11.2 and 11.7 maf per year, respectively. Unless future hydrology is similar to or drier than the most recent historical 5-year average (11.2 maf, shown by the dotted line in the reference hydrology panel), it can be reasonably expected that these two alternatives will result in Lake Powell elevations above 3,600 feet more than 40 percent of months. For the action alternatives and the CCS Comparative Baseline, over 50 percent of traces in the reference hydrology have a median 5-year average associated with satisfying the preferred minimum performance level.

Figure TA 8-4
Inundation of Critical Habitat of Colorado Pikeminnow and Razorback Sucker by San Juan River Inflow: Vulnerability.
Conditions that could cause Lake Powell to go above 3,600 feet in more than 40% of months

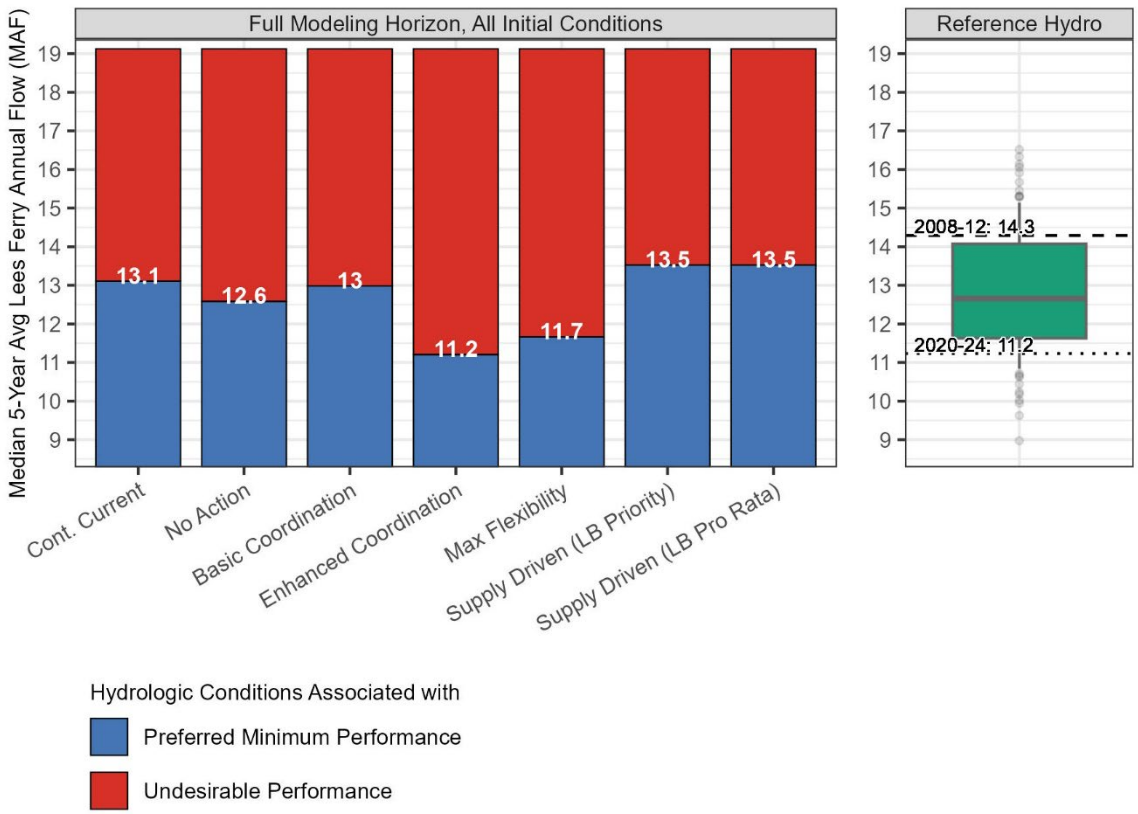
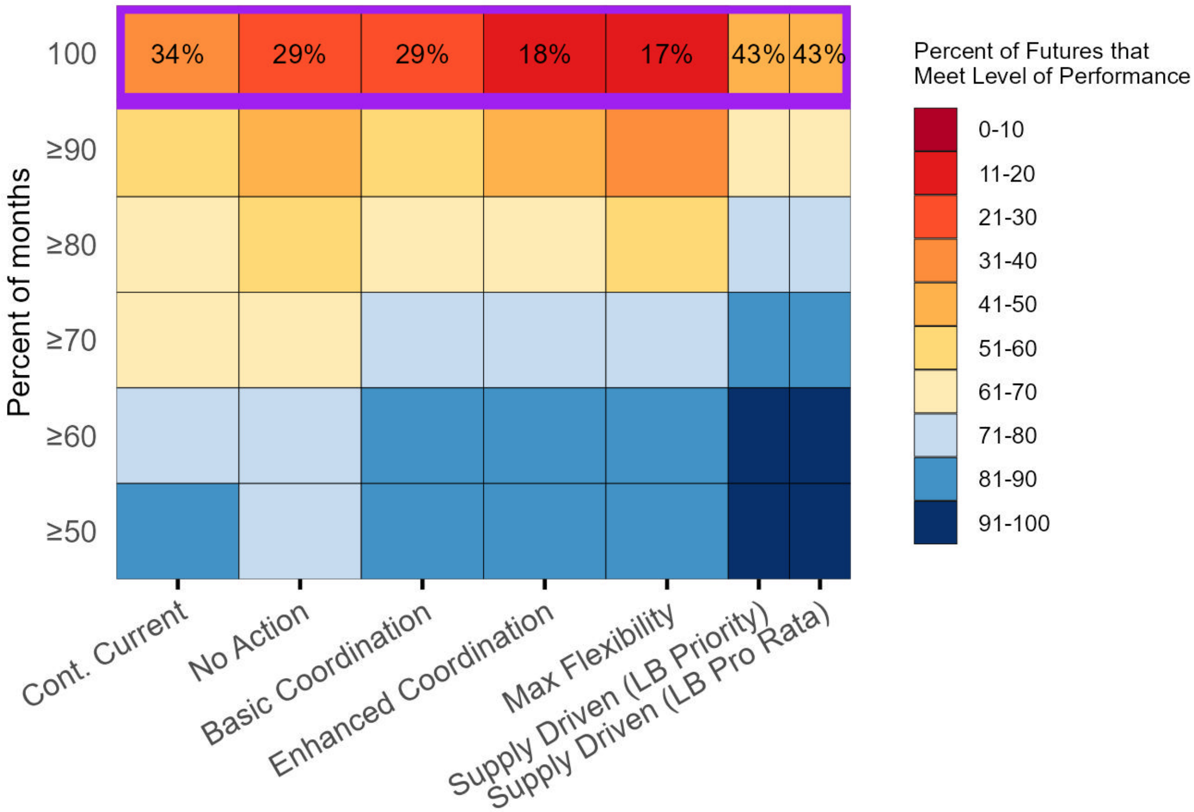


Figure TA 8-5 shows the percentage of modeled futures for each alternative in which Lake Powell elevation is below 3,666.5 feet in the percent of months specified by each row. Maintaining an elevation below 3,666.5 feet in 100 percent of months (the preferred minimum performance level indicated by the highlighted row), ensures that Paiute Farms Waterfall continues to serve as a barrier preventing both native and nonnative fish from moving upstream from the reservoir into the historical riverine habitat of the San Juan River. The Supply Driven alternative is the best performing alternative, but only satisfies the preferred minimum performance in 43 percent of modeled futures. Although this is better than the CCS Comparative Baseline (34 percent of futures), the results indicate that maintaining an elevation below 3,66.5 feet in every month over the next 34-years may be difficult to achieve. If the performance level is relaxed to 90% or more of months, the robustness scores improve significantly. For example, the Supply Driven Alternative and CCS Comparative Baseline meet this performance level in over 61 percent and 41 percent of futures, respectively.

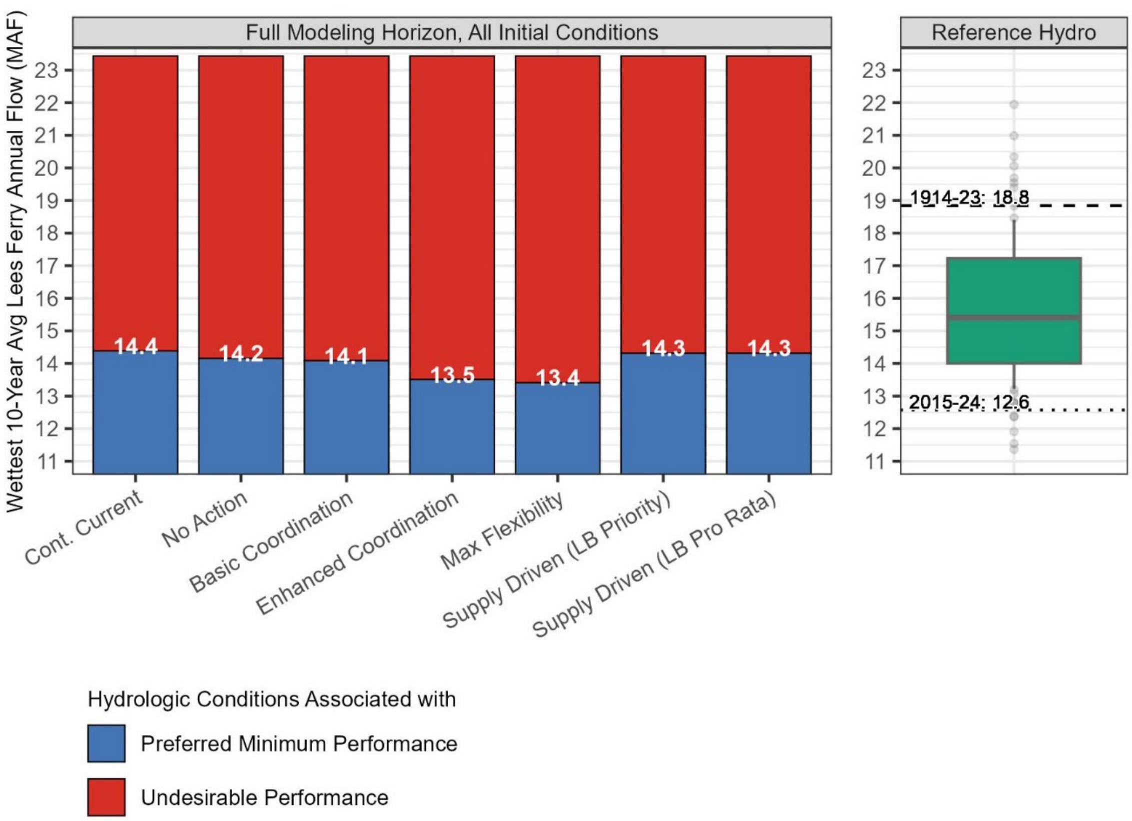
Figure TA 8-5
Paiute Farms Waterfall remains a barrier to upstream fish passage: Robustness. Percent of futures in which the Lake Powell elevation is below 3,666.5 feet in the percent of months specified by each row



When the elevation of Lake Powell is above 3,666.5 feet, the Paiute Farms Waterfall on the San Juan River becomes inundated, allowing fish to move above the waterfall from Lake Powell into the San Juan River. This would allow the native fish to access historical riverine habitat that could benefit their populations. But these same conditions would also allow nonnative fish to access the river as potential predators and competitors of native fish.

Figure TA 8-6 shows the streamflow conditions likely to cause each alternative to fail the preferred minimum performance level, which means Lake Powell elevation above 3,666.5 feet in one or more months. The wettest 10-year average streamflow in each future, plotted on the vertical axis, was the most skillful streamflow statistic at identifying if a future will satisfy or fail the preferred minimum performance level. The Maximum Operational Flexibility and Enhanced Coordination Alternatives are the most vulnerable, being expected to fail the minimum preferred performance level in futures that experience a 10-year period averaging 13.4 and 13.5 maf or more, respectively. The remaining alternatives and the CCS Comparative Baseline are vulnerable to similar conditions, ranging from 14.1 to 14.4 maf. All alternatives would be expected to satisfy the preferred minimum performance level if the wettest 10-year period in the future is similar to or drier than the most recent 10-year average of 12.6 maf (shown by the dotted line in the reference hydrology panel). Due to the highly variable nature of streamflow in the Colorado River, however, 10-year periods wetter than 2015-2024 could occur. For example, greater than 90% of futures in the reference hydrology experience a 10-year period wetter than the recent historical average.

Figure TA 8-6
Paiute Farms Waterfall remains a barrier to upstream fish passage: Vulnerability. Conditions that could cause Lake Powell elevation to go above 3,666.5 feet in 1 or more Months



While maintaining the Paiute Farms Waterfall as a fish barrier could reduce the effectiveness of river population augmentation efforts (e.g., entrainment of native fish over the waterfall into Lake Powell) and may influence long-term genetic diversity of native fish, it is still considered a net benefit for native fish recovery. Preventing upstream movement of predatory nonnative species, such as striped bass, largemouth bass, and walleye, is critical to protecting recovery efforts. To mitigate impacts on native species, the San Juan River Basin Recovery Implementation Program currently funds manual translocation of native fish from below the falls to above them and is working to secure funding for a selective fish passage structure that would support native fish movement at this site in the future.

Table TA 8-5 shows the area of inundated shoreline habitat (in acres) at different elevations of Lake Powell, based on the relationship from Root and Jones (2022). The analysis quantifies how changes in lake elevation influence the availability of nearshore habitat. At higher lake elevations, this typically consists of shallow, vegetated areas. At lower elevations, rocky talus shorelines would likely predominate. Higher lake levels tend to favor popular sportfish, like largemouth bass, bluegill, and crappie, whereas lower levels favor smallmouth bass (Gustaveson 2018). Expansion of these nonnative sportfishes will likely add predation pressure to native fishes, with the smallmouth bass likely having the highest impact (Eppehimer et al. 2025).

Table TA 8-5
Estimated Acres of Shoreline Habitat at Varying Lake Powell Elevations

Elevation of Lake Powell (Feet)	Inundated Shoreline Habitat (Acres)	Available Shoreline Habitat (Acres)
3,700	160,784	0
3,680	145,647	13,900
3,660	130,899	27,110
3,640	118,054	39,880
3,620	105,929	51,700
3,600	95,387	62,920
3,580	85,667	73,240
3,560	75,981	82,580
3,540	67,206	91,690
3,520	59,476	99,640
3,500	52,386	106,600
3,490	44,883	109,790

Source: Root and Jones 2022
Note: Acres of habitat are computed from the relationship $y=0.00113145734416358*X^3+(-0.9757039*X^2) + (35570.7021*X) + (-38508306.3)$ (R^2 = 0.9998), where x=lake elevation, y=acres of inundated shoreline (relationship from Root and Jones [2022]).

In the Average Flow Category (12–14 maf) for water year maximums, 90 percent, and 75 percent interquartile ranges for all alternatives are projected to increase inundation of shoreline habitat around Lake Powell, except for the No Action Alternative (see **Table TA 3-6**, Water Year Minimum and End of Water Year (EOWY) Elevations and Storage Volumes of Lake Powell, in **TA 3**, Hydrologic Resources). The increase or decrease (-) in shoreline habitat for each alternative, compared to the CSS Comparative Baseline, is the No Action Alternative (7,484), the Basic Coordination Alternative (-9,286), the Enhanced Coordination Alternative (-2,187), the Maximum Operational Flexibility Alternative (-732), and the Supply Driven Alternative (-18,740).

Figure TA 8-7 shows the percentage of modeled futures in which October 1st Lake Powell elevation is above 3,570 feet in the percent of years specified by each row. Maintaining an elevation above 3,570 would reduce the risk of entraining smallmouth bass from the forebay of Glen Canyon Dam and escaping into the Colorado River downstream of the dam. An elevation below 3,570 feet could increase the abundance of smallmouth bass in the Lees Ferry sub-reach and possibly further downstream, and increase predation and competition on native fish, especially the threatened humpback chub. The preferred minimum performance level, shown by the highlighted row, is for October 1st Lake Powell elevation to be above 3,570 feet in at least 80 percent of years. 80 percent of years was chosen because, from 2014 to 2024, the October 1st elevation was above 3,570 feet in nine of eleven years (81.8%).

This analysis shows that the Enhanced Coordination and Maximum Operational Flexibility Alternatives are the best performing, satisfying the preferred minimum performance level in 73 and 61 percent of futures, respectively. The remaining alternatives and the CCS Comparative Baseline have similar but much worse performance, ranging from 28 to 34 percent of futures.

Figure TA 8-7
Smallmouth Bass Entrainment: Robustness.
Percent of futures for each alternative in which the October 1st Lake Powell elevation is above 3,570 feet in percent of years specified by each row

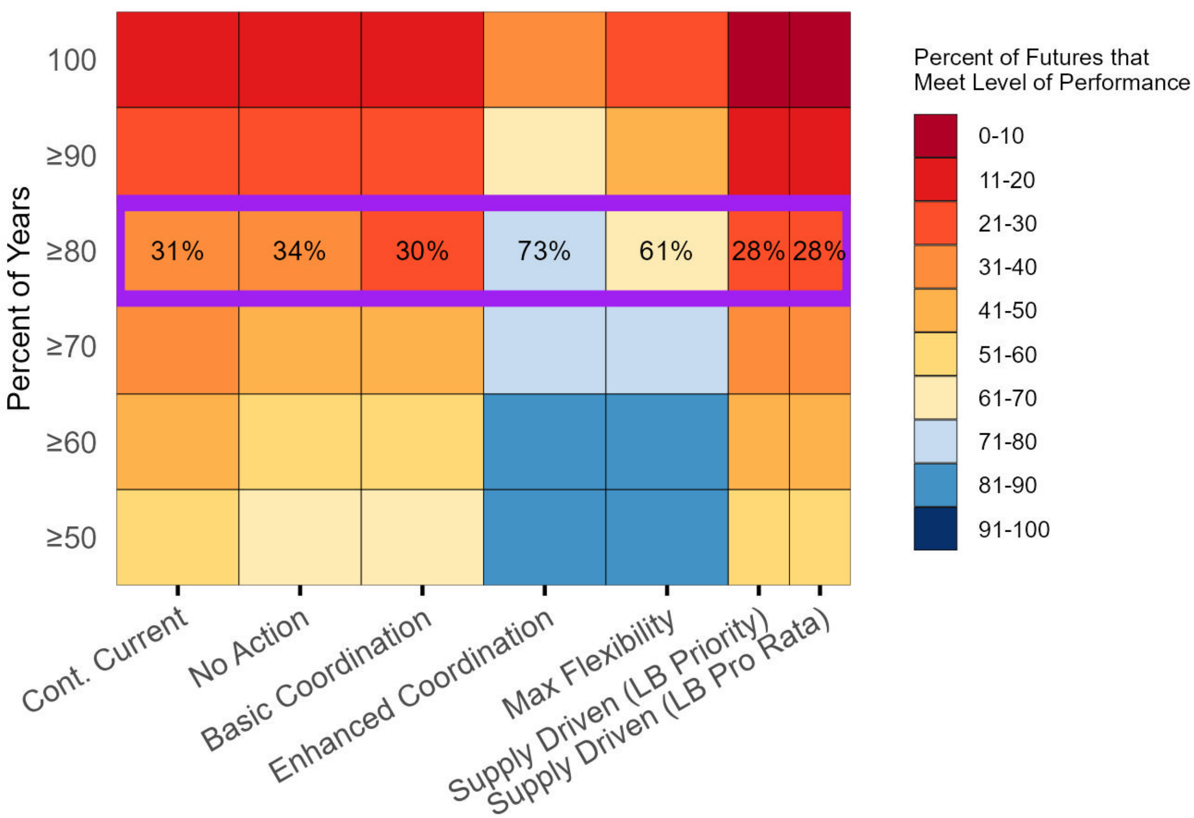
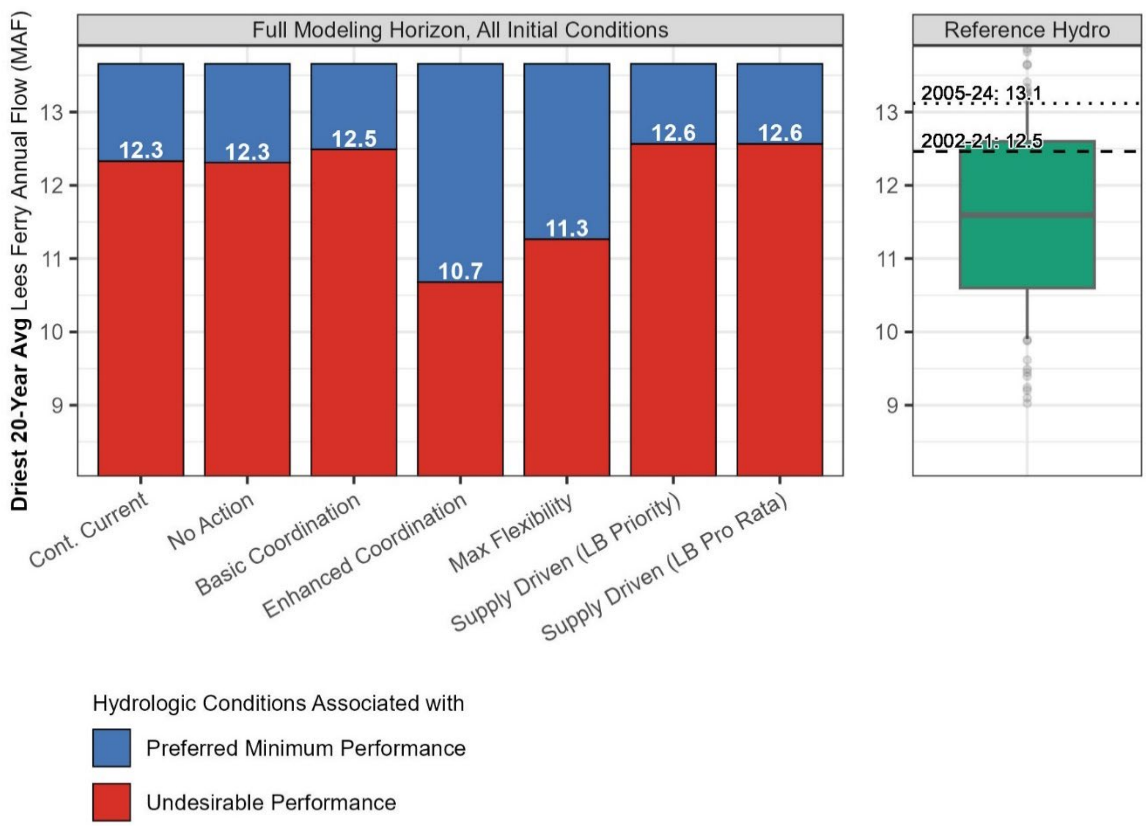


Figure TA 8-8 shows the streamflow conditions in which each alternative is likely to fail the preferred minimum performance level, meaning October 1st Lake Powell elevation below 3,570 feet more than 20 percent of years. The driest 20-year average streamflow in each future, plotted on the vertical axis, was the most skillful streamflow statistic at identifying if a future will satisfy or fail the preferred minimum performance level. The Enhanced Coordination and Maximum Operational Flexibility Alternatives are the least vulnerable, being expected to fail the preferred minimum performance level in futures that experience a 20-year average of 10.7 and 11.1 maf or drier, respectively. For comparison, these levels are significantly drier than the driest 20-year period in the historical record (12.5 maf, shown as the dashed line in the reference hydrology panel), and less than half of the traces in the reference hydrology ensemble experience conditions drier than these levels. The remaining alternatives and the CCS Comparative Baseline are vulnerable in conditions similar to the driest 20-year period observed in the historical record, so it is reasonable to expect they could result in October 1st elevations above 3,570 feet at a frequency greater than observed in recent history (i.e. greater than 20 percent of years), indicating a greater risk of smallmouth bass moving through the dam into the river. This could increase the abundance of smallmouth bass in the Lees Ferry sub-reach and possibly further downstream, and increase predation and competition on native fish, especially the threatened humpback chub.

Figure TA 8-8
Smallmouth Bass Entrainment: Vulnerability.
Conditions that could cause the October 1 Lake Powell Elevation to go Below 3,570 feet in more than 20 percent of years



TA 8.2.3 Issue 2: Changes in water quantity, release timing, temperature, and quality from Glen Canyon Dam downstream through the Grand Canyon to Pearce Ferry.

The reach of the Colorado River from Glen Canyon Dam to Lake Mead has undergone ecological changes due to modifications in water quantity, timing, temperature, and quality resulting from dam operations. These changes have affected the habitats and populations of both native and nonnative fish species throughout the Grand Canyon. Of particular concern are the threatened humpback chub and endangered razorback sucker, whose survival and abundance are closely tied to flow regimes and habitat availability. Other native species, such as flannelmouth sucker and bluehead sucker, as well as sportfish such as rainbow trout, are similarly affected by altered river conditions. As the elevation of Lake Powell decreases, the epilimnion, the warm, well-mixed surface water where most fish reside, aligns more closely with the dam's penstocks. The reduction in water elevation increases the likelihood of nonnative fish, such as smallmouth bass, being entrained, passing through the dam, and entering the Colorado River downstream. Entrainment modeling (Eppehimer et al. 2025) results suggest that propagule pressure increases dramatically when elevations are below 3,530. If reservoir elevations fall below 3,510 feet, it becomes difficult to control the invasion of smallmouth bass, and the risk persists as long as water continues to be released through the penstocks (greater than 3,490 feet, minimum power pool). Once below 3,490 feet, the risk of invasion declines as water is released through the river outlet works. Under current dam operations, the model predicts that maintaining reservoir elevations above 3,570 feet at the start of the year would substantially reduce the risk of entraining smallmouth bass and other undesirable nonnative species (Eppehimer et al. 2025). Additionally, as Lake Powell's elevations decline, warmer water from the epilimnion is released through the dam, resulting in increased water temperatures downstream. These warmer water conditions are likely to facilitate the reproduction and establishment of warmwater nonnative fish that pose a major threat to ESA-listed fish species and other native and sportfish living downstream of Glen Canyon Dam. If highly predatory smallmouth bass, which are efficient predators, were to establish downstream of the dam, removal efforts would be difficult and expensive, with potentially limited success.

To analyze these issues, a suite of biological and ecological indicators and metrics was used. Results of the hydrologic CRSS model were used to evaluate the effects of flows on aquatic resources. Preliminary quantitative models integrating information on smallmouth bass population dynamics, potential entrainment rates, water temperature, and other variables to assess invasion risk and potential management options for smallmouth bass were also used. Key indicators include changes in available habitat and species abundance by life stage, shifts in catch rates for rainbow trout and brown trout in the Lees Ferry sub-reach, and the capture of smallmouth bass downstream of Glen Canyon Dam.

A key metric for this analysis includes water temperature thresholds at Lees Ferry, the Little Colorado River confluence, the Havasu Creek confluence, and Pearce Ferry along the longitudinal extent of the river between Glen Canyon Dam and Lake Mead. For humpback chub, temperatures below 12 °C limit or inhibit growth while temperatures above 16 °C allow for mainstem spawning (Valdez and Speas 2007). For smallmouth bass, 16 °C was selected for analysis as temperatures near or exceeding 16 °C promote successful spawning (Bestgen and Hill 2016). For trout species, 20 °C was used as the thermal threshold at which trout display stress and mortality, with brown trout being more tolerant than rainbow trout at temperatures exceeding 20 °C (Valdez and Speas 2007).

The smallmouth bass model was used to predict both propagule pressure, defined as the number of smallmouth bass entrained or passing through Glen Canyon Dam, and downstream population growth rate (lambda) based on factors such as Lake Powell elevation, temperature, inflow volumes, and outflows (Yackulic et al. 2024; Eppehimer et al. 2025; Eppehimer and Yackulic 2026)). The model provides predictions for propagule pressure estimated at Glen Canyon Dam and lambda at specific river miles, including RM 15, RM 61 at the Little Colorado River confluence, RM 156 at the Havasu Creek confluence, and RM 281 at Pearce Ferry, which can help assess the potential impacts of different management alternatives on smallmouth bass populations. Lambda indicates whether the population is increasing (values greater than 1) or declining (values less than 1). It serves as an important metric for evaluating the reproduction and survival success of smallmouth bass downstream of Glen Canyon Dam. A lambda threshold of less than 1 was chosen as the analysis threshold.

The smallmouth bass model estimates the number of fish that become entrained and survive passage, to determine how dam operations and environmental conditions might influence smallmouth bass population dynamics. When entrainment rates are relatively high (mean estimate of greater than or equal to 100 adult smallmouth bass per year), controlling the invasion could be challenging due to continued propagule pressure (Eppehimer et al. 2025). However, lower entrainment rates could be a management concern during early stages of the invasion, as is the current situation in the Colorado River below Glen Canyon Dam. The mean estimated abundance of adult smallmouth bass in the Glen Canyon Dam tailwater in 2022 and 2023 ranged from approximately 40-50 individuals, with 50 percent credible intervals ranging from approximately 25-55 individuals (Eppehimer et al. 2025). Propagule pressure from reservoirs is an important consideration for smallmouth bass population and invasion dynamics. In the Yampa and Green Rivers of the Upper Colorado River Basin (Breton et al. 2015), 2003-2009 mean estimates of propagule pressure from Elkhead Reservoir into the Yampa River by tagged, translocated smallmouth bass ranged from 21 to 522 individuals per year (Breton et al. 2013). Assuming a theoretical, constant reservoir elevation throughout a year, Lake Powell elevations of 3,536 feet and 3,520 feet correspond to estimated mean annual entrainment and survival rates of approximately 50 and 100 adult smallmouth bass, respectively (Eppehimer et al. 2025; Eppehimer and Yackulic 2026). Therefore, an estimated entrainment rate of greater than or equal to 50 adult smallmouth bass per year could represent more than a doubling of the population of reproductively capable fish. An entrainment of less than 50 individuals was chosen as the analysis threshold below.

River Flows

River flows under each alternative are reported under **TA 3**, Hydrologic Resources, and referenced as appropriate within this issue analysis as a baseline for flow conditions that may impact water quality and quantity as it relates to native fish, trout species, and smallmouth bass.

Since 1996, Reclamation has conducted High-Flow Experiment (HFE) releases from Glen Canyon Dam to manage sediment and maintain or increase sandbar size. These releases are much larger than typical base flows and are specifically designed to enhance sediment deposition. HFE releases are the only current method for generating river stages high enough to build significant sandbars. While HFEs can be as low as 31,500 cubic feet per second, flows of 34,000 cubic feet per second or higher are needed for effective sandbar formation. Most HFEs are of relatively short duration and do not use enough water to significantly impact Lake Powell's elevation or long-term flow conditions. HFEs are primarily a sediment management tool and have no direct effect on fish populations within Lake Powell. The only notable biological impact is on fish in the Grand Canyon, especially in the Lees Ferry sub-reach, due to the use of both penstocks and river outlet works, which release colder water, during HFEs. Please refer to **TA 5**, Geomorphology and Sediment Resources, for a description and analysis of the effects of HFEs downstream of Glen Canyon Dam.

Water Quality

Dissolved oxygen is a critical factor for fish health. Research on dissolved oxygen thresholds for both warmwater and coldwater fish species shows that salmonids are particularly vulnerable to mortality when exposed to low dissolved oxygen concentrations compared to warmwater species (Saari et al. 2018). Sustained dissolved oxygen levels below 3 milligrams per liter (mg/L) can significantly reduce survival rates and feeding efficiency, whereas concentrations in the range of 6–9 mg/L are considered optimal for growth and survival across all life stages (USEPA 1986).

Analysis of minimum annual dissolved oxygen concentrations released from Glen Canyon Dam (**Figure TA 6-11**) shows that, in the Average and Wet Flow Categories (12–14 maf and 16–31.11 maf), all management alternatives result in similar median dissolved oxygen levels between 7 and 8 mg/L, with relatively narrow interquartile ranges. As flow categories become drier, both the median dissolved oxygen concentrations and interquartile ranges decrease, and variability among alternatives increases. In dry scenarios, the Enhanced Coordination and Maximum Operational Flexibility Alternatives generally maintain higher minimum annual dissolved oxygen concentrations compared to other alternatives, making them more robust against low dissolved oxygen and low reservoir elevations during extended droughts.

Robustness analysis (see **Figure TA 6-12** in **TA 6.2.4**, Issue 3) demonstrates that the Enhanced Coordination and Maximum Operational Flexibility Alternatives have the greatest percentage of futures where minimum annual dissolved oxygen concentrations exceed the hypoxic threshold of 2 mg/L in at least 90 percent of years, across the full modeling period. In contrast, the CCS Comparative Baseline and the Supply Driven Alternative (both LB Priority and LB Pro Rata approaches) have the fewest futures meeting this criterion. When considering operational constraints—specifically, Lake Powell elevations below 3,490 feet and the use of river outlet works (with a conservative minimum dissolved oxygen assumption of 8 mg/L)—the Enhanced Coordination and Maximum Operational Flexibility Alternatives remain the most robust (see Figure **TA 6-13** in **TA 6.2.4**, Issue 3). These alternatives are the only ones to maintain dissolved oxygen concentrations above 2 mg/L in at least 90 percent of years in a majority of futures. Similar alternative rankings occur in higher thresholds (e.g., greater than 5 mg/L) with reduced probabilities.

Vulnerability analysis (see **Figure TA 6-14** in **TA 6.2.4**, Issue 3) reveals that the CCS Comparative Baseline, No Action, Basic Coordination, and Supply Driven Alternatives are susceptible to undesirable performance (dissolved oxygen below 2 mg/L or Lake Powell below 3,490 feet) under hydrologic conditions similar to those already observed in the reference ensemble, including the driest recent 10- and 20-year periods. By contrast, the Enhanced Coordination and Maximum Operational Flexibility Alternatives only show such vulnerability at much lower flows (below 9.4 maf and 10.6 maf, respectively), which are lower than the lowest 25 percent of historic hydrologic conditions.

While all alternatives perform similarly under wet conditions, the Enhanced Coordination and Maximum Operational Flexibility Alternatives are more resilient to low dissolved oxygen and reservoir elevations during droughts. The general performance comparisons by alternative would hold true at higher thresholds necessary to maintain fish health.

Water Temperature

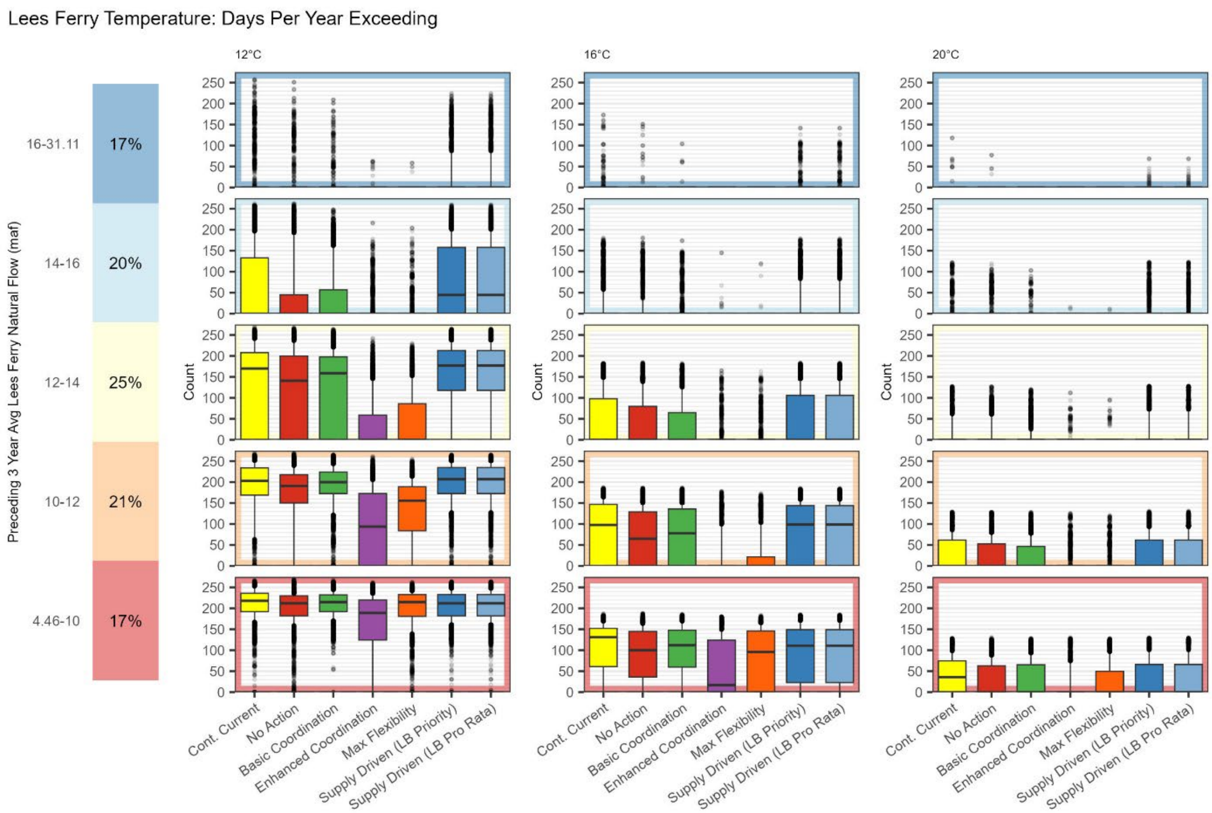
Water temperatures are presented using DMDU box plot figures, robustness heat maps, and vulnerability bar plots for four key locations along the Colorado River between Glen Canyon Dam and Lake Mead: Lees Ferry, the Little Colorado River confluence, the Havasu Creek confluence, and Pearce Ferry. This approach highlights longitudinal warming patterns as the river flows downstream from Glen Canyon Dam and provides a basis for comparing projected temperature outcomes across assessed futures. By displaying results in this format, we can evaluate how Lake Powell water temperatures, dam operations, and hydrology may influence thermal regimes under the DMDU platform. Performance qualifying years are those in which a single year during the 34-year modeling horizon with fewer days means that the entire future is considered a failure. Thus, performance percentages may be low, but relative in ranking across alternatives.

Lees Ferry

Across the boxplots in each flow category, the full range of potential resource impacts for each alternative is shown at Lees Ferry under temperature thresholds of 12, 16, and 20 °C (**Figure TA 8-9**). Exceeding the 12 °C threshold improves humpback chub growth rates (exceeding is a desirable response), exceeding the 16 °C threshold improves smallmouth bass reproduction in Lees Ferry (exceeding is an undesirable response), and exceeding the 20 °C threshold reduces rainbow trout survival (exceeding is an undesirable response). Based on the preceding 3-year average natural flow at Lees Ferry under each of the five alternatives, each temperature threshold is assessed based on the number of days in which modeled traces exceed the threshold compared to different flow regimes.

For 12 °C, in the Average Flow Category (12–14 maf), the No Action and Basic Coordination Alternatives have similar interquartile ranges spanning from 0 to approximately 200 days, but different medians (141 and 159, respectively) (**Figure TA 8-9**). Although the CCS Comparative Baseline has a higher median (170 days), the interquartile ranges are similar (0 to 208 days). The Enhanced Coordination and Maximum Operational Flexibility Alternatives exceed 12 °C less than the other three alternatives, and the CCS Comparative Baseline with interquartile ranges from 0 days up to 59 and 86 days, respectively. The Supply Driven Alternative (both LB Priority and LB Pro Rata approaches) has a higher median (177 days) with a narrower interquartile range (118 to 213 days) compared to all other alternatives and the CCS Comparative Baseline.

Figure TA 8-9
Water Temperature at Lees Ferry Based on 3-year Preceding Average River Temperatures at Lees Ferry Exceeding 12° C, 16 °C, and 20 °C Under Each Alternative



If drier conditions are present (i.e., 10–12 and 4.46–10 maf), similar data patterns to the 12–14 maf Average Flow Category exist with higher medians and smaller interquartile ranges across alternatives (**Figure TA 8-9**). Under the driest conditions (4.46–10 maf), temperatures are expected to exceed 12 **°**C at a higher probability and frequency across all alternatives, with the Enhanced Coordination Alternative having the greatest interquartile range of 124.5 to 220 days.

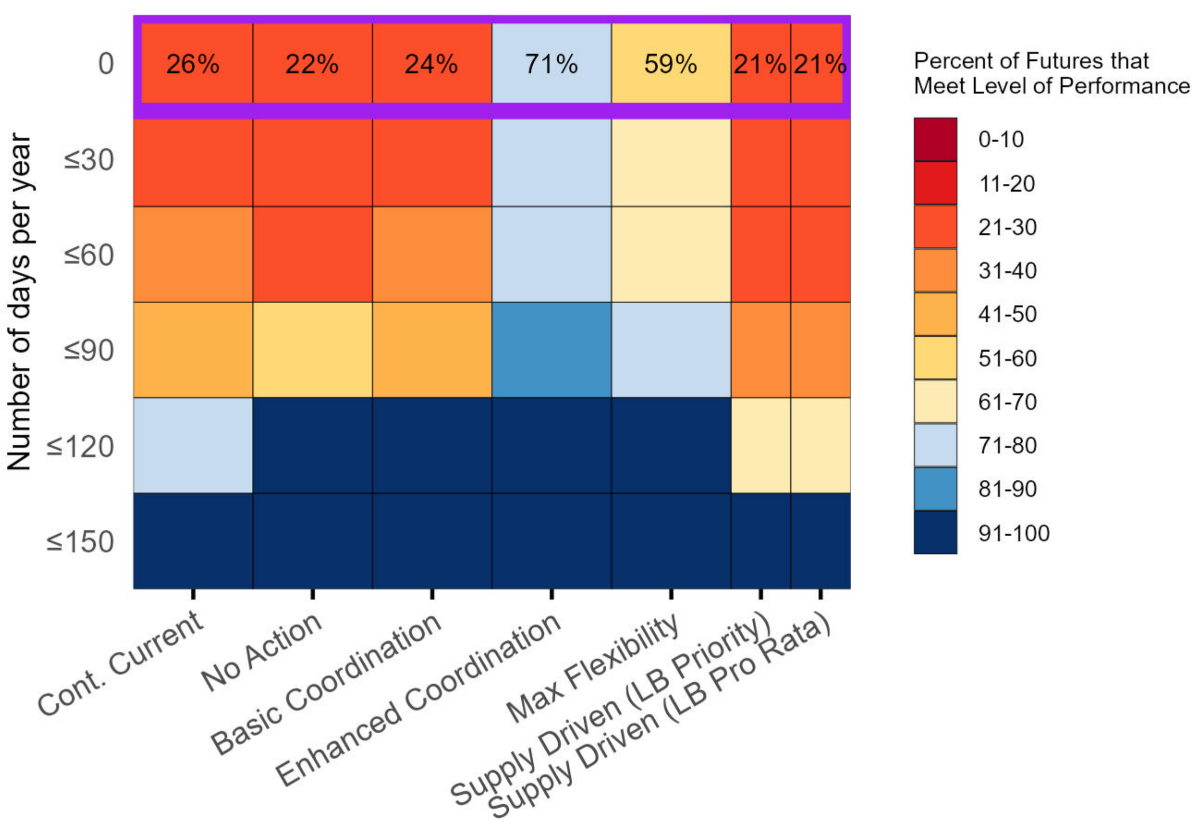
At 16 °C, the variability is similar to that at 12 **°**C, but the median number of days exceeding 16 °C is 0 days across all alternatives at 12–14 maf (**Figure TA 8-9**). At the same flows, the Enhanced Coordination and Maximum Operational Flexibility Alternatives have relatively few traces with days above 16 °C. Under drier conditions, the pattern is similar, but the median number of days increases among all alternatives and the CCS Comparative Baseline.

At 20 °C, the variability remains similar across alternatives, but the likelihood of exceeding 20 °C under moderate flow conditions is greatly reduced (**Figure TA 8-9**). However, under low flow conditions (less than 10 maf), the likelihood increases.

Each alternative was evaluated based on how it influences the likelihood of water temperatures exceeding 20 °C. According to the life histories of rainbow trout and native Grand Canyon fishes, temperatures above 20 °C are likely to decrease rainbow trout survival while creating conditions more favorable for the growth of native fish and smallmouth bass. Additionally, warmer water temperatures may increase the competitive advantage for brown trout, as this species is more tolerant of elevated temperatures.

Figure TA 8-10 compares alternatives with respect to their time in days that allow for water temperatures to exceed 20 °C. The comparison described here focuses on the preference for water temperatures to remain below 20 °C for all days during the analysis period, in the purple highlighted row, because it reduces the threat of rainbow trout mortality due to thermal exposure. Rows towards the top of the figure require fewer days below 20 °C and are harder to achieve, while rows towards the bottom of the figure are easier to achieve.

Figure TA 8-10
Rainbow Trout and Native Grand Canyon Fishes Survival: Robustness.
Percent of futures in which the number of days exceeding 20 °C at Lees Ferry is less than the number of days specified by each row

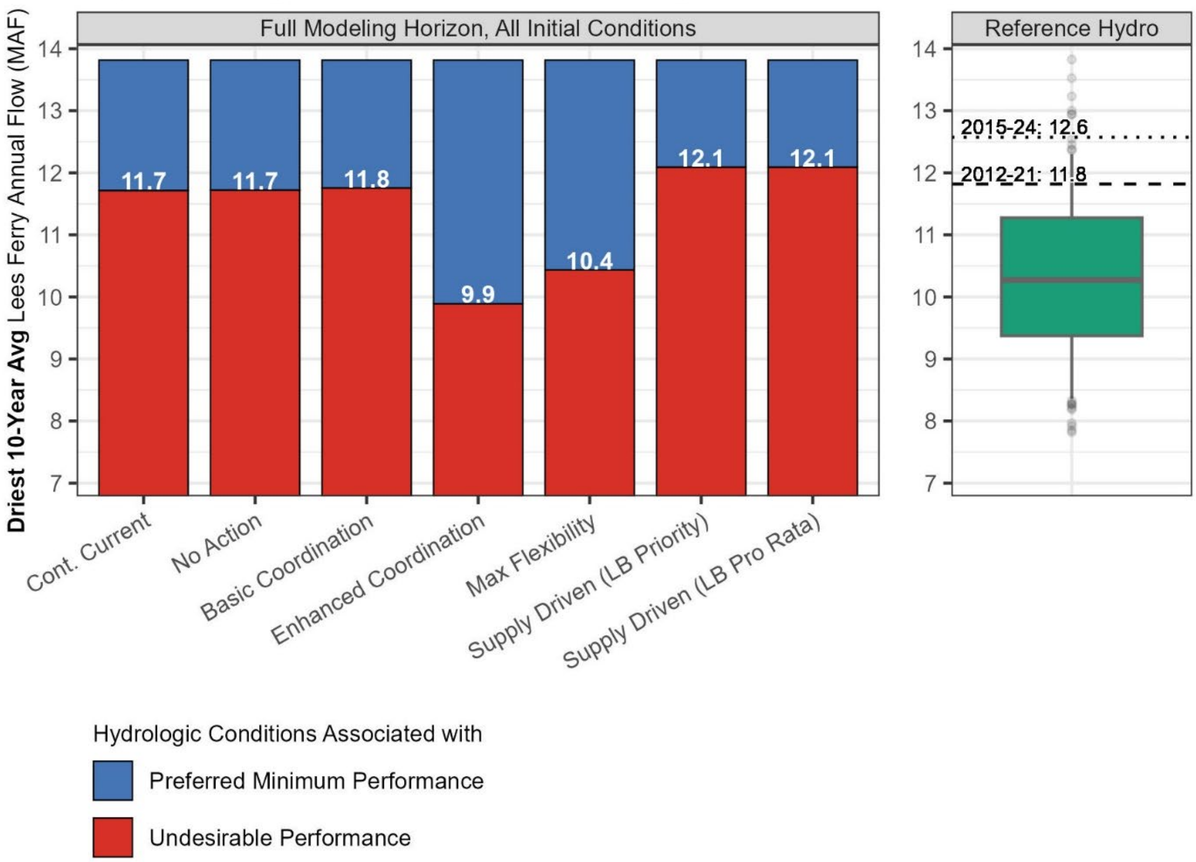


Over the full modeling period, the Enhanced Coordination Alternative is the most robust, meeting the preferred level of performance in 71 percent of futures (**Figure TA 8-10**). Since a greater than 10 percent difference would be considered significant, the Maximum Operational Flexibility Alternative performs significantly lower than the Enhanced Coordination Alternative at 59 percent, but is significantly higher than the other three alternatives. The No Action and Supply Driven

Alternatives (both LB Priority and LB Pro Rata approaches) perform similarly and are the least robust (21–26 percent) and perform similarly to CCS Comparative Baseline.

Figure TA 8-10 highlights the percentage of futures for each alternative in which the water temperature at Lees Ferry exceeds 20 °C. To identify hydrologic conditions that could cause undesirable performance, the incidence of Lees Ferry temperatures exceeding 20 °C in one or more days out of the 34-year modeling period was analyzed (**Figure TA 8-11**). The driest 10-year average Lees Ferry natural flow was identified as a good predictor of undesirable performance. The reference hydrology panel shows the range of driest 10-year average flows represented in a reference hydrology, along with the most recent and driest observed 10-year periods.

Figure TA 8-11
Rainbow Trout and Native Grand Canyon Fishes Survival: Vulnerability. Conditions that could cause the temperature at Lees Ferry to exceed 20°C in one or more days



If future conditions include a 10-year average flow of 11.7 maf or lower, both the CCS Comparative Baseline and the No Action Alternative are likely to perform undesirably; over 75 percent of reference hydrology traces fall at or below this threshold (**Figure TA 8-11**). Similarly, the Basic

Coordination Alternative is vulnerable if the 10-year average flow drops to 11.8 maf or lower, with more than 75 percent of traces meeting this criterion. For the Enhanced Coordination Alternative, undesirable performance is expected if the average flow falls to 9.9 maf or lower, with less than 50 percent of traces in the reference hydrology experiencing these conditions. The Maximum Operational Flexibility Alternative is likely to result in undesirable performance at 10.4 maf or lower, which corresponds to about 50 percent of traces in the reference hydrology. The Supply Driven Alternative (LB Priority and LB Pro Rata) is at risk if the flow averages 12.1 maf or less, which corresponds to over 75 percent of traces in the reference hydrology. Notably, the driest 10-year period in the observed record averaged approximately 11.8 maf, indicating that the No Action, Basic Coordination, Supply Driven Alternatives, and the CCS Comparative Baseline are all susceptible to conditions similar to those already experienced.

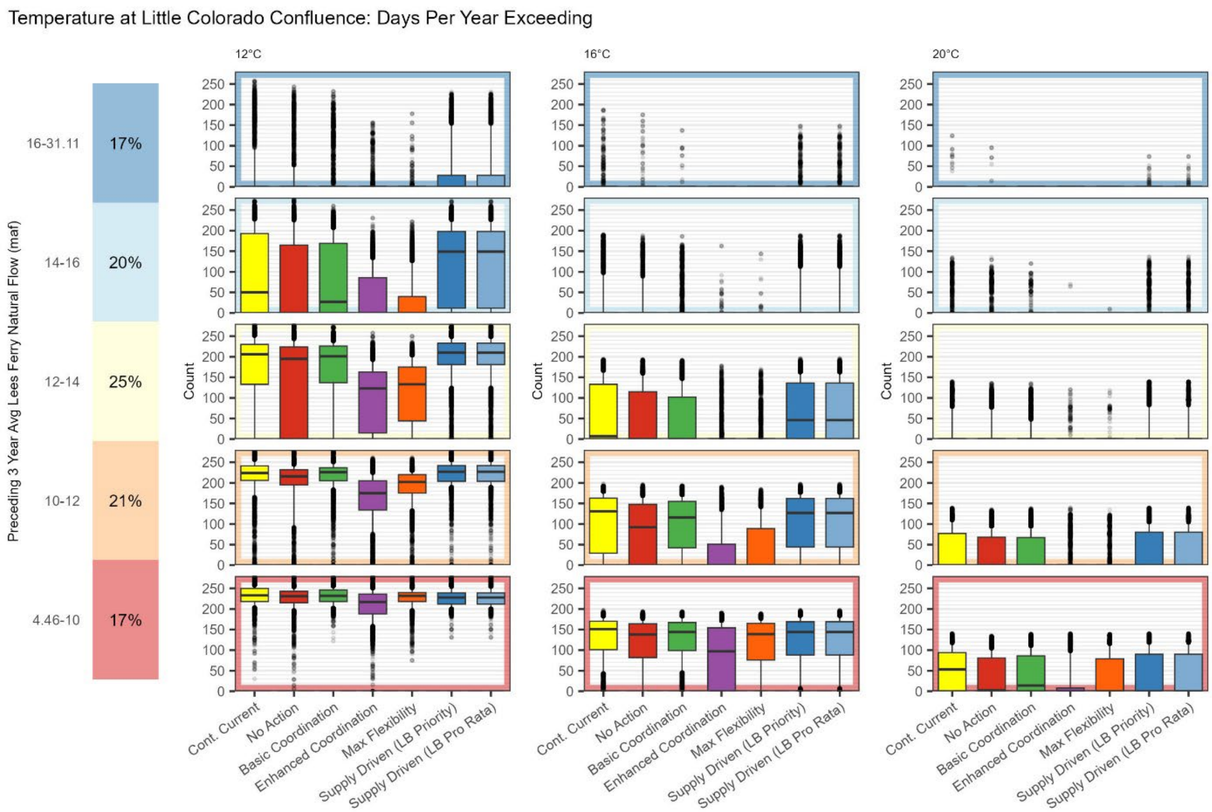
The Enhanced Coordination Alternative is the most robust for maintaining cooler water temperatures at Lees Ferry, benefiting rainbow trout and limiting smallmouth bass reproduction, but potentially inhibiting native fish growth. Alternatives that support higher river flows and Lake Powell elevations generally perform better than the No Action Alternative and CCS Comparative Baseline in sustaining cooler temperatures. However, during extended droughts, all alternatives become more vulnerable to rising temperatures, which can negatively affect fish habitat and species composition. In less robust scenarios, warmer water may expand available habitat near Lees Ferry, increasing the presence of native fish and smallmouth bass, and potentially favoring brown trout over rainbow trout.

Little Colorado River Confluence

Across the boxplots in each flow category, the full range of potential resource impacts for each alternative is shown at the Lower Colorado River confluence under temperature thresholds of 12, 16, and 20 °C (**Figure TA 8-12**). Based on the preceding 3-year average natural flow at Lees Ferry under each of the five alternatives, each temperature threshold is assessed based on the number of days in which modeled traces exceed the threshold compared to different flow regimes.

For 12 °C, in the Average Flow Category (12–14 maf), the Basic Coordination Alternative and CCS Comparative Baseline have similar interquartile ranges spanning from 133 to approximately 230 days, but different medians (201 and 206 days, respectively) (**Figure TA 8-12**). The No Action Alternative has a similar median to Basic Coordination Alternative and CCS Comparative Baseline with a broader interquartile range of 0–224 days. The Enhanced Coordination and Maximum Operational Flexibility Alternatives exceed 12 °C less than the other three alternatives, and the CCS Comparative Baseline with interquartile ranges from 15 days up to 163 and 44 days up to 175 days, respectively. The Supply Driven Alternative (both LB Priority and LB Pro Rata approaches) have a higher median (210 days) with a narrower interquartile range (181 to 233 days) compared to all other alternatives and the CCS Comparative Baseline.

Figure TA 8-12
Water Temperature at the Little Colorado River Confluence Based on 3-year Preceding Average River Temperatures at Lees Ferry Exceeding 12 °C, 16 °C, and 20 °C Under Each Alternative



As drier conditions become present (i.e., 10–12 and 4.46–10 maf), similar data patterns to the 12-14 maf Average Flow Category exist with higher medians and smaller interquartile ranges across alternatives (Figure 3.12). Under the driest conditions (4.46–10 maf), temperatures are expected to exceed 12 °C at a higher probability and frequency across all alternatives, with the Enhanced Coordination Alternative having the greatest interquartile range of 188 to 236 days.

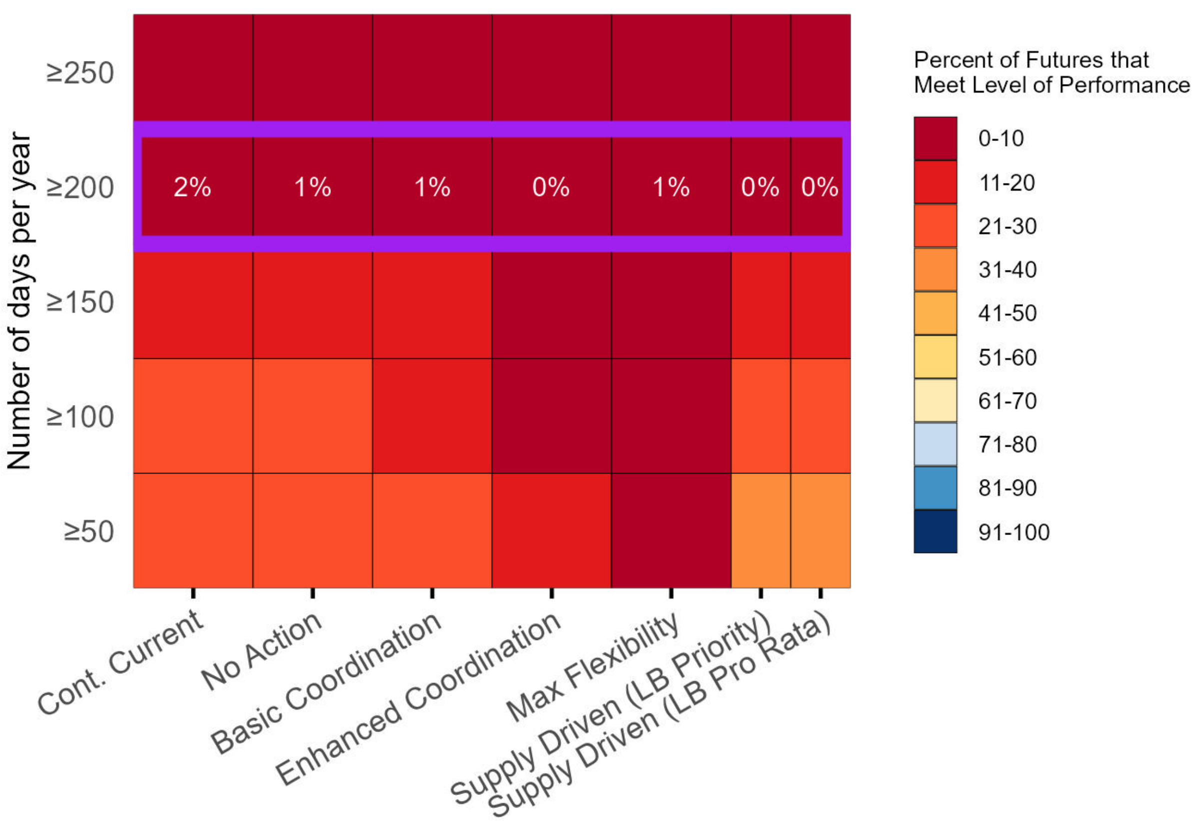
At 16 °C, the variability is similar to that at 12 °C, but the median number of days exceeding 16 °C is 0 days across all alternatives at 12-14 maf (Figure 3.12). At the same flows, the Enhanced Coordination and Maximum Operational Flexibility Alternatives have relatively few traces with days above 16 °C. Under drier conditions, the pattern is similar, but the median number of days increases among all alternatives and the CCS Comparative Baseline.

At 20 °C, the variability remains similar across alternatives, but the likelihood of exceeding 20 °C under moderate flow conditions is greatly reduced (Figure 3.12). However, under low flow conditions (less than 10 maf), the likelihood increases.

Each alternative was analyzed based on how it influences the likelihood of water temperatures exceeding 12 °C and 16 °C. Based on smallmouth bass and native Grand Canyon fishes life histories, temperatures above 12 °C would be conducive to humpback chub growth. Water temperatures exceeding 16 °C promote humpback chub and smallmouth bass spawning. Although warmer water temperatures are beneficial to native fish, such as humpback chub, maintaining cooler water temperatures may limit smallmouth population growth within critical humpback chub habitats.

Figure TA 8-13 compares alternatives with respect to their time in days that allow for water temperatures to exceed 12 °C, which promotes humpback chub growth. The comparison described here focuses on the preference for water temperatures to exceed 12 °C for at least 200 days during the analysis period, in the purple highlighted row. That threshold was selected because it is approximately the historical, modeled 10th percentile at the Lower Colorado River. Higher rows capture a greater percentage of time and are harder to achieve, while lower rows represent lower time values that are easier to achieve.

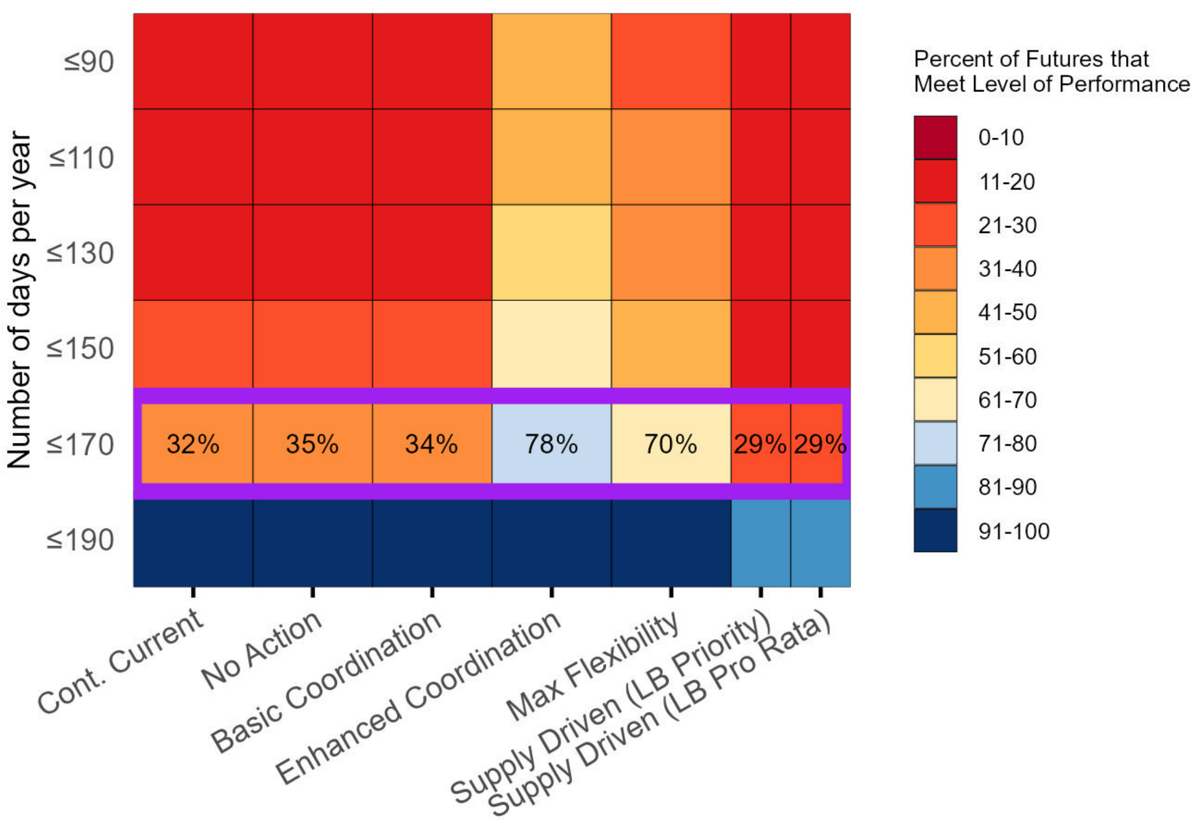
Figure TA 8-13
Humpback Chub Growth at the Little Colorado River Confluence: Robustness. Percent of futures in which the temperature at the Littler Colorado River confluence exceeds 12 °C in the days per year specified by each row



Over the full modeling period, all of the alternatives perform similarly, and none of the alternatives perform well at this temperature and within the threshold analyzed (**Figure TA 8-13**). Decreasing the threshold below 100 days indicates some variation among alternatives, but also poor performance.

Figure TA 8-14 compares alternatives with respect to their time in days that allow for water temperatures to exceed 16 °C, which promotes humpback chub spawning but also increases the likelihood of successful smallmouth bass spawning. The comparison described here focuses on the preference for water temperatures to exceed 16 °C for 170 days or less during the analysis period, in the purple highlighted row. That threshold was selected because it is approximately the historical, modeled 10th percentile at the Lower Colorado River. Higher rows capture a lower percentage of time and are harder to achieve, while lower rows represent higher time values that are easier to achieve.

Figure TA 8-14
Humpback Chub Spawning at the Little Colorado River Confluence: Robustness. Percent of futures in which the temperature at the Littler Colorado River confluence exceeds 16 °C in the days per year specified by each row



Over the full modeling period, the Basic Coordination and Supply Driven (both LB Priority and LB Pro Rata approaches) Alternatives are similarly robust to the No Action Alternative and the CCS Comparative Baseline (29–35 percent). The Enhanced Coordination and Maximum Operational

Flexibility Alternatives perform similarly with higher robustness at 70 percent and 78 percent, respectively (**Figure TA 8-14**). As the number of days decreases, robustness decreases across all alternatives. If selecting for a reduced number of days exceeding 16 °C to reduce the threat of smallmouth bass spawning, the Enhanced Coordination and Maximum Operational Flexibility Alternatives both perform better than the No Action Alternative and CCS Comparative Baseline.

Exceeding the 12 °C threshold creates thermal conditions that support mainstem humpback chub growth. While these warmer temperatures can benefit native humpback chub by enhancing growth and recruitment, temperatures exceeding 16 °C also favor smallmouth bass, which may increase predation and competition pressures on native fish populations. The comparative analysis of alternatives at the 12 °C and 16 °C thresholds is critical for identifying management scenarios that balance the needs of native and nonnative species. Alternatives that limit the number of days above these temperature thresholds (Enhanced Coordination and Maximum Operational Flexibility Alternatives) may help reduce smallmouth bass population growth but could also restrict opportunities for humpback chub reproduction.

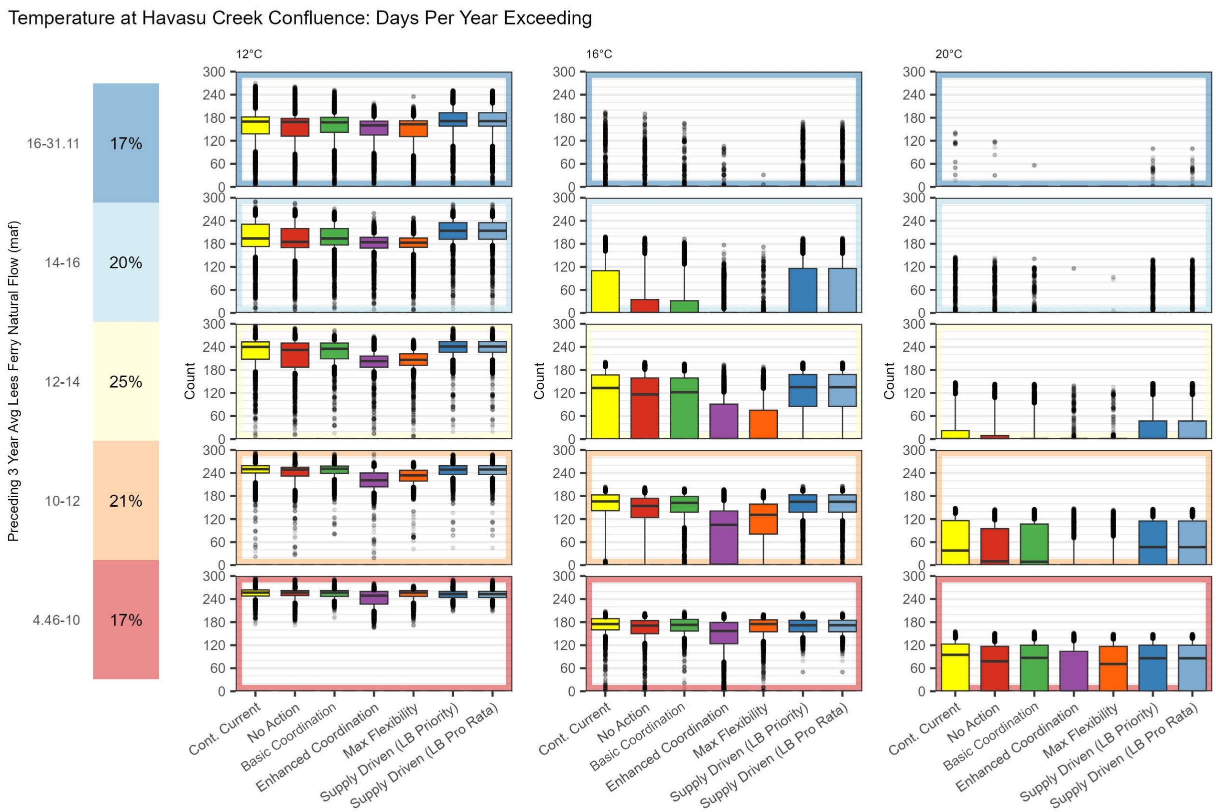
Havasu Creek Confluence

Across the boxplots in each flow category, the full range of potential resource impacts for each alternative is shown at the Havasu Creek confluence under temperature thresholds of 12, 16, and 20 °C (**Figure TA 8-15**). Based on the preceding 3-year average natural flow at Lees Ferry under each of the five alternatives, each temperature threshold is assessed based on the number of days in which modeled traces exceed the threshold compared to different flow regimes.

For 12 °C, in the Average Flow Category (12–14 maf), the Basic Coordination Alternative and CCS Comparative Baseline have similar interquartile ranges spanning from 208 to approximately 253 days, but different medians (235 and 240 days, respectively) (**Figure TA 8-15**). No Action Alternative has a similar median to the Basic Coordination Alternative and CCS Comparative Baseline, with a broader interquartile range of 187–250 days. The Enhanced Coordination and Maximum Operational Flexibility Alternatives exceed 12 °C less than the other three alternatives, and the CCS Comparative Baseline with interquartile ranges from 187 days up to 216 and 192 days up to 222 days, respectively. The Supply Driven Alternative (both LB Priority and LB Pro Rata approaches) has a higher median (241 days) with a narrower interquartile range (226 to 254 days) compared to all other alternatives and the CCS Comparative Baseline. If drier conditions are present (i.e., 10–12 and 4.46–10 maf), similar data patterns to the 12–14 maf Average Flow Category exist with higher medians and smaller interquartile ranges across alternatives (**Figure TA 8-15**). Under the driest conditions (4.46–10 maf), temperatures are expected to exceed 12 °C at a similar probability and frequency across all alternatives, with the Enhanced Coordination Alternative having the greatest interquartile range of 226 to 254 days.

At 16 °C, the variability is similar to that at 12 °C, but the median number of days exceeding 16 °C is 0 days for the Enhanced Coordination and Maximum Operational Flexibility Alternatives at 12–14 maf and ranges from 116–135 days across the other alternatives and the CCS Comparative Baseline (**Figure TA 8-15**). Under drier conditions, the pattern is similar, but the interquartile ranges decrease among all alternatives and the CCS Comparative Baseline.

Figure TA 8-15
Water Temperature at the Havasu Creek Confluence Based on 3-year Preceding Average River Temperatures at Lees Ferry Exceeding 12 °C, 16 °C, and 20 °C Under Each Alternative



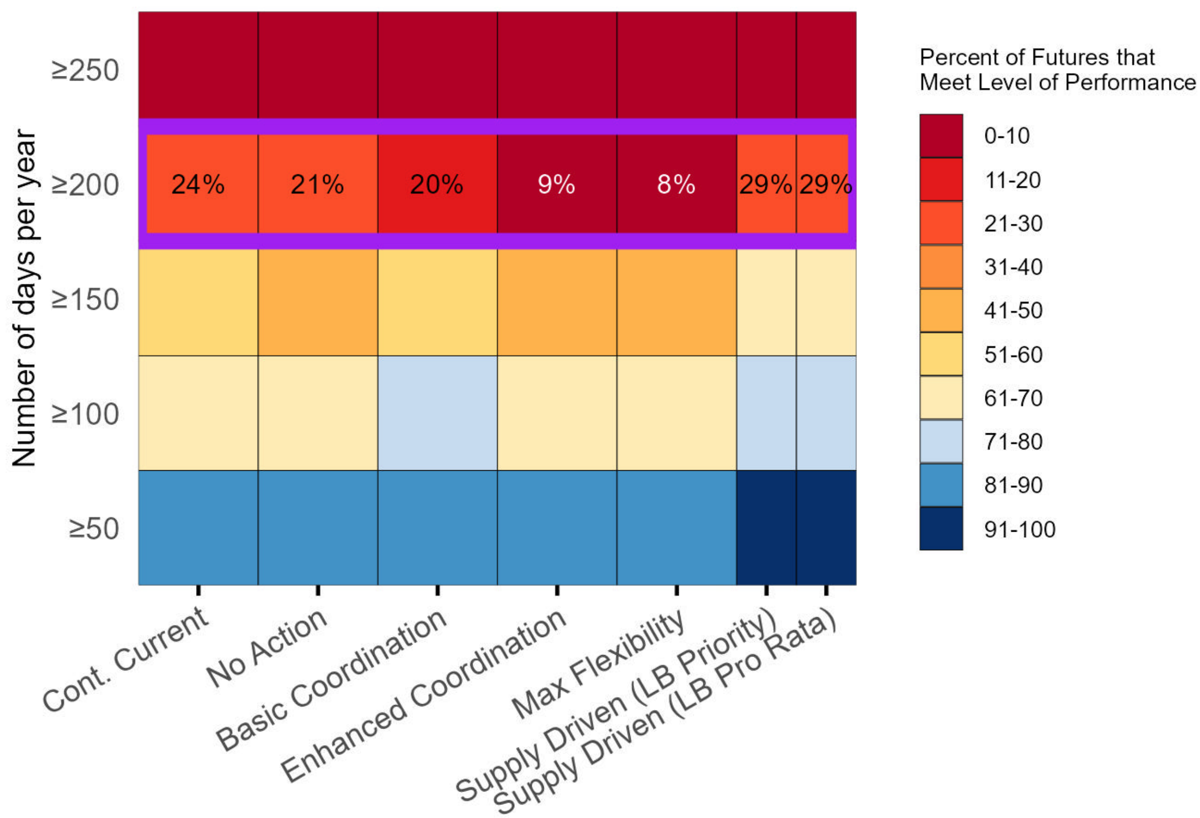
At 20 °C, the variability remains similar across alternatives, but the likelihood of exceeding 20 °C under moderate flow conditions is greatly reduced (**Figure TA 8-15**). However, under low flow conditions (less than 10 maf), the likelihood of exceeding 20 °C increases. The Enhanced Coordination Alternative has the lowest median (0.0) under the driest conditions at 20 °C.

Like the Lower Colorado River, the analysis consists of the effects of each alternative on the likelihood of water temperatures exceeding 12 °C and 16 °C at the Havasu Creek confluence. **Figure TA 8-16** compares alternatives with respect to their time in days that allow for water temperatures to exceed 12 °C, which promotes humpback chub growth. The comparison described here focuses on the preference for water temperatures to exceed 12 °C for at least 200 days during the analysis period, in the purple highlighted row. That threshold was retained from the analysis at the Lower Colorado River for comparable results and discussion. Higher rows capture a greater percentage of time and are harder to achieve, while lower rows represent lower time values that are easier to achieve.

Over the full modeling period, none of the alternatives perform well with the No Action and Supply Driven (both LB Priority and LB Pro Rata approaches) Alternatives, and the CCS Comparative

Baseline perform better than the Basic Coordination, Enhanced Coordination, and Maximum Operational Flexibility Alternatives at greater than 200 days (**Figure TA 8-16**). Decreasing the threshold below 200 days indicates better performance, but with similar variation between alternatives. Among differing thresholds (days), the Supply Driven Alternative (both LB Priority and LB Pro Rata approaches) performs similarly to the No Action Alternative and CCS Comparative Baseline.

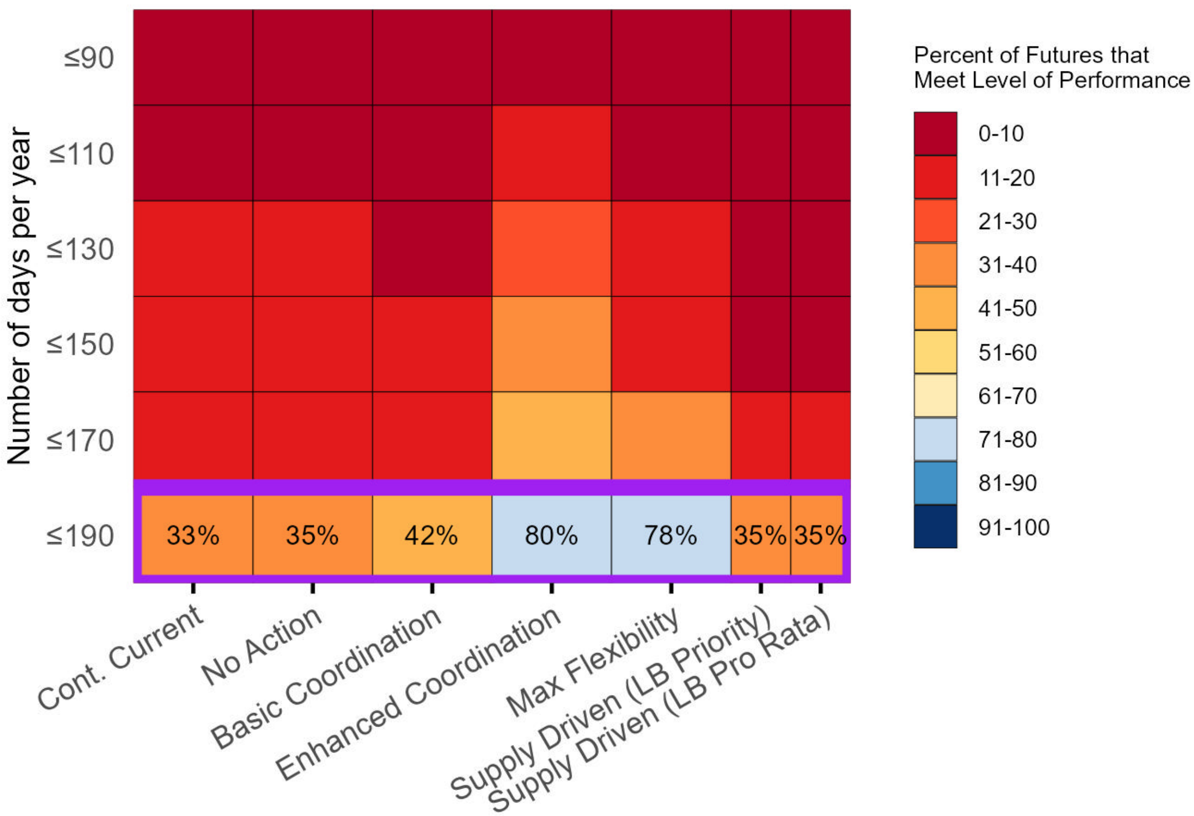
Figure TA 8-16
Humpback Chub Growth at the Havasu Creek Confluence: Robustness.
Percent of futures in which the temperature at the Havasu Creek confluence exceeds 12 °C in the days per year specified by each row



Similar to the Lower Colorado River analysis, **Figure TA 8-17** compares alternatives with respect to their time in days that allow for water temperatures to exceed 16 °C, which promotes humpback chub spawning but also increases the likelihood of successful smallmouth bass spawning. The comparison described here focuses on the preference for water temperatures to exceed 16 °C for 190 days or less during the analysis period, in the purple highlighted row. That threshold was selected because it is approximately the historical, modeled 10th percentile at Havasu Creek. Higher rows capture a lower percentage of time and are harder to achieve, while lower rows represent higher time values that are easier to achieve.

Over the full modeling period, Basic Coordination and Supply Driven (both LB Priority and LB Pro Rata approaches) Alternatives are similarly robust to the No Action Alternative and the CCS Comparative Baseline (33–42 percent). The Enhanced Coordination and Maximum Operational Flexibility Alternatives perform similarly with higher robustness at 80 percent and 78 percent, respectively (**Figure TA 8-17**). As the number of days decreases, robustness decreases across all alternatives. If selecting for a reduced number of days exceeding 16 °C to reduce the threat of smallmouth bass spawning, the Enhanced Coordination and Maximum Operational Flexibility Alternatives both perform better than the No Action Alternative and the CCS Comparative Baseline.

Figure TA 8-17
Humpback Chub Spawning at the Havasu Creek Confluence: Robustness. Percent of futures in which the temperature at the Havasu Creek confluence exceeds 16 °C in the days per year specified by each row



As with the Lower Colorado River, exceeding the 12 °C threshold creates thermal conditions that support mainstem humpback chub growth. While these warmer temperatures can benefit native humpback chub by enhancing growth and recruitment, temperatures exceeding 16 °C also favor smallmouth bass, which may increase predation and competition pressures on native fish populations. The comparative analysis of alternatives at the 12 °C and 16 °C thresholds is critical for identifying management scenarios that balance the needs of native and nonnative species. Alternatives that limit the number of days above these temperature thresholds (Enhanced

Coordination and Maximum Operational Flexibility Alternatives) may help reduce smallmouth bass population growth but could also restrict opportunities for humpback chub reproduction.

Pearce Ferry

Water temperature analysis at Pearce Ferry allows for the comparison of variability among alternatives, providing insight into the likelihood of exceeding levels that affect humpback chub and smallmouth bass population growth. Although water temperatures at Pearce Ferry can be influenced by Glen Canyon Dam release temperatures, the location is more influenced by ambient air temperatures. In the fall, when dam releases can be at their warmest, when the lake naturally turns over, and the warmer water is more aligned with the penstock, mainstem water temperatures may cool as they move downstream over 280 river miles. During warmer months, cooler releases from the dam result in cooler water moving downstream (Valdez et al. 2013). This can create a scenario where average and maximum temperatures may be more similar across alternatives at this location.

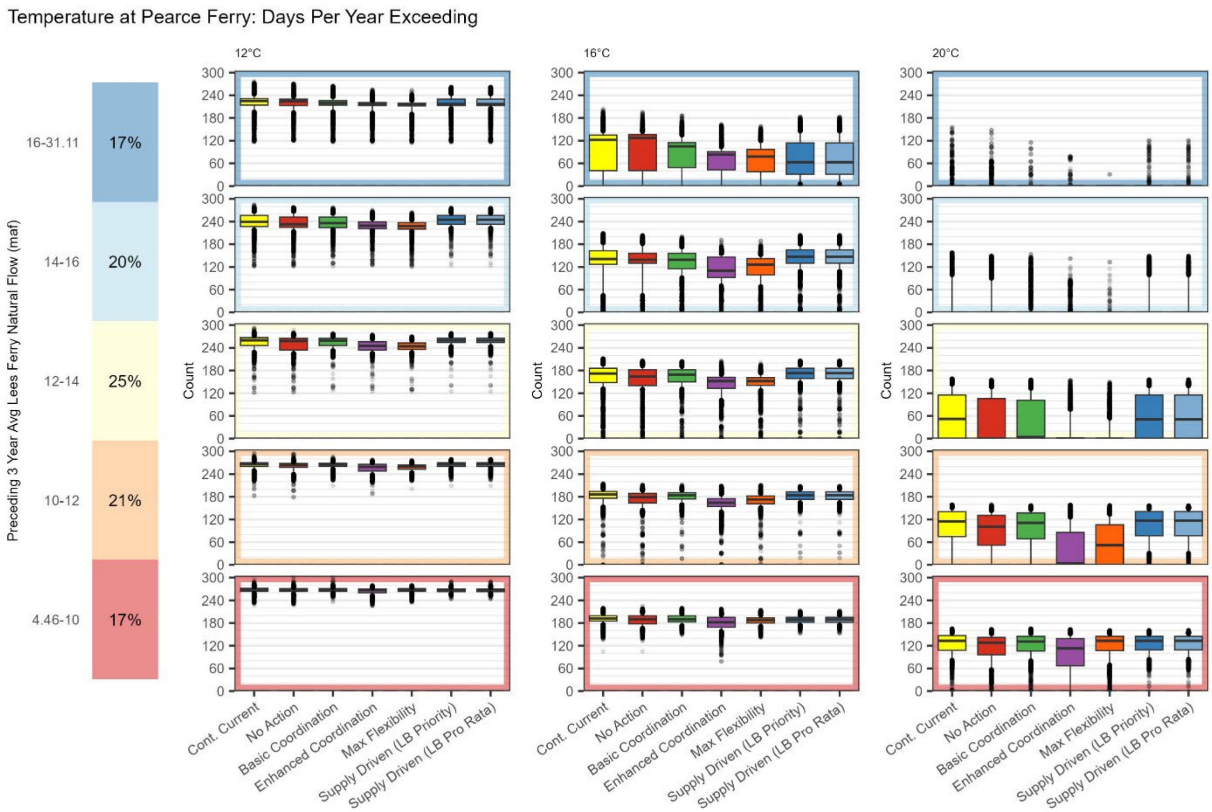
Across the boxplots in each flow category, the full range of potential resource impacts for each alternative is shown at the Havasu Creek confluence under temperature thresholds of 12, 16, and 20 °C (**Figure TA 8-18**). Based on the preceding 3-year average natural flow at Lees Ferry under each of the five alternatives, each temperature threshold is assessed based on the number of days in which modeled traces exceed the threshold compared to different flow regimes.

For 12 °C, in the Average Flow Category (12–14 maf), the Basic Coordination Alternative is similar to the No Action Alternative and CCS Comparative Baseline regarding interquartile ranges and medians (259, 258, and 260, respectively) (**Figure TA 8-18**). The Enhanced Coordination and Maximum Operational Flexibility Alternatives exceed 12 °C less than the other three alternatives, and the CCS Comparative Baseline with very similar interquartile ranges and medians (245 and 244 days, respectively). The Supply Driven Alternative (both LB Priority and LB Pro Rata approaches) has a similar median to the No Action Alternative and CCS Comparative Baseline, with a narrower interquartile range (254 to 266 days) compared to all other alternatives and the CCS Comparative Baseline. As drier conditions are present (i.e., 10–12 and 4.46–10 maf), similar data patterns to the 12–14 maf Average Flow Category exist with slightly higher medians and smaller interquartile ranges across alternatives (**Figure TA 8-18**). Under the driest conditions (4.46–10 maf), temperatures are expected to exceed 12 °C at a similar probability and frequency across all alternatives.

At 16 °C, the variability is similar to that at 12 °C, but with lower median values and larger interquartile ranges across all alternatives and the CCS Comparative Baseline under all flow categories (**Figure TA 8-18**). Under drier conditions, interquartile ranges decrease among all alternatives and the CCS Comparative Baseline.

At 20 °C, the variability remains similar across alternatives, but the likelihood of exceeding 20 °C under moderate flow conditions is greatly reduced (**Figure TA 8-18**). However, under low flow conditions (less than 10 maf), the likelihood increases. The Enhanced Coordination Alternative has the lowest median (182) under the driest conditions at 20 °C.

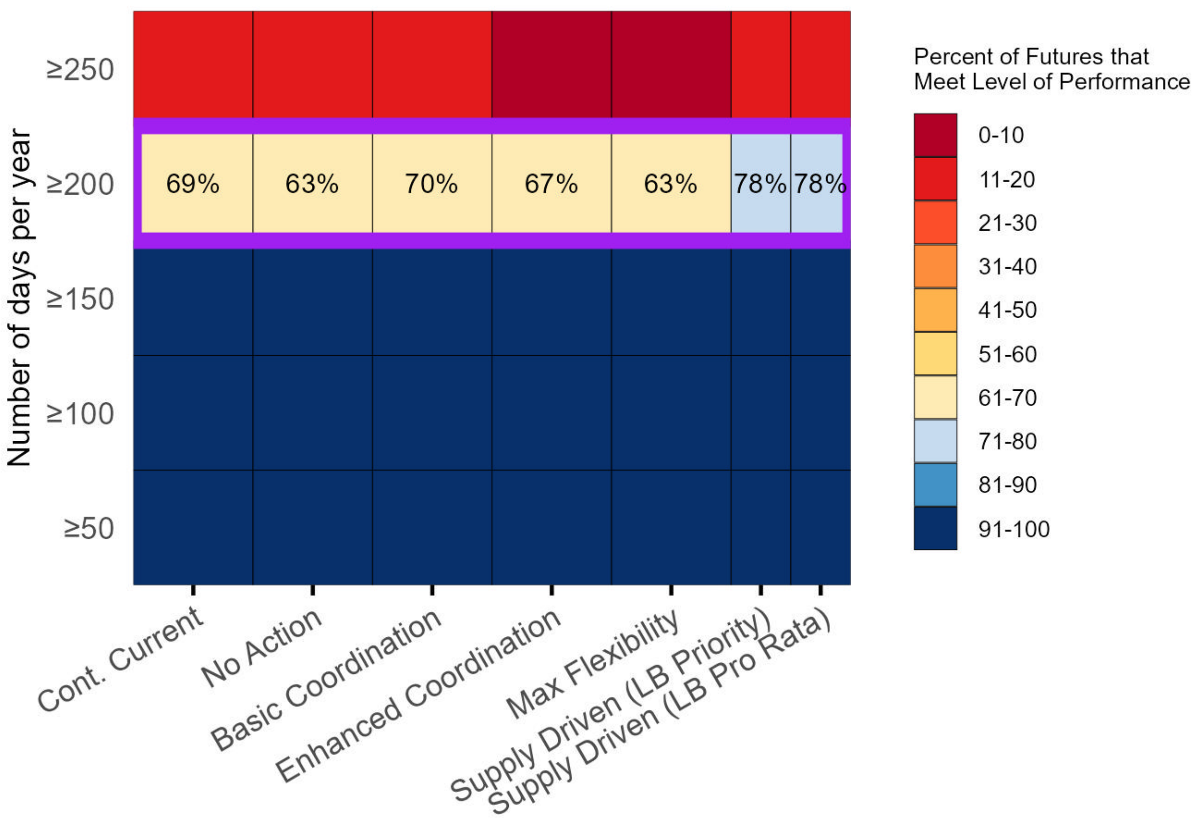
Figure TA 8-18
Water Temperature at Pearce Ferry Based on 3-year Preceding Average River Temperatures at Lees Ferry Exceeding 12 °C, 16 °C, and 20 °C Under Each Alternative



Like the Lower Colorado River, we analyzed how each alternative affects the likelihood of water temperatures exceeding 12 °C and 16 °C at Pearce Ferry. **Figure TA 8-19** compares alternatives with respect to their time in days that allow for water temperatures to exceed 12 °C, which promotes humpback chub growth. The comparison described here focuses on the preference for water temperatures to exceed 12 °C for at least 200 days during the analysis period, in the purple highlighted row. That threshold was retained from the analysis at the Lower Colorado River for comparable results and discussion. Higher rows capture a greater percentage of time and are harder to achieve, while lower rows represent lower time values that are easier to achieve.

Over the full modeling period, all alternatives perform relatively well, with the Supply Driven Alternative (both LB Priority and LB Pro Rata approaches) performing better than the Maximum Operational Flexibility, Enhanced Coordination, and No Action Alternatives at greater than 200 days (**Figure TA 8-19**). Decreasing the threshold below 200 days provides better performance with little variation between alternatives.

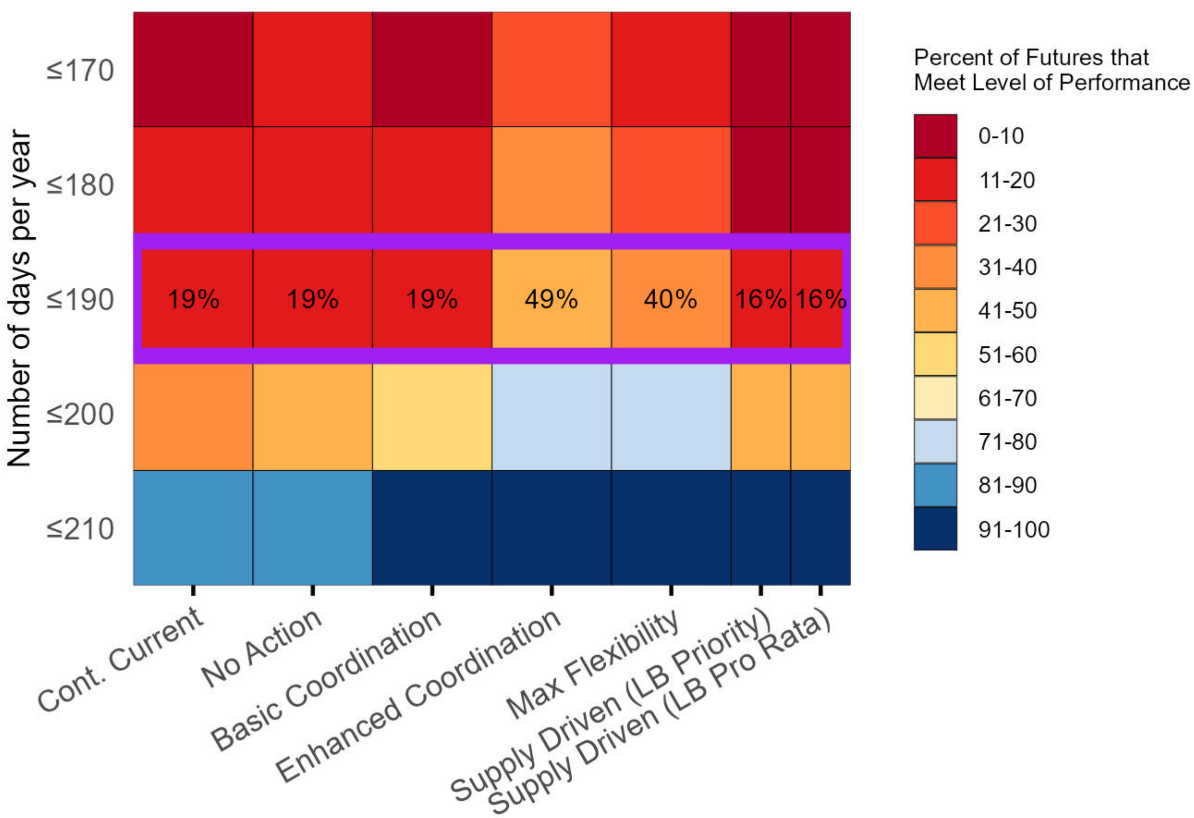
Figure TA 8-19
Humpback Chub Growth at Pearce Ferry: Robustness.
Percent of futures in which the temperature at Pearce Ferry exceeds 12 °C in the days per year specified by each row



Similar to the Lower Colorado River analysis, **Figure TA 8-20** compares alternatives with respect to their time in days that allow for water temperatures to exceed 16 °C, which promotes humpback chub spawning but also increases the likelihood of successful smallmouth bass spawning. The comparison described here focuses on the preference for water temperatures to exceed 16 °C for 190 days or less during the analysis period, in the purple highlighted row. That threshold was selected because it is approximately the historical, modeled 10th percentile at Pearce Ferry. Higher rows capture a lower percentage of time and are harder to achieve, while lower rows represent higher time values that are easier to achieve.

Over the full modeling period, Basic Coordination and Supply Driven (both LB Priority and LB Pro Rata approaches) Alternatives are similarly robust to the No Action Alternative and the CCS Comparative Baseline (16–19 percent). The Enhanced Coordination and Maximum Operational Flexibility Alternatives perform similarly with higher robustness at 49 percent and 40 percent, respectively (**Figure TA 8-20**). As the number of days decreases, robustness decreases across all alternatives. If selecting for a reduced number of days exceeding 16 °C to reduce the threat of smallmouth bass spawning, the Enhanced Coordination and Maximum Operational Flexibility Alternatives both perform better than the No Action Alternative and CCS Comparative Baseline down to 150 days.

Figure TA 8-20
Humpback Chub Spawning at Pearce Ferry: Robustness.
Percent of futures in which the temperature at Pearce Ferry exceeds 16 °C in the days per year specified by each row



As with the Lower Colorado River and Havasu Creek, exceeding the 12 °C threshold creates thermal conditions that support mainstem humpback chub growth. While these warmer temperatures can benefit native humpback chub by enhancing growth and recruitment, temperatures exceeding 16 °C also favor smallmouth bass, which may increase predation and competition pressures on native fish populations. The comparative analysis of alternatives at the 12 °C and 16 °C thresholds is critical for identifying management scenarios that balance the needs of native and nonnative species. Alternatives that limit the number of days above these temperature thresholds (Enhanced Coordination and Maximum Operational Flexibility Alternatives) may help reduce smallmouth population growth but could also restrict opportunities for humpback chub reproduction.

Smallmouth Bass Population Growth

An analysis comparing the percent of futures in which smallmouth bass lambdas are less than 1 was used to compare alternatives in controlling smallmouth bass survival and recruitment throughout the Grand Canyon. For each year and trace within an alternative, the predicted population growth rate, lambda, was calculated based on predicted water temperatures using the model described in Eppehimer et al. (2025). To keep smallmouth bass population growth declining, lambda values would need to remain below 1. Using a 5-year rolling product of lambda values in this analysis, lambda could still exceed 1 on any given year, but not necessarily result in positive or sustained population growth (Pine et al. 2025; Eppehimer, D., GCMRC, personal communication). For consistency with previous analyses (i.e., Reclamation 2024), we use less than 1 as the threshold for analysis and comparison across alternatives.

This figure shows the percentage of modeled futures under each alternative in which the 5-year smallmouth bass population growth rate (lambda) at Lees Ferry is less than 1. Lambda values below 1 indicate a declining smallmouth bass population, which is favorable for native fish conservation, whereas values above 1 indicate potential population growth and increased predation and competition with native fishes. The analysis allows comparison across alternatives to evaluate the likelihood that each scenario limits smallmouth bass population growth and, consequently, reduces predation pressure on native and listed fish species.

The 5-year product of smallmouth bass lambda at Lees Ferry is used to assess performance with respect to the growth of smallmouth bass in the Grand Canyon because it captures the importance of sequences of annual lambda values. **Figure TA 8-21** compares alternatives with respect to their maximum 5-year lambda in 100 percent of years because a single incidence of a high 5-year lambda could cause irreversible smallmouth bass population growth. The comparison described here focuses on the preference to stay below a maximum value of 1.0, in the purple highlighted row, because it accounts for variability while mathematically limiting the number of years that can be above 1.0 (signifying likely population increase in that year). Higher rows capture lower 5-year lambdas and are harder to achieve, while lower rows represent higher values that are easier to achieve.

Over the full modeling period, the Enhanced Coordination and Maximum Operational Flexibility Alternatives are the most robust, meeting the preferred level of performance in 69 percent and 57 percent of futures, respectively **Figure TA 8-21**. CCS Comparative Baseline and the No Action, Basic Coordination, and Supply Driven (both LB Priority and LB Pro Rata approaches) Alternatives perform similarly between 22 percent and 15 percent. The alternatives perform similarly in slightly higher 5-year products (easier, lower row) and slightly lower 5-year products (harder, higher row).

Figure TA 8-21 highlights the preferred minimum performance level of the 5-year smallmouth bass growth rate staying below 1.0, 100 percent of the time. To identify hydrologic conditions that could cause undesirable performance, the incidence of the 5-year growth rate is at least 1.0 in at least 1 year out of the 34-year modeling period was analyzed. The driest 10-year average Lees Ferry natural flow was identified as a good predictor of undesirable performance. The reference hydrology panel (**Figure TA 8-22**) shows the range of driest 10-year average flows represented in a reference hydrology, along with the most recent and driest observed 10-year periods. If the future includes a 10-year average flow of 11.8 maf or lower, the CCS Comparative Baseline and No Action Alternative are likely to result in undesirable performance; greater than 75 percent of the reference hydrology traces include averages this low or lower. If the future includes a 10-year average flow of 11.6 maf or lower, Basic Coordination is likely to result in undesirable performance; more than 75 percent of the reference hydrology traces include averages this low or lower. If the future includes a 10-year average flow of 10.7 maf or lower, the Enhanced Coordination Alternative is likely to result in undesirable performance; slightly less than 70 percent of the reference hydrology traces include averages this low or lower. If the future includes a 10-year average flow of 10.9 maf or lower, the Maximum Operational Flexibility Alternative is likely to result in undesirable performance; slightly more than 50 percent of the reference hydrology traces include averages this low or lower. If the future includes a 10-year average flow of 11.9 maf or lower, the Supply Driven Alternative (both LB Priority and LB Pro Rata approaches) is likely to result in undesirable performance; over 75 percent of the reference hydrology traces include averages this low or lower. The driest 10-year period in the observed record had an average of about 11.8 maf; CCS Comparative Baseline and the No Action, Basic Coordination, and Supply Driven (both LB Priority and LB Pro Rata approaches) Alternatives are all vulnerable to conditions that are close to what has already occurred.

Figure TA 8-21
5-Year Smallmouth Bass Growth Rate: Robustness.
Percent of futures in which the 5-year product of Smallmouth Bass growth rate (lambda) at Lees Ferry is always less than the value specified by each row

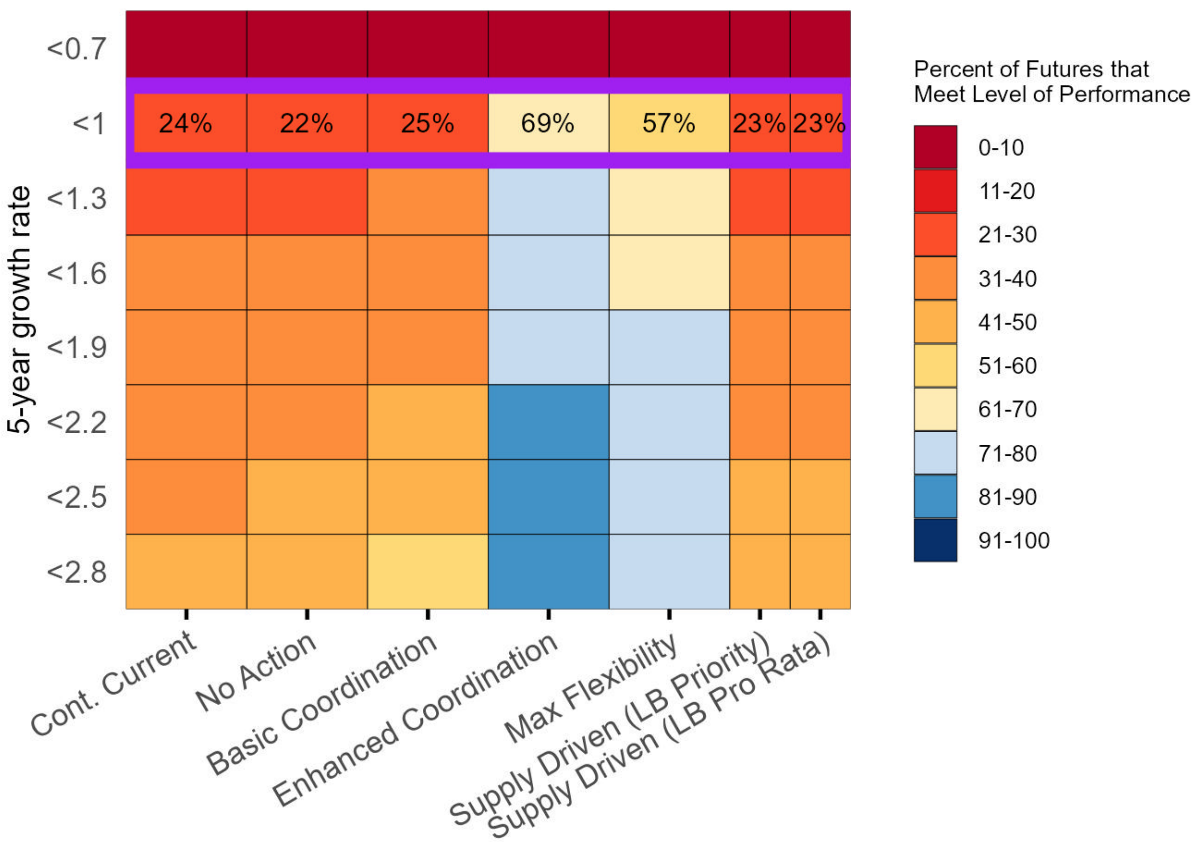
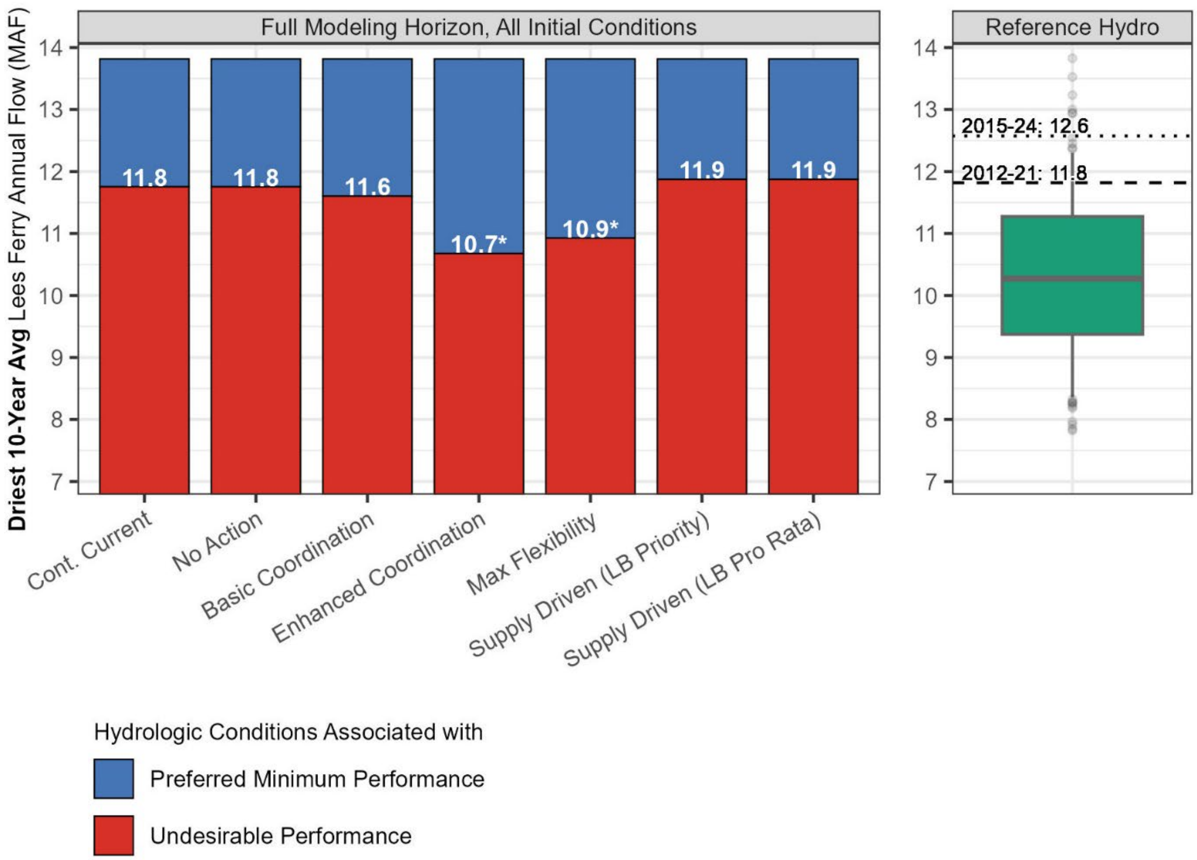


Figure TA 8-22
5-Year Smallmouth Bass Growth Rate: Vulnerability.
Conditions that could cause the 5-year smallmouth bass population growth rate to be at least 1 in one or more years



Note: Results of vulnerability analyses are based on dividing data into two categories and identifying hydrologic conditions that skillfully predict which category a future belongs in. The primary calculation is based on ensuring a high level of skill in predicting futures that meet preferred minimum performance (blue portion) and then confirming that the predictions of futures belonging in the undesirable performance category (red portion) are also skillful. The Enhanced Coordination and Maximum Operational Flexibility Alternatives have lower skill in predicting the undesirable performance category because they are inherently adaptive and can sometimes achieve minimum preferred performance in drier conditions than their thresholds indicate, but not at the frequency required to have high confidence in placing those futures in the blue category. While there is the same level of confidence for all alternatives in predicting when they will meet preferred minimum performance, the Enhanced Coordination and Maximum Operational Flexibility Alternatives have a significantly better chance of achieving preferred minimum performance below their respective thresholds than the other alternatives.

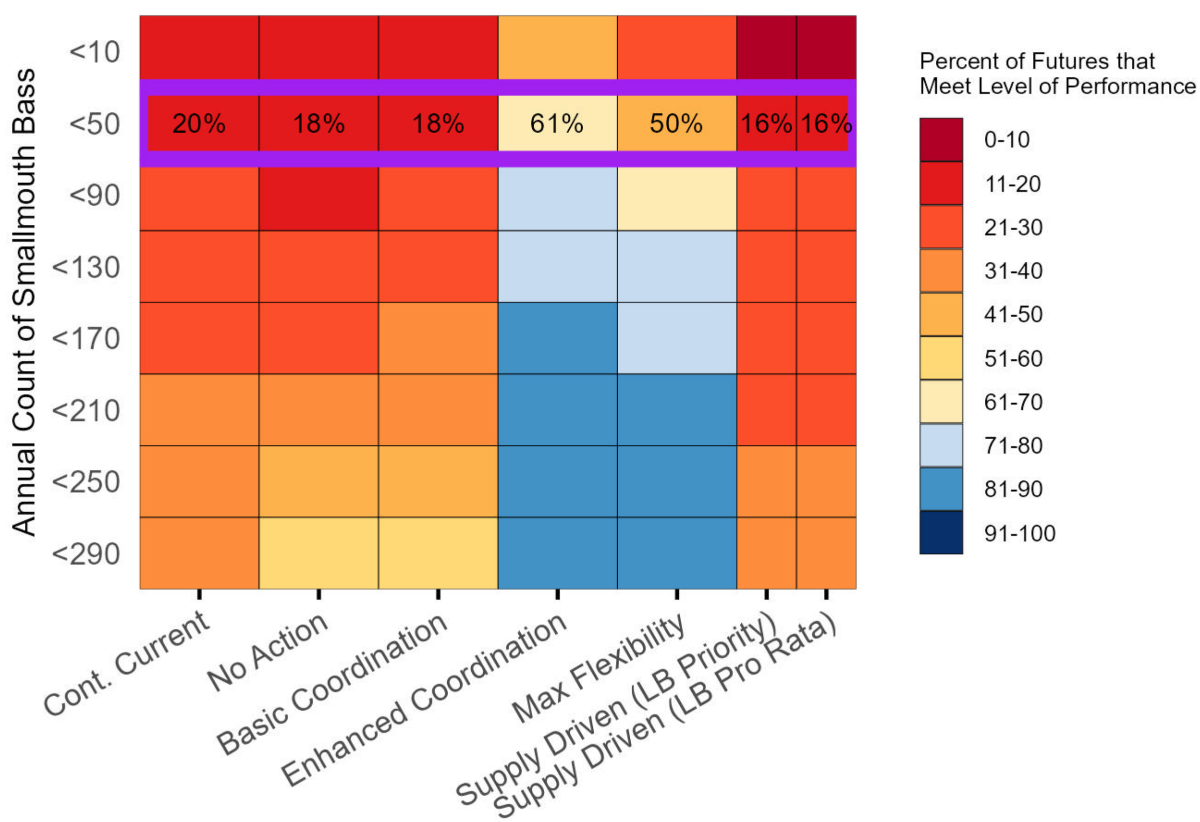
The inputs of flow and water temperature into the smallmouth bass model drive the lambda results. Based on the analysis of water temperature at Lees Ferry (see **Figure TA 8-9**), the Enhanced Coordination and Maximum Operational Flexibility Alternatives are predicted to have fewer days exceeding minimum temperature thresholds (i.e., lower water temperatures), which would more likely decrease lambda relative to the other alternatives.

Smallmouth Bass Entrainment

Entrainment of smallmouth bass through Glen Canyon Dam poses a threat to the native fish community, including humpback chub downstream. Data from the smallmouth bass model assesses entrainment of adult smallmouth bass through Glen Canyon Dam. We use the annual elevation of Lake Powell to calculate potential for entrainment and use a threshold of less than 50 adult individuals as the threshold for assessment, as described in the methodology.

The 5-year product of smallmouth bass entrainment and survival through Glen Canyon Dam is used to assess performance with respect to the threat of smallmouth bass population growth in the Grand Canyon. **Figure TA 8-23** compares alternatives with respect to the annual abundance estimate of adult smallmouth bass. The comparison described here focuses on the preference to stay below a maximum value of 50 adults, in the purple highlighted row, because greater than or equal to 50 adult smallmouth bass per year could represent more than a doubling of the population of reproductively capable fish based on 2022 and 2023 abundance estimates at the beginning of the invasion. Higher rows capture a lower number of adults allowed through entrainment and are harder to achieve, while lower rows represent higher values that are easier to achieve.

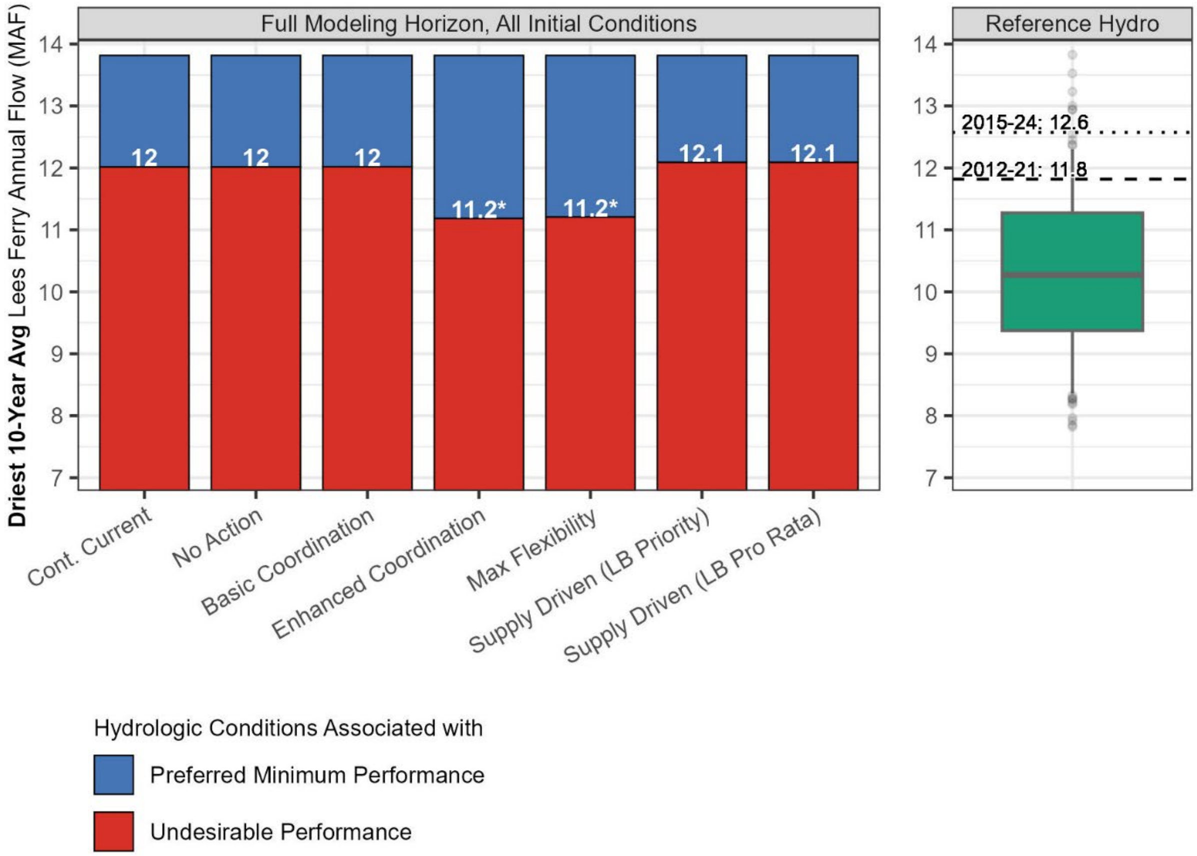
Figure TA 8-23
Smallmouth Bass Entrainment and Survival: Robustness.
Percent of futures in which the annual count of adult smallmouth bass that are entrained and survive is always less than amount specified by each row



Over the full modeling period, the Enhanced Coordination and Maximum Operational Flexibility Alternatives are the most robust, meeting the preferred level of performance in 61 percent and 50 percent of futures, respectively **Figure TA 8-23**. CCS Comparative Baseline and the No Action, Basic Coordination, and Supply Driven (both LB Priority and LB Pro Rata approaches) Alternatives perform similarly between 16 percent and 20 percent. The alternatives perform similarly in higher annual counts (easier, lower row) and slightly lower annual counts (harder, higher row). Results indicate that lower Lake Powell elevations increase the probability of entrainment of smallmouth bass because the warmer water where smallmouth bass reside more closely aligns with the dam's penstocks at elevations nearing 3,490 feet.

Figure TA 8-23 highlights the preferred minimum performance level of annual smallmouth bass entrainment remaining below 50 adults. To identify hydrologic conditions that could cause undesirable performance, the incidence of 50 or more adults being entrained 1 year out of the 34-year modeling period was analyzed. The driest 10-year average Lees Ferry natural flow was identified as a good predictor of undesirable performance. The reference hydrology panel (**Figure TA 8-24**) shows the range of driest 10-year average flows represented in a reference hydrology, along with the most recent and driest observed 10-year periods. If the future includes a 10-year average flow of 12.0 maf or lower, the CCS Comparative Baseline and the No Action and Basic Coordination Alternatives are likely to result in undesirable performance; greater than 75 percent of the reference hydrology traces include averages this low or lower. If the future includes a 10-year average flow of 11.2 maf or lower, the Enhanced Coordination and Maximum Operational Flexibility Alternatives are likely to result in undesirable performance; about 75 percent of the reference hydrology traces include averages this low or lower. If the future includes a 10-year average flow of 12.1 maf or lower, the Supply Driven Alternative (both LB Priority and LB Pro Rata approaches) is likely to result in undesirable performance; greater than 75 percent of the reference hydrology traces include averages this low or lower. The driest 10-year period in the observed record had an average of about 11.8 maf; CCS Comparative Baseline and the No Action, Basic Coordination, and Supply Driven (both LB Priority and LB Pro Rata approaches) Alternatives are all vulnerable to conditions that are close to what has already occurred.

Figure TA 8-24
Smallmouth Bass Entrainment and Survival: Vulnerability.
Conditions that could cause the count of smallmouth bass entrained and surviving in any year to be 50 individuals or more



Note: Results of vulnerability analyses are based on dividing data into two categories and identifying hydrologic conditions that skillfully predict which category a future belongs in. The primary calculation is based on ensuring a high level of skill in predicting futures that meet preferred minimum performance (blue portion) and then confirming that the predictions of futures belonging in the undesirable performance category (red portion) are also skillful. The Enhanced Coordination and Maximum Operational Flexibility Alternatives have lower skill in predicting the undesirable performance category because they are inherently adaptive and can sometimes achieve minimum preferred performance in drier conditions than their thresholds indicate, but not at the frequency required to have high confidence in placing those futures in the blue category. While there is the same level of confidence for all alternatives in predicting when they will meet preferred minimum performance, the Enhanced Coordination and Maximum Operational Flexibility Alternatives have a significantly better chance of achieving preferred minimum performance below their respective thresholds than the other alternatives.

TA 8.2.4 Issue 3: Lower reservoir elevations have reduced lake habitat and extended river inflows, including about 30 miles on the Colorado River and 20 miles on the Virgin River, with similar extensions observed at Las Vegas Bay, Echo Bay, and other locations in Lake Mead. How will these inflow changes affect the quantity and quality of Lake Mead, wetland, and riverine inflow habitats?

The Lower Basin from Lake Mead to the SIB has experienced ecological changes driven by fluctuations in reservoir elevations and tributary inflows. These changes affect the quantity and quality of aquatic habitats in both lake and inflow areas, with consequences for native, nonnative, and recreationally important fish species. Lower lake elevations reduce shoreline habitat for sportfish and razorback sucker, while simultaneously extending riverine inflow habitats on the Colorado River, Virgin River, and other tributaries. Such shifts may alter designated critical habitat for razorback sucker and influence the distribution of native species, including humpback chub and razorback sucker. Spawning and nursery habitats for razorback sucker may change with increases in elevation of Lake Mead and could become limited. Pearce Ferry Rapid functions as a natural barrier to fish movement (native and nonnative), but at elevations above 1,090 feet, the rapid no longer serves as a barrier, while elevations below 1,000 feet compromise hydropower generation.

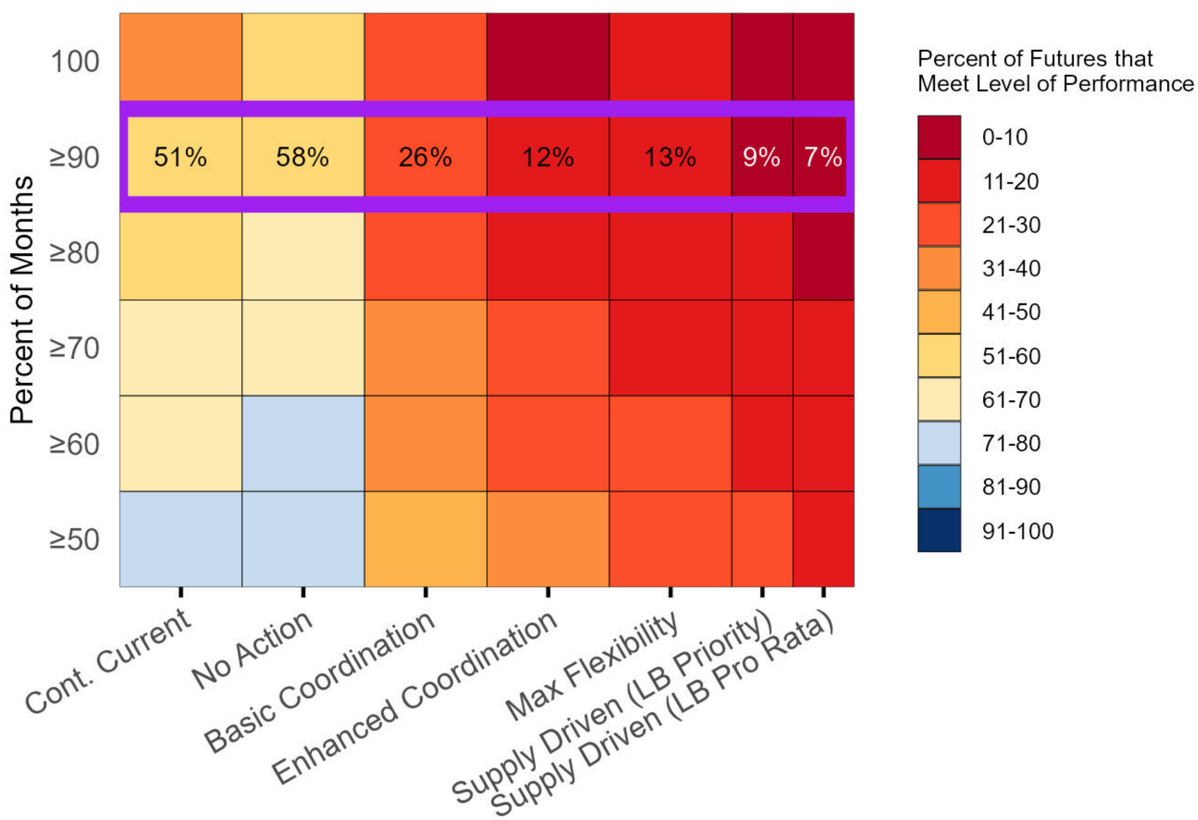
To evaluate these changes, indicators include monitoring data on sportfish catch, shifts in the distribution and abundance of native fish species, and the extent and quality of inflow and shoreline spawning habitats. Bathymetric analyses or lake elevation data provide critical insight into thresholds where razorback sucker spawning habitats become desiccated or reduced. Thresholds also include lake elevations outside the historical Interim Guidelines period (2008–2024) (FWS 2007), which represent untested conditions for fish communities within Lake Mead. Assessment methods integrate monitoring data from AZGFD, NDOW, and the LCR MSCP, as well as DMDU analyses of lake elevation scenarios to evaluate differences in aquatic habitat and fish community response under alternative management strategies.

Thresholds used in the analysis include: (1) Lake Mead elevation above 1,090 feet, at which Pearce Ferry Rapid becomes inundated and no longer functions as a barrier to upstream fish passage, and (2) lake elevations at which razorback sucker spawning locations in the lake are desiccated or, more broadly, when elevations and habitats fall outside the historical range of lake elevations observed during 2015–2025. These thresholds are biologically significant because they may influence the distribution, survival, and reproduction of both native fish populations and sportfish in Lake Mead and represent conditions not previously experienced in recent decades. In all cases, an important assumption is that fisheries management practices, such as stocking of listed fish species, would generally continue as they have during 2008–present.

We analyzed how each alternative affects the likelihood of sustaining Pearce Ferry Rapid as a natural deterrent to upstream movement. Based on current conditions and historic fish captures relative to lake elevation it is thought that if Lake Mead remains below approximately 1,090 feet and follows the historical levels observed during the interim guideline period, the current species composition and population dynamics within the Grand Canyon and Lake Mead would remain similar. The analysis sheds light on upstream movement by native species, such as razorback sucker, while also allowing for upstream movement of nonnative fishes into the Grand Canyon.

Figure TA 8-25 presents a comparison of alternatives based on the number of months in which Lake Mead elevations fall below 1,090 feet. The analysis emphasizes the scenario where elevations remain below 1,090 feet for at least 90 percent of the months, as highlighted in the purple row. This threshold is significant because it helps contain nonnative fish species under most conditions and modeling traces during the analysis period (2015–2025), thereby limiting the Grand Canyon's exposure to additional nonnative fish pressure. The importance of the 90 percent threshold is reinforced by recent data: over the past ten years (January 2015 to May 2025), Lake Mead elevations have been below 1,090 feet for 94.4 percent of the months. This ten-year period was selected for analysis as it more accurately reflects current conditions at Pearce Ferry Rapid. Higher rows capture a greater percentage of time and are harder to achieve, while lower rows represent lower time values that are easier to achieve.

Figure TA 8-25
Pearce Ferry Rapids as Non-Native Fish Barrier: Robustness.
Percent of futures in which Lake Mead elevation stays below 1,090 feet in the percent of months specified by each row



Note: Supply Driven LB Priority and Supply Driven LB Pro Rata results differ primarily because of how the two shortage-distribution approaches interact with the modeled assumptions governing the storage and delivery of conserved water (see **Appendix B**, Modeling Assumptions: Lake Powell and Lake Mead Storage and Delivery of Conserved Water).

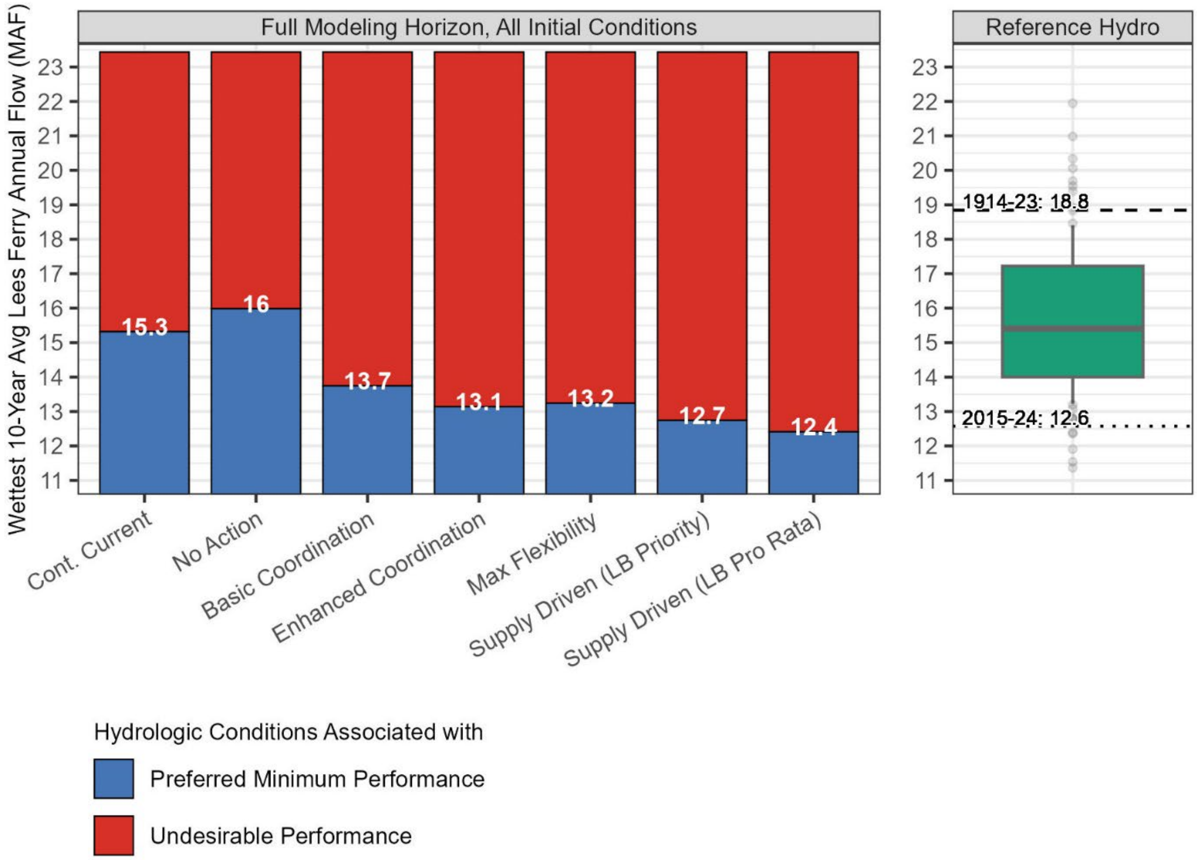
Over the full modeling period, the No Action Alternative is the most robust, meeting the preferred level of performance in 58 percent of futures. CCS Comparative Baseline performs similarly at 51 percent. Since a greater than 10 percent difference would be considered significant, all other alternatives are greater than 10 percent lower than the No Action Alternative or CCS Comparative Baseline and considered less robust. Of the action alternatives, Basic Coordination Alternative performs the best with 26 percent of futures maintaining Lake Mead elevations greater than or equal to 90 percent of months. The Enhanced Coordination, Maximum Operational Flexibility, and Supply Driven (both LB Priority and LB Pro Rata approaches) Alternatives are the least robust between 7 and 12 percent (**Figure TA 8-25**).

Figure TA 8-26 shows the conditions that could cause Lake Mead elevation to exceed 1,090 feet in more than 10 percent of months. When the elevation of the lake is above 1,090 feet, fish can move above the rapid from Lake Mead into the Grand Canyon. This would allow the native fish to access historical riverine habitat that could benefit their populations. But these same conditions would also allow nonnative fish to access the river as potential predators and competitors of native fish. If future conditions include a 10-year average flow of 16.0 maf or higher, the CCS Comparative Baseline is likely to perform undesirably; less than 50 percent of reference hydrology traces are above this threshold (**Figure TA 8-26**). Similarly, the Basic Coordination Alternative is vulnerable if the 10-year average flow exceeds 13.7 maf, with over 75 percent of traces meeting this criterion. Similar results are noted for the remaining alternatives with the Supply Driven Alternative performing the worst. The wettest 10-year period in the observed record averaged 18.8 maf (1914-2023), indicating that all alternatives are susceptible to conditions that were experienced in the early 1900s. However, all alternatives, except the Supply Driven Alternative (LB Pro Rata) would be expected to satisfy the preferred minimum performance if the wettest 10-year average is no wetter than the most recent 10-year period of 12.6 maf (2015-2024).

Alternatives that allow for more flow and Lake Mead storage perform worse than the No Action Alternative and CCS Comparative Baseline for native fishes such as razorback sucker, flannelmouth sucker, or even humpback chub that may wish to move from downstream to upstream and into the Grand Canyon. Thus, under a less robust scenario, native fish movement upstream out of Lake Mead would occur more frequently, which could allow for additional spawning and genetic mixing within the Colorado River in the Grand Canyon but would also allow for more competition from nonnative fish moving into the western Grand Canyon, where native fish abundance currently dominates the fish community.

To explore shoreline habitat availability in Lake Mead, we used the 2008–2024 lake elevation data as a proxy to determine if there would be suitable littoral shoreline habitats for the fish community in Lake Mead. An important assumption is that at any given lake elevation, the amount of littoral shoreline habitat available for fishes and other aquatic species is related to elevations and shoreline habitats present from 2008 to 2024. The lowest elevation observed in Lake Mead from 2008 to 2024 was 1,040.92 feet in July 2022. Because it appears that the current fish and aquatic species assemblage within Lake Mead survived at that lake elevation, we presume the shoreline habitat at that elevation is suitable for the overall current fish community.

Figure TA 8-26
Pearce Ferry Rapid remains a barrier to upstream fish passage: Vulnerability. Conditions that could cause Lake Mead elevation to go above 1,090 feet in more than 10 percent of months

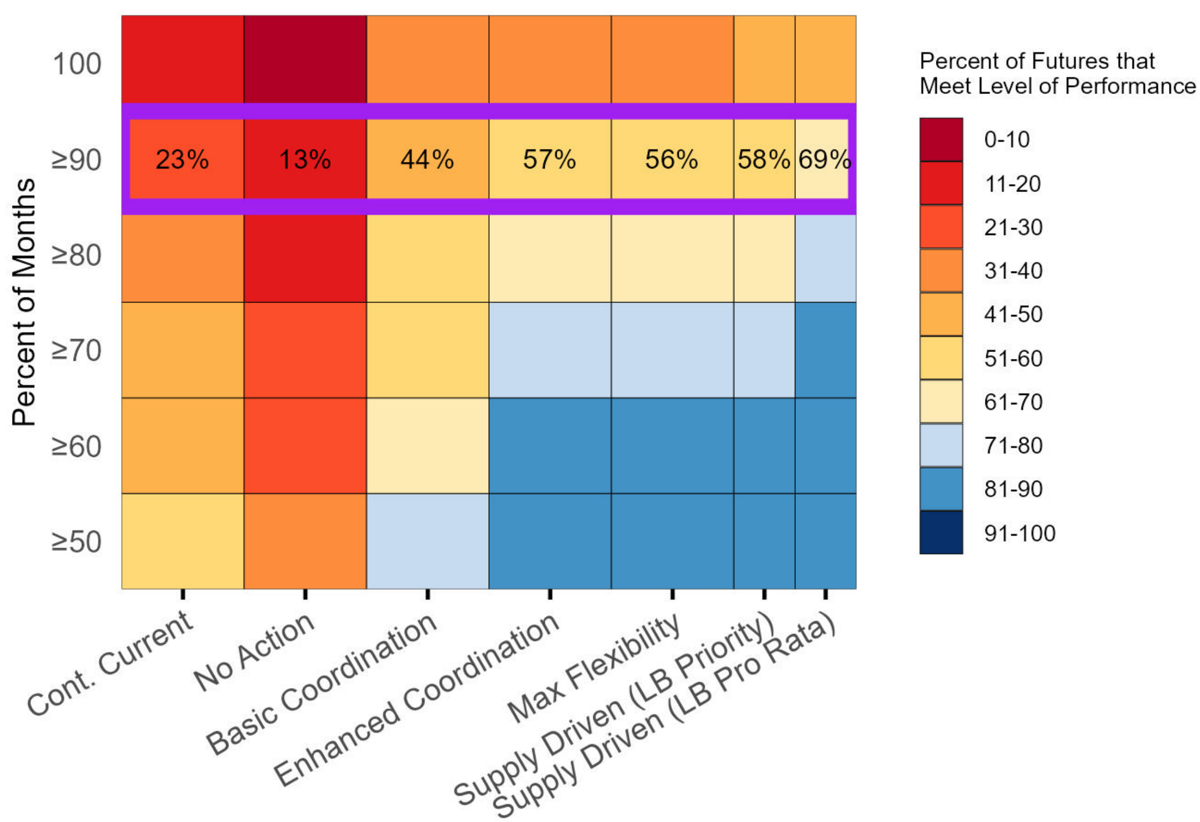


The robustness analysis illustrates how well different alternatives sustain Lake Mead's elevation above the historical minimum of 1,040.92 feet across a wide range of future conditions. **Figure TA 8-27** shows the percent of modeled futures for each alternatives in which Lake Mead elevation remains above the historical minimum elevation (1,040.92 feet) in the percent of months specified by each row. The "greater than or equal to 90" threshold was selected because, under the 'Low' initial condition scenario, Lake Mead is projected to be at 1,038 feet on January 1, 2027. If the 100 percent row were used, one-third of the modeled futures would automatically fail to meet the performance criteria. By using the greater than or equal to 90 threshold, the analysis reflects scenarios in which Lake Mead's elevation remains at or above 1,040.92 feet for at least 90 percent of the months throughout the full modeling period (2027–2060) (**Figure TA 8-27**).

The No Action Alternative model shows only 13 percent of the simulated future models (futures) maintain lake elevations at/or above the 1,040.92 feet minimum for at least 90 percent of months. CCS Comparative Baseline increases this to 23 percent of futures, while the Basic Coordination Alternative increases it further to 44 percent of modeled futures (**Figure TA 8-27**). The Enhanced

Coordination Alternative (57 percent), the Maximum Operational Flexibility Alternative (56 percent), the Supply Driven Alternative (LB Priority approach [58 percent]), and the Supply Driven Alternative (LB Pro Rata approach [69 percent]) all demonstrate similar percentages of futures, respectively (**Figure TA 8-27**). Among these, the Supply Driven Alternative (LB Pro Rata approach) is most robust (69 percent of futures) and most likely to maintain Lake Mead elevations at/or above 1,040.92 feet.

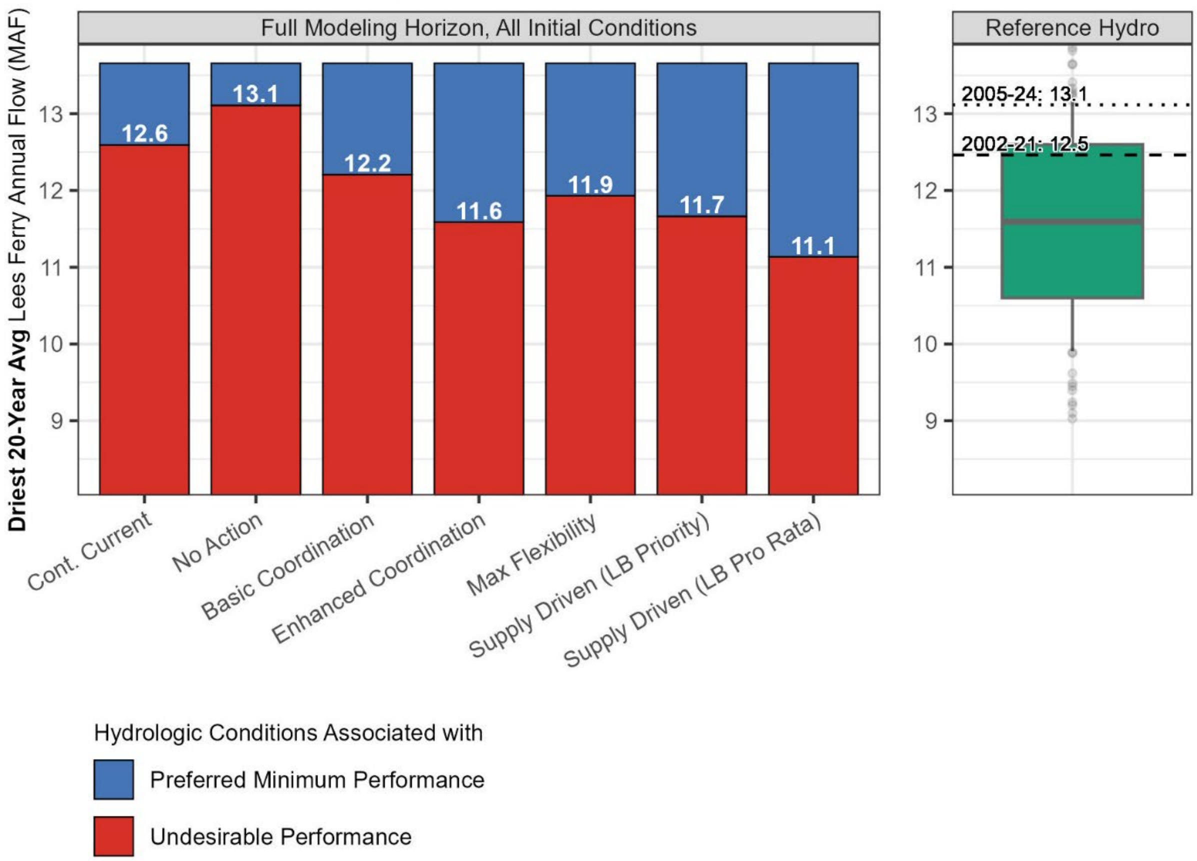
Figure TA 8-27
Lake Mead Historical Minimum Shoreline Habitat: Robustness.
Percent of futures in which Lake Mead elevation remains above the historical minimum elevation (1,040.92 feet) in the percent of months specified by each row



Note: Supply Driven LB Priority and Supply Driven LB Pro Rata results differ primarily because of how the two shortage-distribution approaches interact with the modeled assumptions governing the storage and delivery of conserved water (see **Appendix B**, Modeling Assumptions: Lake Powell and Lake Mead Storage and Delivery of Conserved Water).

Figure TA 8-28 shows the streamflow conditions in which each alternative is likely to fail the preferred minimum performance level, meaning Lake Mead elevation falling below the historical minimum observed during 2008–2024 (1,040.92 feet) in more than 10 percent of months during the full modeling horizon (2027–2060). The driest 20-year average streamflow in each future, plotted on the vertical axis, was the most skillful streamflow statistic at identifying if a future will satisfy or fail the preferred minimum performance level. The Supply Driven Alternative (LB Pro Rata) is the least vulnerable, being vulnerable to futures that experience a 20-year average of 11.1 maf or drier, which is experienced in less than 50 percent of reference hydrology traces. The Enhanced Coordination, Supply Driven (LB Priority), and Maximum Operational Flexibility Alternatives are vulnerable to similar conditions (11.6, 11.7, and 11.9 maf, respectively), which corresponds to about 50 percent or more traces in the reference hydrology. All action alternatives are expected to meet the preferred minimum performance if the future is no more severe than the worst 20-year average on record (12.5 maf, observed 2002-2021), while this drought is expected to result in undesirable performance for the No Action Alternative and the CCS Comparative Baseline.

Figure TA 8-28
Lake Mead Historical Minimum Shoreline Habitat: Vulnerability.
Conditions that could cause Lake Mead to fall below 1040.92 feet in more than 10% of months



Note: Supply Driven LB Priority and Supply Driven LB Pro Rata results differ primarily because of how the two shortage-distribution approaches interact with the modeled assumptions governing the storage and delivery of conserved water (see **Appendix B**, Modeling Assumptions: Lake Powell and Lake Mead Storage and Delivery of Conserved Water).

The fish community in Lake Mead has adapted to fluctuating lake elevations for decades (Rogers et al. 2025). If lake elevations remain within the historical range observed from 2008–2024, the conditions that support current species composition and population dynamics are likely to persist. If the Lake Mead elevation remains relatively stable and within the historical elevations, the aquatic community would likely continue to find spawning and/or nursery habitats. However, prolonged low-elevation periods may increase vulnerability of shoreline habitats that aquatic species utilize by reducing spawning and/or nursery habitat beyond what has been routinely experienced by those species within this system from 2008 to 2024.

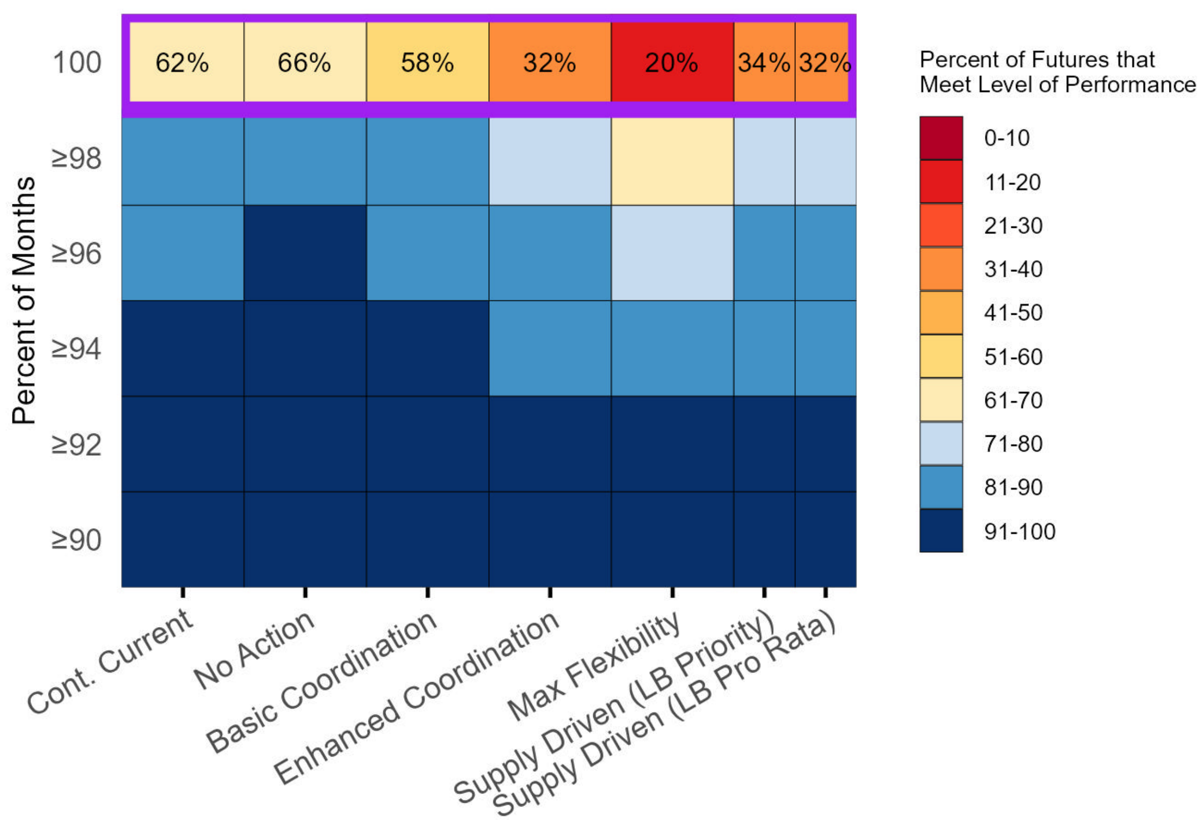
TA 8.2.5 Issue 4: How will modifications in water quantity, timing, temperature, and quality released from Hoover Dam affect the Lower Colorado River, including LCR MSCP Reaches 2 through 6 (Hoover Dam to Davis Dam and Lake Mohave; Davis Dam to Parker Dam and Lake Havasu; Parker Dam to Cibola Gage; Cibola Gage to Imperial Dam and Imperial Reservoir; and Imperial Dam to the Northerly International Boundary)?

If the alternatives remain within the historical levels of flow releases from Hoover Dam, the current fish species composition and population dynamics will likely remain unchanged. However, modified flows from Hoover Dam to Imperial Dam may affect the habitats and abundances of razorback sucker and bonytail, as well as backwaters and riverine habitats that support these endangered and other native fishes. Flow modifications may also impact known razorback sucker spawning sites, while flannelmouth sucker downstream of Davis Dam to Parker Dam could similarly be affected. In addition, desert pupfish inhabiting off-channel ponds in the Lower Colorado River may experience impacts from modified flows that could fall outside of historical 2008–2024 bounds, particularly if flooding of constructed refuge ponds occurred. Furthermore, the presence of flathead catfish downstream of Cibola gage to Imperial Dam poses an additional concern, as reduced flows may allow this predator to gain access to upper reaches, increasing risks to endangered and other native fish populations. Impact indicators include fish habitats, distributions, and the assumption that populations are influenced by dam releases and flow conditions, which also affect the extent and availability of backwater areas. In addition, the location, size, and overall suitability of razorback sucker spawning sites could vary with different flow regimes, highlighting the importance of flow management for maintaining critical life history habitats. Assessment methods include reports and monitoring data from agencies, including AZGFD, NDOW, the California Department of Fish and Wildlife, Reclamation, LCR MSCP, and the FWS. In all cases, an important assumption is that fisheries management practices, such as stocking of listed fish species, would generally continue as they have during 2008–present.

Figure TA 8-29 illustrates how different alternatives would affect monthly releases downstream of Hoover Dam comparing the percent of modeled futures for each alternative in which monthly releases from Hoover Dam are within the range observed during the Interim Guidelines period (2008–2024) in the percent of months specified by each row. The highlighted row (100 percent) indicates the maximum and preferred performance level, while lower rows capture a lower percentage of time and are therefore more likely to be achieved. Alternatives that satisfy the preferred performance level in a high percentage of futures are better at protecting critical habitat in the Colorado River inflow for Colorado pikeminnow and razorback sucker, whereas alternatives that satisfy the preferred minimum preferred performance in a low percentage of futures are worse.

The Maximum Operational Flexibility Alternative shows that only 20 percent of the simulated futures would maintain releases downstream of Hoover Dam 100 percent of months. The Enhanced Coordination Alternative increases favorable futures to 32 percent during 100 percent of the months (**Figure TA 8-29**). The Supply Driven Alternative (LB Pro Rata approach [32 percent]) and Supply Driven Alternative (LB Priority approach [34 percent]) show somewhat comparable results of modeled futures during 100 percent of the months (**Figure TA 8-29**). While the Basic Coordination Alternative (58 percent), CCS Comparative Baseline (62 percent), and the No Action Alternative (64 percent) of the futures are performing similarly during 100 percent of the months (**Figure TA 8-29**). It is important to consider that this robustness figure suggests that all alternative futures should meet the historical range of flows observed from 2008–2024, during more than 92 percent of the months under all modeled futures (**Figure TA 8-29**).

Figure TA 8-29
Below Hoover Monthly Flows: Robustness.
Percent of futures in which monthly releases from Hoover Dam are within the range observed during the Interim Guidelines period in the percent of months specified by each row

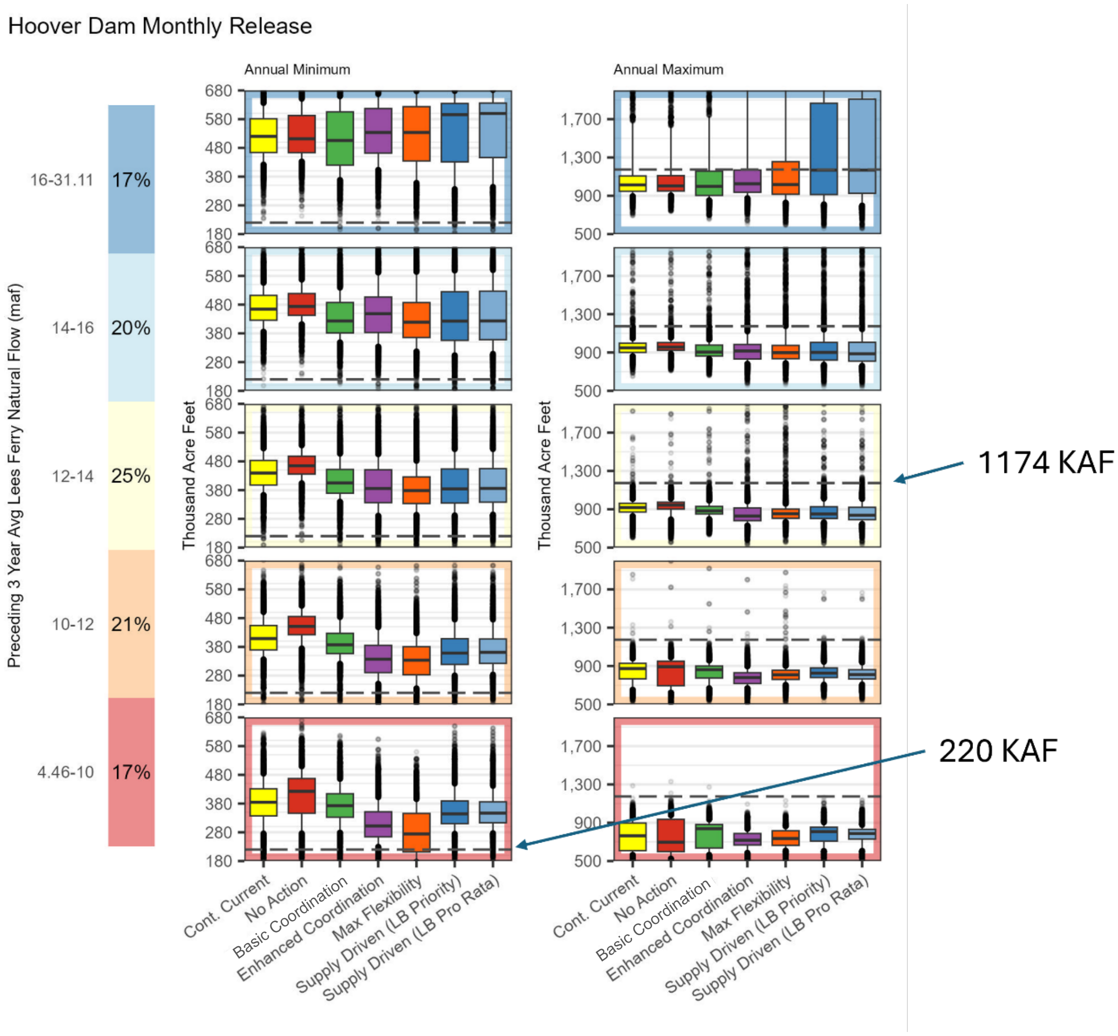


Note: Supply Driven LB Priority and Supply Driven LB Pro Rata results differ primarily because of how the two shortage-distribution approaches interact with the modeled assumptions governing the storage and delivery of conserved water (see **Appendix B**, Modeling Assumptions: Lake Powell and Lake Mead Storage and Delivery of Conserved Water).

In **Figure TA 8-30**, the boxplots illustrate how each alternative responds to these hydrologic scenarios, especially under challenging conditions like drought, enabling direct comparison of the aquatic habitats in the study area. The figure shows monthly release patterns from Hoover Dam under the various alternative futures, derived from calendar years 2008–2024. The x-axis shows the alternative futures, while the y-axis shows the monthly release volume in thousand acre-feet (kaf). The left column presents the annual minimum releases, and the right column shows the annual maximum releases (**Figure TA 8-30**). Each row corresponds to a range of preceding 3-year average inflows at Lees Ferry, from wet years at the top (16.00–31.00 maf) to dry years at the bottom (4.46–10.00 maf) (**Figure TA 8-30**). The colored boxplots depict the range of flows by alternative futures, with longer boxes showing a larger variation of flows. Dashed lines are the minimum (left column) and maximum (right column) monthly releases observed from calendar years 2008–2024. In the column on the left, it is considered a "good outcome" if the boxplots are above the dotted line (220 kaf) (**Figure TA 8-30**). In the column on the right, it is considered a "good outcome" if the boxplots are below the dotted line (1,174 kaf) (**Figure TA 8-30**). During wet conditions, most alternatives maintain comparable release ranges, but as inflows decline, the differences in management adaptability become clear. The No Action Alternative and CCS Comparative Baseline sustain high release volumes even during droughts, which may increase the rate at which Lake Mead elevation declines. In contrast, the Supply Driven Alternative (LB Pro Rata approach) shows reduced but stable releases under dry conditions (**Figure TA 8-30**). This alternative may help to maintain Lake Mead elevations within historical ranges (at/or above 1,040.92 feet, the lowest elevation Lake Mead has reached) as well as maintain the habitats in the LCR MSCP reaches downstream of Hoover Dam. **Figure TA 8-29** and **Figure TA 8-30** cumulatively appear to suggest that the LCR MSCP reaches will likely remain relatively similar to the 2008–2024 time period.

Within the LCR MSCP, the highest priority of the habitat conservation plan is the protection and preservation of backwater, isolated ponds, refuge (predator-free), and marsh habitats, especially with respect to areas occupied by ESA-listed species (LCR MSCP 2024). These areas are vital for maintaining the ecological integrity and biodiversity of the Lower Basin (Holmquist-Johnson et al. 2016; LCR MSCP 2024) as well as allowing ESA-listed species to complete their life cycles. Because downstream habitats are largely re-regulated by Hoover Dam releases, changes in aquatic habitat are reflected by the patterns of Hoover Dam releases. Further, elevations and flows at Lake Mohave and Lake Havasu follow a guide curve and stay nearly constant across alternatives following releases from Hoover Dam. Multiple scenarios involving downstream impoundments were modeled and evaluated, but no significant differences were found to warrant further analysis for those locations. As a result, this analysis focuses on releases from Hoover Dam as a surrogate for all impoundments and features downstream. As described, modeled results generally suggest that most alternatives downstream of Hoover Dam will result in relatively similar futures for fishes (in terms of modeled flows), much like what has been experienced from 2008–2024, (assuming ongoing and similar fisheries and aquatic management practices remain in place and continue). Due to these factors, it is likely that habitats in the LCR MSCP reaches will remain relatively similar to historical conditions, and hence the fish communities will likely remain similar to the 2008–2024 time period.

Figure TA 8-30
Hoover Dam Monthly Releases Under Varying 3-Year Average Lees Ferry Inflows with Alternative Futures. Boxplots Show the Distribution of Annual Minimum (Left) and Annual Maximum (Right) Monthly Releases Across Simulated Futures, Thousand Acre Feet



Note: Supply Driven LB Priority and Supply Driven LB Pro Rata results differ primarily because of how the two shortage-distribution approaches interact with the modeled assumptions governing the storage and delivery of conserved water (see **Appendix B**, Modeling Assumptions: Lake Powell and Lake Mead Storage and Delivery of Conserved Water).

TA 8.2.6 Summary Comparison of Alternatives

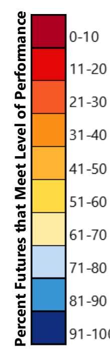
The preceding analysis uses a combination of hydrologic modeling, ecological indicators, temperature thresholds, and robustness/vulnerability assessments to compare management alternatives and their effects on aquatic habitats, native and nonnative fish populations, and ecosystem resilience along the Colorado River from Lake Powell downstream to the international boundaries. A summary matrix of robustness analysis results is provided to depict differences among alternatives across all issues and indicators (**Figure TA 8-31** and summarized in **Section 2.11**).

Issue 1 assesses the impact of lake elevation on several native fish indicators. The greatest threat to native fish would be the establishment of smallmouth bass downstream of Glen Canyon Dam, which becomes more likely with lower lake elevations. Additionally, reservoir elevations below 3,666.5 feet will help to reduce the threat of nonnative fish movement in the San Juan River. Lastly, higher lake elevations would inundate Colorado pikeminnow and razorback sucker critical habitat and reduce habitat value for those species. When considering the risk of smallmouth bass entrainment at Glen Canyon Dam, the Enhanced Coordination and Maximum Operational Flexibility Alternatives perform best by maintaining higher elevations and cooler water, reducing the likelihood of entrainment and downstream invasion. The Supply Driven Alternative (both LB Priority and LB Pro Rata approaches) and CCS Comparative Baseline keep lake elevations low enough to maintain the Piute Farms waterfall as a fish barrier, while the Enhanced Coordination and Maximum Operational Flexibility Alternatives allow for a higher potential of inundation and fish passage. The low Powell elevations allowed by the No Action Alternative increase the length of exposed riverine habitat for Colorado pikeminnow and razorback sucker, benefitting these endangered species, while all other alternatives tend to decrease habitat through inundation with higher lake levels. However, this comes at the cost of reduced lake habitat for sportfish, potentially negatively impacting recreational fishing.

Issue 2 assesses several factors and measures that influence nonnative fish populations. The primary threat to native fish populations below Glen Canyon Dam is the invasion of warmwater nonnative fish species such as smallmouth bass. Colder water temperatures and reduced entrainment through Glen Canyon Dam result from higher lake levels and help to prevent these populations from establishing and expanding. The Enhanced Coordination and Maximum Operational Flexibility Alternatives generally keep Lake Powell higher and consequently are the most robust in maintaining cooler water temperatures at Lees Ferry, the Little Colorado River confluence, Havasu Creek, and Pearce Ferry. The colder water limits smallmouth bass population growth but may also restrict native fish spawning. These two alternatives also keep smallmouth bass population growth rates (lambda) below 1.0 more consistently than all other alternatives. The No Action, Supply Driven (both LB Priority and LB Pro Rata approaches) Alternatives, and CCS Comparative Baseline are less robust, often failing to maintain preferred temperature and entrainment thresholds, especially under drought scenarios. While warmer water temperatures can benefit native fish growth and recruitment, they also favor nonnative species like smallmouth bass, which increases competition and predation risks. Thus, alternatives that limit the number of days above critical temperature thresholds (Enhanced Coordination and Maximum Operational Flexibility Alternatives) are preferable for controlling nonnative fish despite tradeoffs for native fish reproductive success.

Figure TA 8-31
Summary of Robustness Results across Issues by Indicator at Reported Thresholds for Each Alternative

Issue	Indicator	Impact Threshold	Alternative: Continued Current Strategy Baseline	No Action	Basic Coordination	Enhanced Coordination	Maximum Operational Flexibility	Supply-Driven (Priority)	Supply-Driven (Pro Rata)
1	**Lake Powell Elevation**								
	Critical Habitat (Colorado River)	Lake Powell is below 3,598 feet ≥60% of months, and Critical Habitat of Colorado Pikeminnow and Razorback Sucker is not inundated in the Colorado River Inflow							
	Critical Habitat (San Juan River)	Lake Powell is below 3,600 feet ≥60% of months, and Critical Habitat of Colorado Pikeminnow and Razorback Sucker is not inundated in the San Juan River Inflow							
	Piute Falls (nonnative barrier)	Lake Powell is below 3,666.5 feet for 100% of months, and the Paiute Farms Waterfall remains a barrier to upstream fish passage							
	Smallmouth Bass Entrainment	Lake Powell is above 3,570 feet ≥60% of months, and the risk of smallmouth bass entrainment is reduced							
2	**Water Temperature**								
	Lee's Ferry 20	Number of Days Exceeds 20°C at Lees Ferry (0 days)							
	Little Colorado River 12	Number of Days Exceeds 12°C at the Little Colorado River (≥200 days)							
	Little Colorado River 16	Number of Days Exceeds 16°C at the Little Colorado River (≤170 days)							
	Havasu Creek 12	Number of Days Exceeds 12°C at Havasu Creek (≥200 days)							
	Havasu Creek 16	Number of Days Exceeds 16°C at Havasu Creek (≤190 days)							
	Pearce Ferry 12	Number of Days Exceeds 12°C at Pearce Ferry (≥200 days)							
	Pearce Ferry 16	Number of Days Exceeds 16°C at Pearce Ferry (≤190 days)							
	Smallmouth Bass Population	5-year Smallmouth Bass growth rate (lambda) at Lees Ferry is less than 1							
	Smallmouth Bass Entrainment	Smallmouth Bass entrainment through Glen Canyon Dam is less than 50 individuals							
3	**Lake Mead Elevation**								
	Within Historical Bounds	Lake Mead Elevations Remain Within Historical Bounds experienced from 2008–2024, ≥90% of months							
	Pearce Ferry Rapid (nonnative barrier)	Colorado River water levels at Pearce Ferry Rapid are Below 1,090 Feet Above Mean Sea Level, ≥90% of months							
4	**Hoover Dam Releases**	Monthly Releases from Hoover Dam are within the range Observed During 2008–2024, 100% of months							



Other factors such as flow regimes and water quality can also impact the native fish community throughout the Grand Canyon. Dissolved oxygen concentrations downstream of Glen Canyon Dam fluctuate seasonally, largely influenced by reservoir elevation and penstock water quality. Salmonid species, such as trout, are more susceptible to low dissolved oxygen, with optimal concentrations for growth and survival between 6–9 mg/L, and thresholds below 3 mg/L leading to reduced survival and feeding efficiency. Across alternatives, dissolved oxygen levels in wetter years are similar, typically between 7 and 8 mg/L, but decrease and become more variable under drier conditions. Modeling assumptions may overestimate dissolved oxygen during low reservoir elevations for some alternatives, but only the Enhanced Coordination and Maximum Operational Flexibility Alternatives are projected to maintain higher minimum dissolved oxygen concentrations—above 3 and 4 mg/L—in most futures, making them more robust in sustaining adequate dissolved oxygen and protecting fish health during extended droughts and low-flow scenarios.

Issue 3 assesses the alternatives for their impact on Lake Mead and tributary inflow habitats. The No Action Alternative and CCS Comparative Baseline are most robust in maintaining Lake Mead elevations below 1,090 feet for at least 90 percent of months, helping to prevent upstream movement of nonnative fish by maintaining the fish barrier at Pearce Ferry and preserving the existing native species composition in the Grand Canyon. However, the fish barrier also works against upstream movement by native species such as razorback sucker, flannelmouth sucker, or humpback chub (as examples) into habitats within the Grand Canyon. The Basic Coordination Alternative performs moderately well at maintaining the Pearce Ferry fish barrier, while Enhanced Coordination, Maximum Operational Flexibility, and Supply Driven (both LB Priority and LB Pro Rata approaches) Alternatives are the least robust, more frequently allowing lake elevations to rise above critical thresholds and potentially facilitating upstream movement of both native and nonnative fishes. The Supply Driven Alternative (LB Pro Rata approach) is most robust for maintaining Lake Mead above the historical minimum elevation, providing greater habitat stability.

Issue 4 assesses the alternative's effects on water quantity, timing, temperature, and quality released from Hoover Dam into the Lower Colorado River, including MSCP habitat. Most alternatives, including the No Action and Basic Coordination Alternatives and the CCS Comparative Baseline, are robust in maintaining monthly releases within the historical range observed during the Interim Guidelines period (2008–2024); thus, supporting stable fish communities and habitat conditions. Enhanced Coordination, Maximum Operational Flexibility, and Supply Driven (both LB Priority and LB Pro Rata approaches) Alternatives are less robust, with a lower percentage of futures maintaining desirable release patterns. Flow modifications under less robust alternatives may negatively affect backwaters, marshes, and spawning sites for listed species such as razorback sucker and bonytail, as well as increase risks from predators like flathead catfish. However, it should be noted that any threshold below 92 percent of months for all traces results in positive outcomes for native fish species. If fisheries management practices continue as in recent years, all alternatives are expected to result in similar habitat and population dynamics as experienced during the historical period, as compared to modeled futures.

Overall, each alternative presents tradeoffs between supporting native endangered species, controlling nonnative fishes, and maintaining recreational sportfish habitats. The No Action Alternative and CCS Comparative Baseline are more effective at preserving riverine habitats above

Lake Powell and limiting upstream movement of nonnative species. The comparative analysis of alternatives across all issues suggests that the Enhanced Coordination and Maximum Operational Flexibility Alternatives generally perform best in controlling nonnative fish risks and maintaining cooler water temperatures below Glen Canyon Dam.

TA 8.3 Glossary

algal bloom	An excessive growth of algae in a waterbody, typically associated with nutrient enrichment.
Analysis Area	Specific geographic boundary and timeframe used to evaluate the potential environmental effects of a proposed action.
anoxic	Describes water conditions in which dissolved oxygen is absent.
critical habitat	A specific geographic area that contains physical or biological features essential to the conservation of a listed species and may require special management or protection.
backwater	A portion of a river or stream where flow is reduced or temporarily stagnant due to downstream obstructions or high-water conditions.
bathymetrically	Pertaining to the measurement, mapping, or description of underwater topography.
benthic	Referring to the bottom (bed) of a waterbody.
broodstock	Sexually mature individuals maintained in captivity or managed populations for the purpose of producing offspring.
bug flows	Managed dam releases intended to enhance aquatic invertebrate production and diversity by mimicking natural flow patterns.
channel migration	The lateral movement of a river or stream channel across its floodplain over time.
Colorado River Simulation System (CRSS)	A long-term planning and forecasting model developed and maintained by the Bureau of Reclamation to support operational decision-making for the Colorado River Basin and used to generate multi-decadal (5–50 year) probabilistic projections of hydrologic conditions, reservoir levels (including Lake Powell and Lake Mead), and system outcomes under varying supply, demand, and policy scenarios.
conservation species	A species that has been identified by a management agency, conservation organization, or research program as needing special attention or protection
debris fan	A fan-shaped deposit of sediment, rock, and other debris formed when high-velocity flows such as debris flows or flash floods exit a narrow conduit onto a flatter plain.
debris flow	A fast-moving mass of water, soil, rock, and organic debris that flows downslope under the influence of gravity

Decision Making Under Deep Uncertainty (DMDU)	A method for making decisions using a framework that evaluates several solutions across multiple future scenarios.
desiccation	The process of drying out or losing moisture due to exposure to air, heat, or lack of water.
dissolved oxygen	The concentration of oxygen gas dissolved in water.
downcutting	The process by which a stream or river erodes downward into its bed, deepening the channel through the removal of sediment and bedrock.
electrofishing	A method of sampling fish populations by applying a controlled electric current throughout the water to temporarily immobilize fish for capture, identification, and release.
endemic	Native to and confined within a particular geographic area or habitat type.
entrained	Drawn into and carried along by the movement of water.
ephemeral	Describing a stream that conveys flow only during and shortly after a precipitation event.
epilimnion	The warm, upper layer of water in a thermally stratified lake or reservoir that is influenced by wind and surface currents.
escapement	Fish moving and surviving through Glen Canyon Dam and into the Grand Canyon.
euryhaline	Describes organisms capable of surviving and functioning across a broad range of salinities.
federally listed species	A species that has been officially designated as endangered or threatened.
fish barrier	Any natural or artificial structure that prevents or limits the upstream or downstream movement of fish.
forage fish	Small, schooling fish species that serve as prey for larger fish.
full-pool elevation	The designated water surface elevation at which a reservoir is considered full under normal operating conditions.
grow-out	The stage in aquaculture or fish culture during which young fish are raised to a larger or more mature size until they are ready for release.
Habitat Conservation Plan (HCP)	A planning document describing general and species-specific conservation measures for species listed under the Endangered Species Act.
hepatotoxins	Toxins that damage or interfere with liver functions.
hypolimnion	The cooler, lower layer of water in a thermally stratified lake or reservoir that is relatively stagnant.
hydropower production	The generation of electricity from the movement of water.

impoundments	A body of water formed by the collection and storage of surface runoff or streamflow behind a dam.
inflow	An area where water flows into a reservoir, lake, or stream channel from upstream sources.
inundated	Covered or submerged by water.
littoral	The shallow, nearshore zone of a lake or reservoir.
Lower Colorado River Multi-Species Conservation Program (LCR MSCP)	A comprehensive program for the protection of 27 species and their habitat in the Lower Colorado River Basin
listed	A species that has been designated as threatened or endangered under the Endangered Species Act (ESA).
mainstem	The primary channel of a river system carrying the largest and most continuous flow.
mesolimnion	The middle layer in a stratified lake or reservoir.
monomictic	Describes a lake or reservoir that undergoes complete mixing once a year, usually because of seasonal changes in air and water temperature.
neurotoxins	Toxic substances that interfere with the normal function of the nervous system.
pelagic	Relating to the open-water zone of a lake or reservoir, away from the shore and bottom.
penstock	A pipe that conveys water under pressure from a reservoir or intake into hydraulic equipment within a hydroelectric facility.
PIT scanning	The process of detecting and recording Passive Integrated Transponder (PIT) tags implanted in fish using an electronic reader.
propagule	Various forms of reproductive material (e.g., smallmouth bass larvae)
rearing	The period in which juvenile fish grow and develop before migrating or recruiting into older life stages
recruitment	The addition of new individuals to a fish population.
refuge ponds	Off-channel or low-flow ponded areas that provide shelter for fish during unfavorable conditions such as high temperatures or low flows.
remote sensing	The collection and interpretation of information about the Earth's surface or atmosphere without direct physical contact.
repatriated	Refers to the intentional reintroduction of a species into an area of its historical range from which it has been locally extinct.
resilience	The capacity of a fish population to absorb disturbance, maintain essential functions, and recover to a stable condition within a reasonable amount of time.

riffle	A shallow, fast-moving section of a stream or river where water flows turbulently over coarse substrate, increasing aeration.
riverine	Relating to or characteristic of a river or stream.
robustness	The degree to which a model or conclusion remains valid and reliable under different assumptions, datasets, or analytical conditions.
sonic-tagged	Describes a fish that has been implanted with or attached to an acoustic sounder that emits sound pulses and can be detected by receivers to track movement and habitat use.
stocked	Describes fish that have been intentionally introduced into a waterbody.
stratify	To form distinct layers within a waterbody based on differences in temperature, quality, or oxygen levels.
state-regulated sportfish	Fish species managed by a state natural resource or wildlife agency for recreational fishing.
talus	An accumulation of loose, angular rock fragments that collect at the base of cliffs or steep valley walls.
tailwater	The water downstream of a dam, spillway, or other hydraulic structure that is directly affected by releases.
trammel-netting	A fish sampling technique that uses a three-layered gill net designed to capture fish of various sizes
varial zone	The zone along the margins of a river or lake that alternates between submerged and exposed due to changes in water level.
young-of-year	Fish that were born or hatched during the current year.

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TA 8 Attachment 1

Fish Species Life Histories

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TA 8 Attachment 1. Fish Species Life Histories

Introduction

Historically, the Colorado River from present-day Lake Powell downstream to the Southerly International Boundary (SIB) was a dynamic, seasonally warm, and turbid river, with flows ranging dramatically from approximately 1,000 to 120,000 cubic feet per second. This natural variability created a diverse and productive aquatic ecosystem, supporting a food base rich in organic material and providing suitable habitat for several warmwater native fish species (Miller 1961). However, the construction of major dams, beginning with Hoover Dam in 1935, along with increasing water demand and changing land use practices, fundamentally altered the river's flow patterns, sediment loads, turbidity, salinity, and temperature. The river was segmented into a series of impounded reservoirs separated by regulated stretches of flowing water, each altered from its original state by upstream impoundments. These changes had profound effects on both the aquatic food base and fish communities throughout the Colorado River ecosystem.

One of the most significant impacts of dam construction, particularly with Glen Canyon Dam, was the shift from natural seasonal flow variability to highly regulated daily fluctuations driven by hydropower production. Prior to dam construction, river levels remained relatively stable on hourly and daily timescales. With the advent of hydropower operations, water releases now peak during daylight hours to meet electricity demand and decrease at night, resulting in daily water level changes of 2-3 feet in some locations. This artificial "tidal" cycle has had a detrimental effect on the abundance and diversity of aquatic insects, which are a critical component of the river's food web (Kennedy et al. 2016). Insect eggs laid along the high-water mark are often left exposed and desiccate rapidly when flows recede, leading to reduced insect populations and, in turn, impacting the food availability for sensitive native fish species.

The closure of Glen Canyon Dam in 1963 marked a turning point for the Grand Canyon reach of the Colorado River, which historically experienced wide seasonal water temperature ranges—from 0 degrees Celsius (°C) in winter to nearly 30 °C in late summer and supported a habitat for several warmwater native fish species (Topping et al. 2003; Cole and Kubly 1976). Yet, even under natural conditions, only 8 of the 35 native fish species of the Colorado River were commonly found in this reach, with others confined to tributaries or present only seasonally (Gloss and Coggins 2005). The altered flow regime and cooler, clearer water released from upstream dams further reduced habitat suitability for native fish, while also affecting nutrient cycling and reducing the productivity of the aquatic food base. As a result, the fish community and food web in the Colorado River have been significantly transformed, with ongoing challenges for the persistence of native species and overall ecosystem health.

Native Fish Species

Threatened and Endangered Species

Humpback chub

The humpback chub (*Gila cypha)* is a medium-sized, long-lived fish species endemic to the Colorado River System. This member of the minnow family attains a length of about 450 millimeters (mm) (17.7 inches), a weight of about 1,000 g (2.2 pounds), and it may live as long as 40 years (Hendrickson 1993; Valdez and Ryel 1995; Andersen et al. 2010; FWS 2018a). The humpback chub was federally listed as endangered in 1967 and transferred into the protection of the Endangered Species Act (ESA) in 1973. It was reclassified from endangered to threatened on October 18, 2021 (FWS 2021). Critical habitat for the humpback chub includes 379 miles of the Colorado River System (59 Federal Register 13374). Within the analysis area, critical habitat includes the Colorado River from Nautaloid Canyon (river mile [RM] 35, river miles downstream from Lees Ferry) to Granite Park (RM 209) (FWS 1994). The lower 8 miles of the Little Colorado River are also critical habitat, but are outside of the analysis area.

In downlisting the humpback chub, the U.S. Fish and Wildlife Service (FWS) evaluated the stressors associated with the listing factors detailed in the species' status assessment (FWS 2018a). These include river flows (Factor A) and predatory, nonnative fish (Factor C) in Upper Basin populations. They also include water temperature (Factor A), food supply (Factor A), and predatory, nonnative fish (Factor C) in the Lower Colorado River Basin (Lower Basin; FWS 2021). Minimizing these threat factors was important for downlisting the species and preventing it from becoming endangered again. However, smallmouth bass present the highest potential impacts on Humpback Chub because the species can co-occur with Humpback Chub in certain canyon habitats and is a potential predator across its entire life history (FWS 2018a). Downlisting the species was also possible because a large reproducing aggregation became established in western Grand Canyon following a decline in the water level of Lake Mead that exposed historical habitat, and because of warmer releases from Glen Canyon Dam due to a lower level in Lake Powell (Rogowski et al. 2018; Van Haverbeke et al. 2017).

Distribution and Abundance

Five populations of humpback chub can be found in the Colorado River Basin: four in the Upper Basin and one in the Lower Basin (FWS 2018a). The Upper Basin populations are in (1) the Colorado River in Cataract Canyon, Utah; (2) the Colorado River in Black Rocks, Colorado; (3) the Colorado River in Westwater Canyon, Utah; and (4) the Green River in Desolation and Gray Canyons, Utah. The population in the Green and Yampa rivers in Dinosaur National Monument was recently extirpated with an effort to reintroduce and reestablish the species in this area beginning in 2021 (Valdez et al. 2021). The population in the Lower Basin is found in the Colorado River from about 30-Mile Spring (RM 30) downstream to Pearce Ferry Rapid (RM 281) in the Lake Mead inflow (Rogowski et al. 2018), in the lower 8 miles of the Little Colorado River to Chute Falls, and from translocations to Havasu Creek and Bright Angel Creek and above Chute Falls (Schelly 2019; FWS 2018a).

Historically, the humpback chub occurred in warm whitewater regions of the Colorado River upstream of present-day Hoover Dam and some larger tributaries in Arizona, Utah, Colorado, and Wyoming. The humpback chub is a specialized derivative of the *Gila robusta* complex that arose in response to the special conditions of large erosive and turbulent Colorado River habitats (Smith et al. 1979). The first description and photographic documentation of the humpback chub ("bony tail") was from an apparent spawning aggregation in 1908 near the mouth of the Little Colorado River near Beamer's Cabin in Grand Canyon (Kolb and Kolb 1914). The species was described in 1945 by Miller (1946) from three partial specimens caught near Bright Angel Creek in the Grand Canyon. The earliest catalogued collections of humpback chub and two congeneric species are from downstream of Lava Cliff Rapid (RM 246) in the 1940s, before the area was inundated by Lake Mead (Miller 1958). These collections consist of 27 specimens archived at the University of Michigan, including 5 humpback chub, 16 bonytail (*Gila elegans*), and 6 roundtail chub (*Gila robusta*); these specimens were described and confirmed morphologically and meristically by Bookstein et al. (1985). Wallis also reported eight juvenile humpback chub from Spencer Creek in 1950 (referenced in Valdez and Ryel 1997).

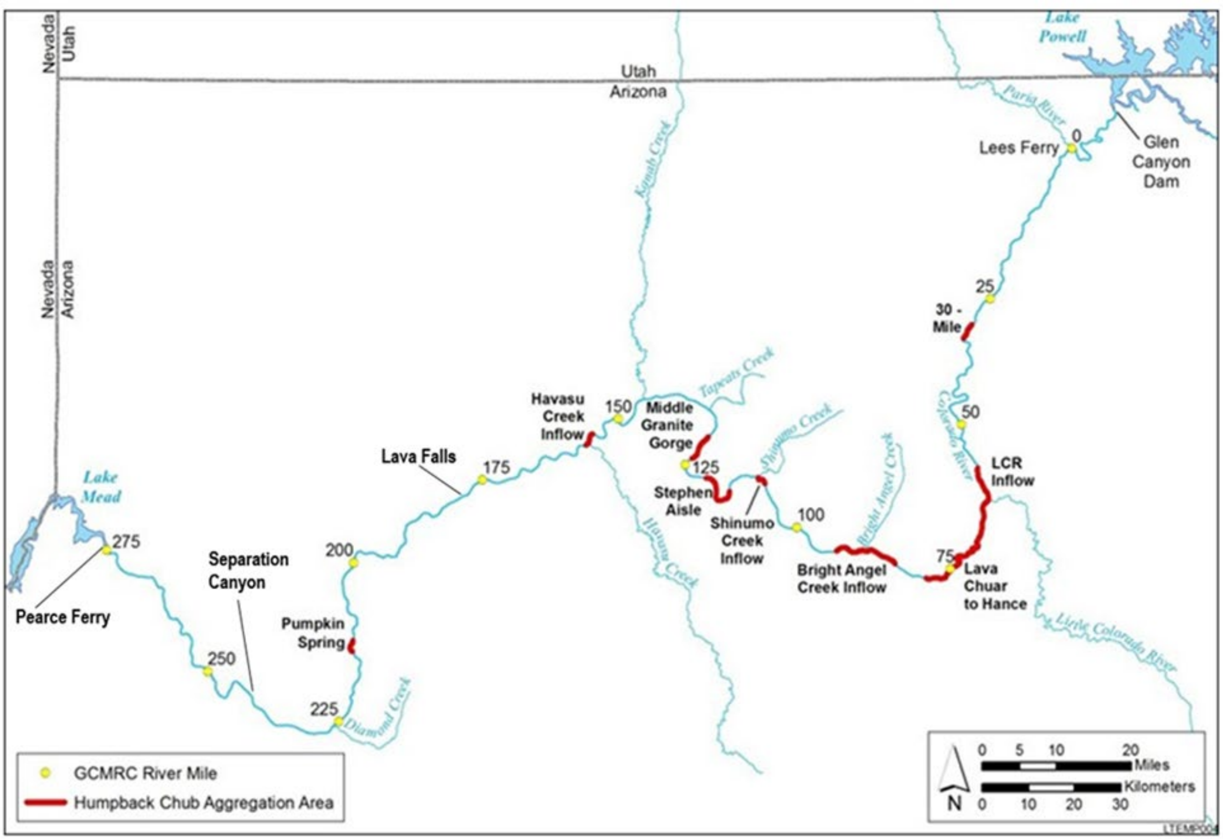
Following completion of Glen Canyon Dam in 1963, humpback chub were commonly reported in creel census from the Colorado River near Lees Ferry during 1963-1967 (Stone 1964, 1966, Stone and Queenan 1967), but sampling of seven major tributaries (excluding the Lower Colorado River) between Lees Ferry and Lake Mead in 1968 yielded none (Stone and Rathbun 1968). Holden (1973) collected 15 humpback chub in July 1967 and 1 in August 1970 in the Colorado River within a few hundred meters downstream of Glen Canyon Dam. Humpback chub have not been captured within 45 miles of the dam since the late 1970s (Carothers and Minckley 1981), or about the time Lake Powell stratified and cold-water releases began to persist year-round (Stanford and Ward 1991). Humpback chub were probably excluded from the tailwater by cold releases, high dam releases in 1983-1984, and by predation from large rainbow trout (*Oncorhynchus mykiss*) weighing up to 7 kg (Carothers and Brown 1991).

From 1970 through 1976, Suttkus and Clemmer (1977) reported young-of-year (YOY) and juvenile humpback chub between President Harding Rapid (RM 44) and Shinumo Creek (RM 108). Researchers from the Museum of Northern Arizona (Carothers and Minckley 1981) also captured chubs during six river trips in 1977-1979, including adults between North Canyon (RM 20) and Boulder Wash (RM 194) and one juvenile (less than 100 mm total length [TL]) just upstream of Granite Rapid (RM 93). Of 19 tributaries sampled from the Paria River (RM 1) to Travertine Creek (RM 230), humpback chub were captured only in the Little Colorado River.

From 1980 through 1981, biologists from the FWS captured 504 adult humpback chub between Nankoweap Canyon (RM 52) and Unkar Rapid (RM 72) (Kaeding and Zimmerman 1983). Fish abundance was reported to resemble a normal or "bell-shaped" distribution with the greatest numbers near the Little Colorado River inflow. Humpback chub smaller than 145 mm TL were not caught in the Colorado River upstream of the Little Colorado River inflow, although many small specimens were caught in spring and fall downstream of the inflow, providing the first evidence that post-dam humpback chub reproduction was occurring primarily in the Little Colorado River. From 1984 through 1989, the Arizona Game and Fish Department (AZGFD) (Maddux et al. 1987; Kubly 1990) reported humpback chub in the Colorado River from South Canyon (RM 32) to RM 217, mostly in or near the Little Colorado River (RM 61). Ninety-six percent of humpback chub captured were between RM 32 and RM 87, and specimens were also captured at the inflows of four tributaries, including Bright Angel, Shinumo, Kanab, and Havasu creeks, but not in the tributaries.

From 1990 to 1995, Valdez and Ryel (1995, 1997) sampled the Colorado River from Lees Ferry (RM 0) to Diamond Creek (RM 226) and found 5,940 YOY, juvenile, and adult humpback chub in nine aggregations (consistent and disjunct groups of fish with no significant exchange of individuals with other aggregations) (**Figure TA 8 Attachment 1-1**). The largest of these aggregations was associated with the Little Colorado River and included an estimated 3,482 adults that moved from the mainstem to the Little Colorado River in spring for spawning. The second largest aggregation was about 98 adults in Middle Granite Gorge, an area of talus slopes and debris fans, and the smallest aggregations had fewer than 10 fish. These aggregations became the basis for ongoing monitoring of humpback chub in the Grand Canyon (Persons et al. 2016; Rogowski et al. 2018).

Figure TA 8 Attachment 1-1
Humpback chub Aggregation Areas along the Colorado River between Glen Canyon Dam and Lake Mead, and the Area of Western Grand Canyon with the Expanded Population of Humpback chub



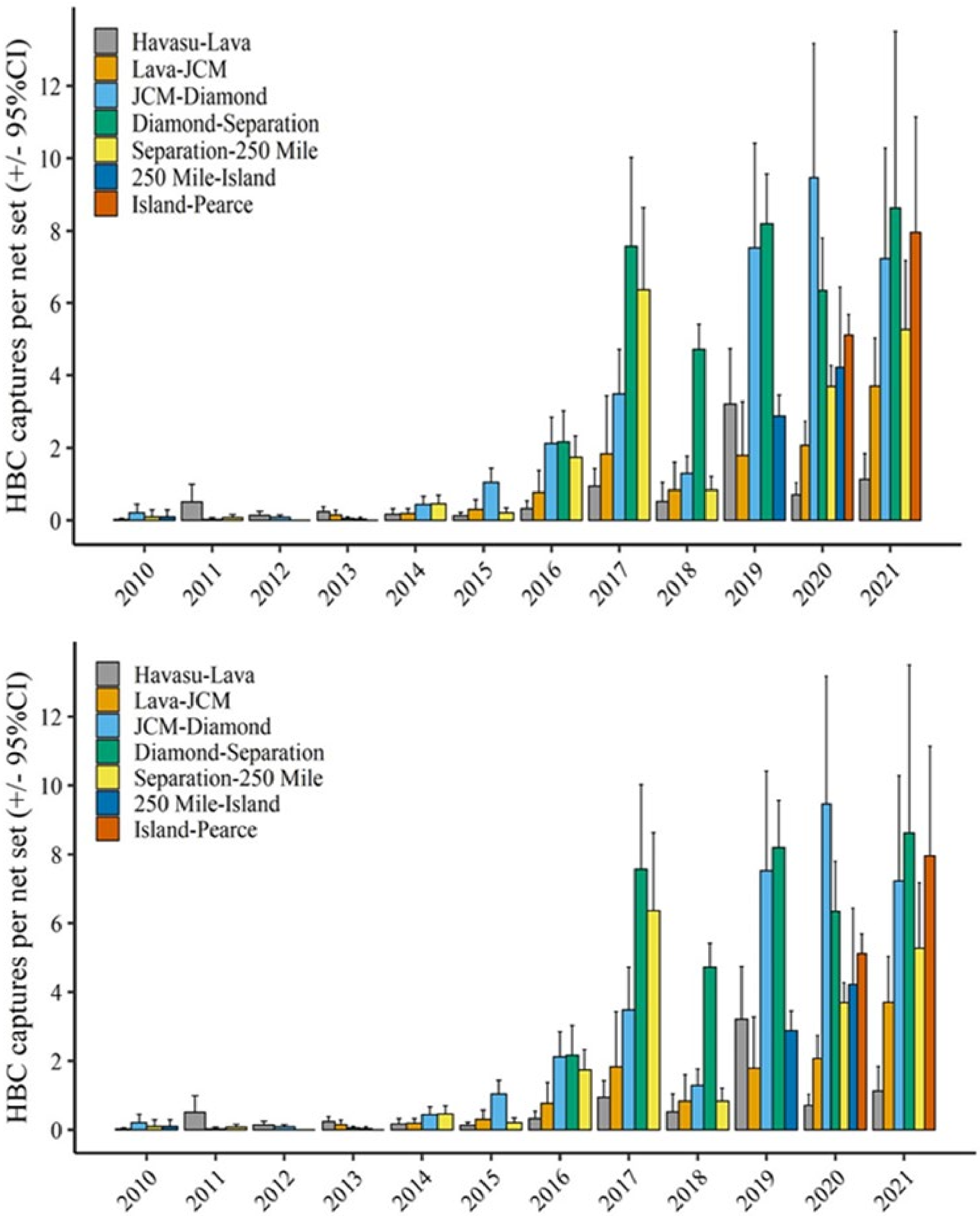
Sources: Valdez and Ryel 1995; Persons et al. 2016; Rogowski et al. 2018

Monitoring humpback chub in the Colorado River through Grand Canyon showed that numbers and distribution in the mainstem were increasing at a time of warmer water releases from Glen Canyon Dam. In 2015, catch rates of humpback chub downstream of Havasu Rapids began to increase and then tripled starting in 2017 (Van Haverbeke et al. 2017; Dzul et al. 2023) (**Figure TA 8 Attachment 1-2**).

Mark-recapture estimates of adults in the Colorado River in western Grand Canyon (Havasu Rapids to Pearce Ferry) showed an increase in numbers of about 20,000 in 2018 to about 66,000 in 2022 (Dzul et al. 2023; Van Haverbeke et al. 2023) (**Figure TA 8 Attachment 1-3**). Since about 2017, the western Grand Canyon has been populated by humpback chub representing all size classes, with the highest density of adults consistently between Lava Falls and Separation Canyon (RMs 180–240; Dzul et al. 2023). It was unclear if the humpback chub downstream of Havasu Rapids were a new population or an expansion of aggregations found upstream, but recent genetic studies have confirmed humpback chub in Grand Canyon to be genetically homogenous (Dzul et al. 2025). The humpback chub in the western Grand Canyon now comprises the largest group of humpback chub in the Colorado River System. These numbers compare to about 10,000–15,000 adults in the Little Colorado River/Colorado River aggregation (GCDAMP 2023; Yackulic et al. 2022) and about 4,000 adults for the sum of the four Upper Basin populations (FWS 2018a).

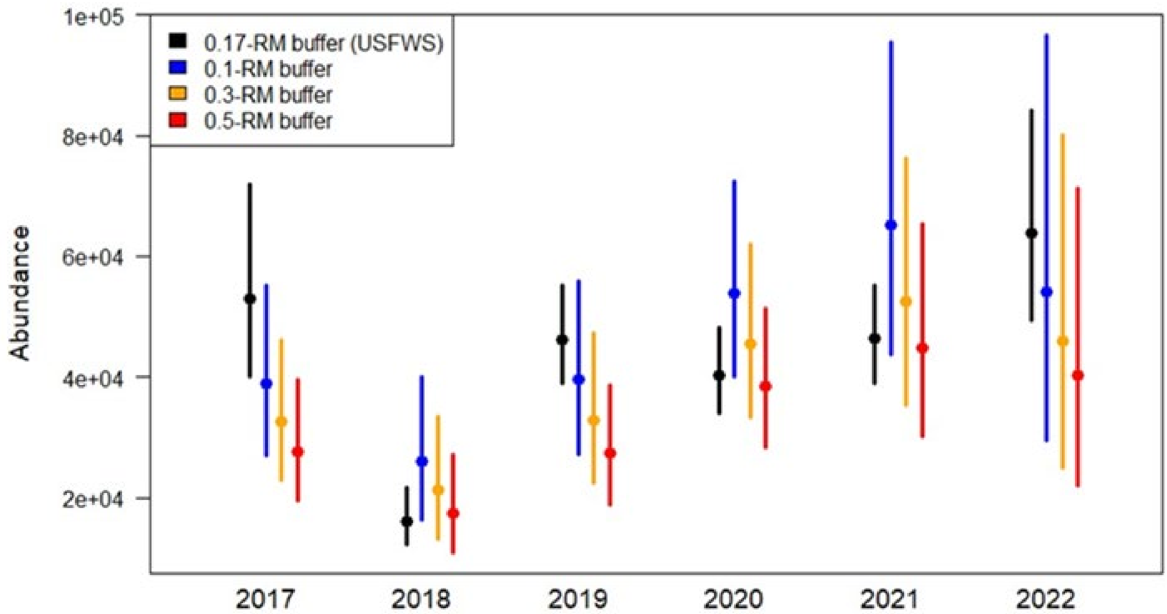
Since 2009, translocations of humpback chub have been made by the FWS to introduce juvenile fish upstream of Chute Falls in the Little Colorado River, and by the National Park Service (NPS) to introduce juveniles into Shinumo, Havasu, and Bright Angel creeks, with the goal of establishing additional spawning populations within the Grand Canyon (Healy et al. 2019; NPS 2013a, 2013b). Survey data from 2013-2015 suggest that translocated humpback chub have successfully spawned in Havasu Creek (Healy et al. 2020; NPS 2013a). Fish translocated to Shinumo Creek were extirpated by a fire and debris flow in August 2014 that scoured the stream channel and displaced or killed most of the fish that were translocated in 2010 (Nelson et al. 2014).

Figure TA 8 Attachment 1-2
Annual Catch-per-unit Efforts of Humpback chub at Sample Sites upstream (top) and downstream (bottom) of Havasu Rapids, 2010–2022



Source: Dzul et al. 2023

Figure TA 8 Attachment 1-3
Abundance Estimates of Humpback chub in western Grand Canyon (Havasu Rapids to Pearce Ferry), 2017–2022



Source: Dzul et al. 2023

Habitat

The humpback chub is typically found in canyon-confined, white-water reaches of large rivers, with adults, juveniles, YOY, and spawning occurring in distinct habitats within those reaches (FWS 2018a). Adults are usually associated with large deep eddy complexes (Tyus and Karp 1990; Valdez and Hoffnagle 1999), talus shorelines, and warm tributaries or springs (Valdez and Ryel 1995). Subadults and juveniles are found at the highest densities along shorelines comprised of vegetation, talus, and large boulders compared to shorelines with bedrock, cobble, or sand (Converse et al. 1998). Juvenile humpback chub along shorelines respond to changing river flows by shifting position to maintain similar habitat conditions of depth, velocity, and cover (Valdez and Ryel 1995; Korman et al. 2004).

Recent studies of an expanding population in western Grand Canyon found humpback chub in open, silt-laden habitats (Boyer et al. 2024), suggesting this species may be able to occupy a wider range of habitats than originally described. This habitat is found in exposed deltaic sediments at the inflow of Lake Mead following a decline in lake elevation. The Humpback chub Near-Shore Ecology Study collected juvenile humpback chub (under 3 years old) in all types of nearshore habitats, with the highest numbers collected from talus slopes (Dodrill et al. 2015). Prior to about 2017, the largest numbers of young humpback chub in the mainstem were in and downstream of the mouth of the Little Colorado River, but since then, large numbers of young have been found in the mainstem, especially downstream of Havasu Rapid to the Lake Mead inflow (Dzul et al. 2023). This indicates that mainstem spawning is occurring and may be widespread with increased water temperatures due to warmer water released from Glen Canyon Dam.

Nearshore habitats are important to humpback chub (and other native fish) because they provide shallow, productive, warm refugia for YOY and juveniles (Reclamation 1995; Hoffnagle 1996). Backwaters are a feature of the mainstem as sheltered habitat that form at the downstream end of recurrent channels in large eddy complexes and are one objective of High-Flow Experiments (HFEs; Melis et al. 2011). Temperature differences between the main channel and backwaters can be pronounced in backwaters. The extent of warming is variable and depends on the timing of the daily minimum and maximum flows, the difference between air and water temperatures, and the topography and orientation of the backwater relative to solar insolation (Korman et al. 2006). For example, summertime water temperatures in backwaters have been reported as high as 25 °C (77 degrees Fahrenheit [°F]), while main channel temperatures have been near 10 °C (50 °F) (Maddux et al. 1987), although mainstem temperatures have become warmer in recent decades due to warmwater releases from Glen Canyon Dam.

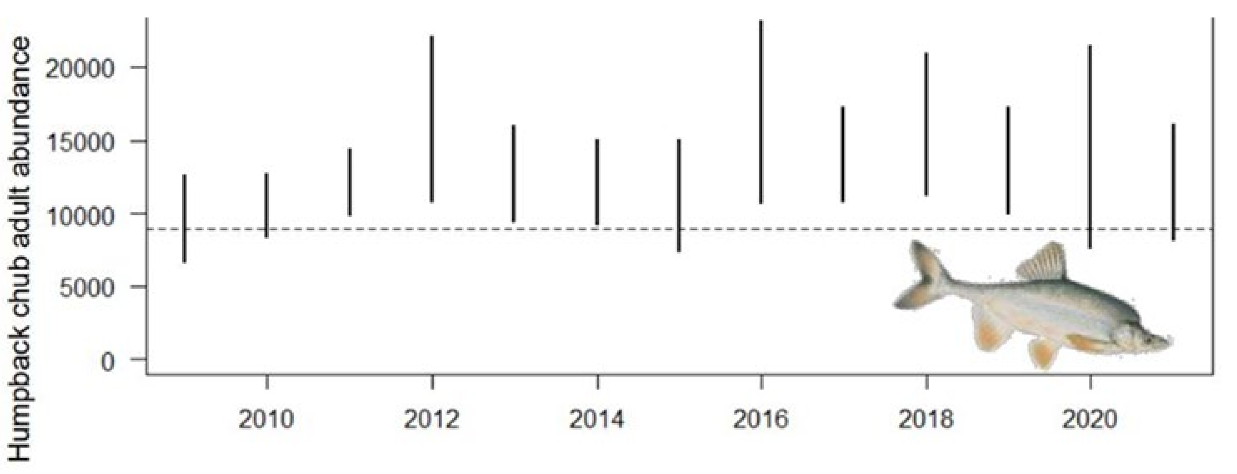
The amount of warming that occurs in backwaters is affected by daily fluctuations, which drain and fill backwater habitats with cold main channel waters (Valdez 1991; Angradi et al. 1992; Behn et al. 2010). During the low, steady, summer flow experiment conducted in 2000 of about 8,000 cubic feet per second, daytime temperatures in one backwater were as much as 13 °C (23 °F) warmer than in the adjacent main channel; temperature differences were much less at night (Vernieu and Anderson 2013). Backwater temperatures in summer have been reported up to 2–4 °C (3.6–7.2 °F) warmer under steady flows than under fluctuating flows (Hoffnagle 1996; Trammell et al. 2002; Korman et al. 2006; Anderson and Wright 2007). In general, the levels of warming observed in nearshore areas and backwaters during the low summer, steady flows in 2000 persisted only for short periods and were smaller than seasonal changes in water temperatures (Vernieu and Anderson 2013). Consequently, the temperature effects of steady flows on native fish were probably small.

Juvenile humpback chub in the Grand Canyon use backwater habitats when they are available, but the spatial extent of these habitats in the Colorado River is small compared to other nearshore habitats, such as talus slopes. Dodrill et al. (2015) reported that the total abundance of juvenile humpback chub was much higher along talus shorelines than in backwaters, and that when relative densities were extrapolated using estimates of backwater prevalence after an HFE, the majority of juvenile humpback chub were found outside backwaters.

When monitored with radiotelemetry or mark-recapture, adult humpback chub occupied localized habitats and showed little movement from those locations (Valdez and Clemmer 1982; Valdez and Ryel 1995). Radio-tagged adults in Grand Canyon exhibited a high degree of spatial fidelity for specific river locales; mean net movement was 1.49 kilometers (km; range= 0-6.11) for 69 radio-tagged fish and 1.64 km (range= 0-99.8) for 238 passive integrated transponders (PIT)-tagged fish (Valdez and Ryel 1997). A separate mark-recapture study reported that approximately 87 percent of recaptured fish were collected in the same mainstem river reach or tributary where they were originally tagged, with 99 percent of all fish recaptured in and around the Little Colorado River (Paukert et al. 2006). Van Haverbeke et al. (2022) reported 288 unique humpback chub detections near the Lower Colorado River confluence (RM 60.19–61.53) in 2021. Forty-four additional unique detections were documented outside of the Lower Colorado River confluence. Although most adults have high fidelity to specific river locales, some marked fish have moved as much as 100-154 km through the Grand Canyon (Valdez and Ryel 1997; Paukert et al. 2006). This large amount of movement is unique to the Grand Canyon and possibly driven by a need to reach seasonally warmed spawning sites, as radio-tagged fish in the Upper Basin did not exhibit long-range movements with spawning sites nearby (Valdez and Clemmer 1982; Kaeding et al. 1990).

Humpback chub have not been observed spawning in the wild because of the associated high turbidity and deep turbulent habitats. Where adults have been captured expressing eggs and milt and recently scraped abdomens, the fish were associated with shoreline or mid-channel bars of cobble/gravel substrate with moderate velocity, where females release eggs that are fertilized by an aggregation of multiple males (Valdez 1990; Tyus and Karp 1990; Gorman and Stone 1999). The main spawning area for the humpback chub within the Grand Canyon has been the Little Colorado River, which provides warm temperatures suitable for spawning and shallow, low-velocity pools for larvae (Gorman 1994). Many of the larval fish remain in the Little Colorado River for one or more years, and growth rates and survival are relatively high compared to estimates for the colder waters of the mainstem Colorado River (Dzul et al. 2014; Yackulic et al. 2014). Spring abundance estimates for age-1 humpback chub within the Little Colorado River from 2009 to 2012 ranged from about 1,000 to more than 9,000 individuals (Dzul et al. 2023), and numbers of adults ranged from about 10,000 to 15,000 (**Figure TA 8 Attachment 1-4**). Within the Little Colorado River, young humpback chub prefer shallow, low-velocity, nearshore pools and backwaters; they move to deeper and faster areas with increasing size and age (AZGFD 2001a; Stone and Gorman 2006). In the mainstem Colorado River, YOY fish may be found in backwater and other nearshore, slow-velocity areas that serve as nursery habitats (AZGFD 2001a; Robinson et al. 1998; Valdez and Ryel 1995). Valdez and Masslich (1999) found YOY humpback chub in a warm spring at RM 30, which indicates spawning, about 45 miles downstream of Glen Canyon Dam.

Figure TA 8 Attachment 1-4
Abundance of Adult Humpback Chub that Spawn in the Little Colorado River, 2009–2021



Source: Yackulic 2022; GCDAMP 2023

Life History

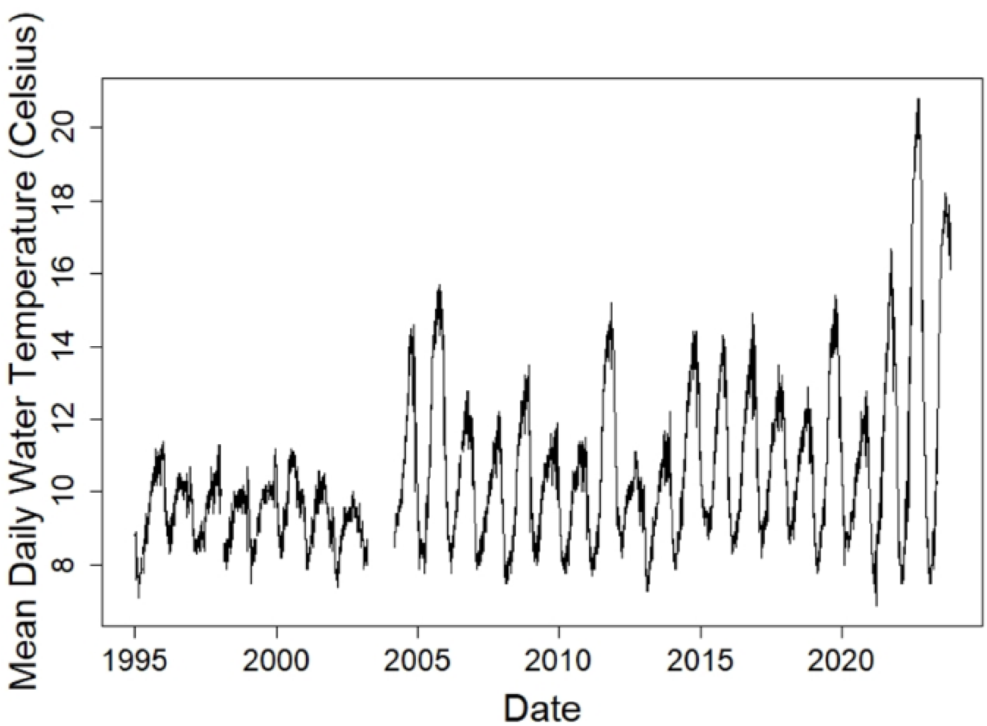
The humpback chub spawns annually in spring, starting at 4-5 years of age, and females broadcast up to 2,500 eggs that are 2-3 mm in diameter and are fertilized by multiple males over cobble/gravel substrates. The eggs are semi-adhesive and incubate 4–11 days at 16–22 ºC (Hamman 1982) before hatching into larvae that are 6-7 mm long (Muth 1990). There is little evidence of long-distance larval drift, as with other native cyprinids such as the Colorado pikeminnow (*Ptychocheilus lucius*), and the larvae are carried passively by currents to nearby sheltered shorelines consisting of boulders, cobble, and gravel (Valdez 1990). The larvae transform to juveniles at 30-40 days of age, to subadults at about 1 year of age, and to adults at 4-5 years of age. There is strong evidence of skipped spawning by humpback chub in the Little Colorado River population, where individuals, especially females, may not spawn every year, possibly due to the need to build up resources prior to spawning in a subsequent year (Pearson et al. 2015).

Although the humpback chub is a warmwater fish species, it persisted in the Colorado River through the Grand Canyon after the construction of Glen Canyon Dam and subsequent cold-water releases. Cold-water temperatures from 1970 to about 2000 provided suitable conditions for the species to persist in the mainstem as small aggregations but precluded mainstem spawning. Instead, adults of the most proximate aggregation moved annually to spawn in the Little Colorado River, and there was no reproduction in the other aggregations (Valdez and Ryel 1995; Douglas and Marsh 1996). Cold mainstem temperatures limited spawning and growth in the mainstem but enabled the species to live longer (Yackulic et al. 2014), with better body condition (Meretsky et al. 2000), and low infestations of diseases and parasites (Hoffnagle et al. 2006). Nine aggregations identified by Valdez and Ryel (1995) were mostly associated with warm springs and warm tributary mouths that enabled the species to persist in the mainstem as juveniles, subadults, and adults, while the largest and only center of reproduction was in the Little Colorado River, from which young fish drifted or moved to the mainstem.

The Grand Canyon humpback chub population has changed dramatically, starting in about 2004 with warmer dam releases. Humpback chub were mostly unable to reproduce in the mainstem Colorado River because of cold water temperatures, except for local reproduction at 30-Mile Spring (Andersen et al. 2010; Valdez and Masslich 1999). Nearly all reproduction was thought to occur in the lower 8 miles of the Little Colorado River (AZGFD 2001a; Van Haverbeke et al. 2017). Declining reservoir elevations in Lake Powell began in about 2002 (Vernieu et al. 2005; Dibble et al. 2021), leading to warmer releases. Warmer water in the mainstem allowed for higher juvenile humpback chub survival and growth that likely led to increases in the Little Colorado River population (Finch et al. 2016; Limburg et al. 2013; Yackulic et al. 2014). In western Grand Canyon, warmer water conditions led to reproduction and recruitment in the mainstem (Van Haverbeke et al. 2017), as well as better survival and what is now the largest population of humpback chub in the entire Colorado River System (Dzul et al. 2023). It is notable that in the 1940s, before Lake Mead filled, 5 humpback chub, 16 bonytail, and 6 roundtail chub were captured downstream of Lava Cliff Rapid (Bookstein et al. 1985; Miller 1958), an area that was inundated by Lake Mead for nearly 60 years and is now occupied by this large population of humpback chub following the decline in reservoir elevation and exposure of the original river channel.

The life history model for humpback chub is mostly known from the Little Colorado River population. Here, adult humpback chub move into the Little Colorado River from the Colorado River to spawn from March to May (Kaeding and Zimmerman 1983; Gorman and Stone 1999; FWS 2018a; Valdez and Ryel 1995). This species requires a minimum temperature of 15.5 °C (60 °F) to reproduce, but mainstem water temperatures typically ranged from 7–12 °C (45–54 °F) because of cold releases from Glen Canyon Dam (Andersen et al. 2010). Drought-induced warming and the lower levels of Lake Powell have resulted in mainstem water temperatures consistently exceeding 12 °C (54 °F) in summer and fall since 2004. Increased spawning and survival likely played a role in the increase of humpback chub in the Colorado River through Grand Canyon (Andersen et al. 2010; Coggins and Walters 2009; Yackulic et al. 2014) that led to the rapid and large expansion of humpback chub in western Grand Canyon (Van Haverbeke et al. 2017; Dzul et al. 2023). Rapid growth of small subadults was observed in the mainstem Colorado River near the Little Colorado River during July–October 2021-2023 that corresponded with the warmest water temperatures observed in decades (**Figure TA 8 Attachment 1-5**).

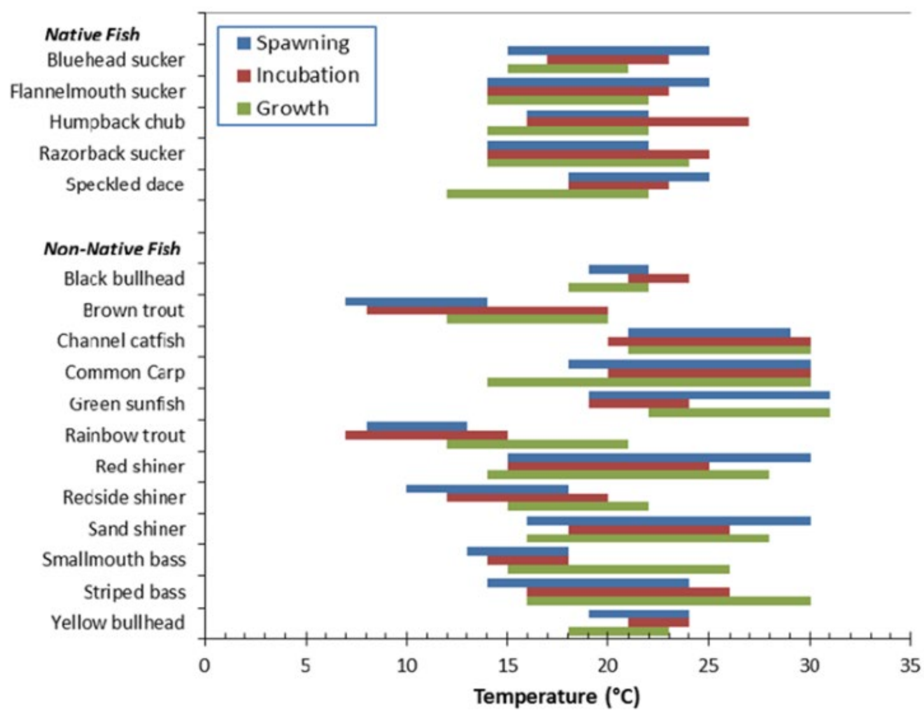
Figure TA 8 Attachment 1-5
Water Temperatures of the Colorado River at Lees Ferry as Measured at United States Geological Survey Gage #09380000, 1995 to Present



Source: USGS 2023
Note: Warmest temperatures of greater than 20 °C (greater than 68 °F) in 2022 were in late September and early October.

In 2022 and 2023, observed river temperatures downstream of the dam were the highest since the filling of Lake Powell; the temperatures were within the range of spawning temperatures for smallmouth bass (*Micropterus dolomieu*), green sunfish (*Lepomis cyanellus)*, and other warmwater nonnative fish (**Figure TA 8 Attachment 1-6**). Smallmouth bass and humpback chub share overlapping temperature preferences for growth, survival, and recruitment. However, prior to 2022, temperatures were sufficient for humpback chub populations to increase while smallmouth bass were relegated primarily to the Lees Ferry sub-reach in low numbers. Evidence of reproduction by smallmouth bass in 2023 raised concerns that smallmouth bass would become established in the Grand Canyon, where they would be likely to prey on humpback chub and other native fish species (Van Haverbeke et al. 2017; Gilbert et al. 2022; Dzul et al. 2023; Smallmouth Bass Ad Hoc Group 2023).

Figure TA 8 Attachment 1-6
Temperature Ranges for Spawning, Egg Incubation, and Growth by Native and Nonnative Fish of the Colorado River System downstream of Glen Canyon Dam



Source: Valdez and Speas 2007

Increased water temperatures can benefit humpback chub growth and survival, but can also allow for the establishment and expansion of warmwater nonnative species, which could result in increased predation on humpback chub (Ward 2011; Yard et al. 2011). Smallmouth bass, green sunfish, and walleye are of greatest concern, but predation by channel catfish (*Ictalurus punctatus)* and black bullhead is also thought to threaten humpback chub in the Grand Canyon, particularly with warmer water conditions (NPS 2013b). Because of their larger size, adult humpback chub are less likely to be preyed on by trout; however, emergent fry, YOY, and juvenile humpback chub in the mainstem Colorado River in the vicinity of the Little Colorado River are susceptible to predation by trout (Yard et al. 2011).

Under cold water conditions, predation by rainbow trout and brown trout (*Salmo trutta*) at the Little Colorado River confluence was identified as a mortality source affecting humpback chub survival, reproduction, and recruitment (Valdez and Ryel 1995; Marsh and Douglas 1997; Yard et al. 2011). This may change with warmer mainstem temperatures. Ward and Morton-Starner (2015) conducted laboratory studies that showed predation by rainbow trout on YOY humpback chub decreased from approximately 95 percent to 79 percent as water temperature increased from 10–20 °C (50–68 °F); however, predation success by brown trout was about 98 percent and did not change significantly over the same temperature range. Yard et al. (2011) examined the effects of temperature on trout piscivory in the Colorado River and reported no relationship between water temperature and the incidence of piscivory by rainbow trout, but a significant positive correlation between increased water temperature and the incidence of piscivory by brown trout.

An additional concern for the humpback chub in Grand Canyon is nonnative fish parasites, including the Asian tapeworm (*Bothriocephalus acheilognathi*) and anchor worm (*Lernaea cyprinacea*) that affect the condition and survival of individuals (Clarkson et al. 1997; Hoffnagle et al. 2006; Andersen et al. 2010). While cold-water releases from Glen Canyon Dam have limited reproduction and recruitment of humpback chub (and other native fish) in the mainstem Colorado River, warmer mainstem temperatures over the last two decades have been sufficiently high to allow these parasites to complete their life cycles and expand, as well as other diseases and parasites (Hoffnagle et al. 2006).

Prevailing ecological conditions and water management actions within the Grand Canyon can affect the food base of the humpback chub. The humpback chub is primarily an insectivore, with larvae, juveniles, and adults all feeding on a variety of aquatic insect larvae and adults, including dipterans (primarily chironomids and simuliids), Thysanoptera (thrips), Hymenoptera (ants, wasps, and bees), and amphipods (such as *Gammarus lacustris*) (AZGFD 2001a; Cross et al. 2013; Kaeding and Zimmerman 1983; Valdez and Ryel 1995). Feeding by all life stages may occur throughout the water column, at the water surface, and on the river bottom.

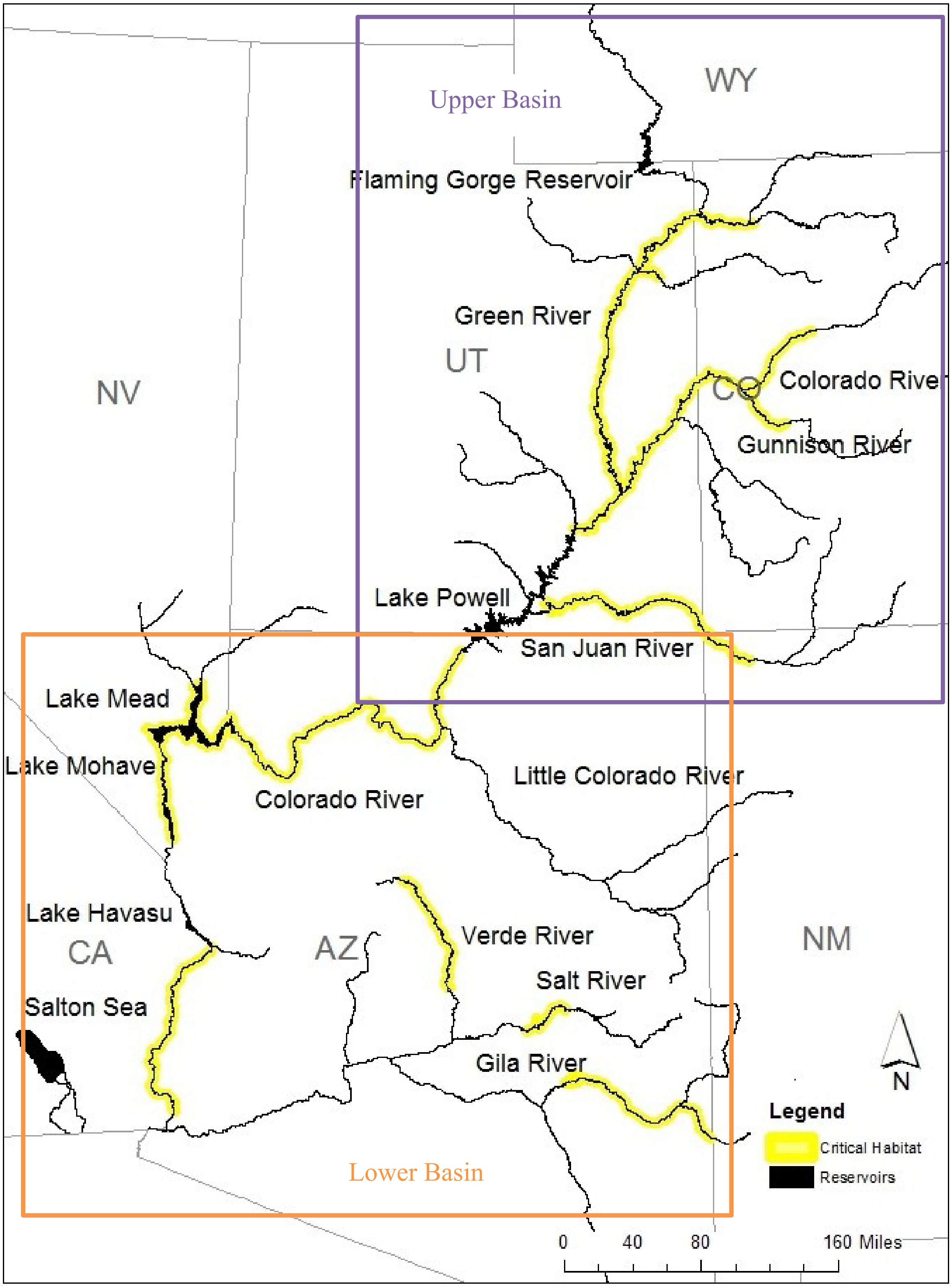
Razorback sucker

The razorback sucker (*Xyrauchen texanus*) is a large river sucker (Catostomidae) endemic to the Colorado River System. The razorback sucker is one of four endemic Colorado River fishes that Minckley (1973) described as rare in the Lower Basin. It is a large fish that may live more than 50 years, with adults reaching lengths of up to 1,000 mm TL (39 inches) and weights of 5–6 kg (11–13 pounds). However, they are more typically found within the 400–700 mm TL (16–28 inch) range and weigh less than 3 kg (7 pounds) (FWS 2018b). Historically, razorback suckers were used extensively as food by native cultures, and populations were commercially harvested as recently as 1949 (FWS 2018b). The razorback sucker was listed as federally endangered in 1991(56 Federal Register 54957) and, more recently, has been included under the auspices of the Lower Colorado River Multi-Species Conservation Program, with the goal of working towards the recovery and persistence of rare fishes in the Lower Basin. A Species Status Assessment (FWS 2018b) found that the status of the razorback sucker has improved, and a Proposed Rule in 2021 (Federal Register Vol.

86, No. 127, July 7, 2021) proposed to reclassify the species from endangered to threatened because the *"...species' status has improved due to conservation actions and partnerships, and the threats to the razorback sucker identified at the time of listing in 1991 have been eliminated or reduced to the point that the species is no longer currently in danger of extinction throughout all or a significant portion of its range, but it is still likely to become so within the foreseeable future without current active and intensive management."*

Critical habitat was designated as 2,776 km (1,724 miles) of the Colorado River Basin on March 21, 1994 (59 Federal Register 13374). This was composed of 1,519 km in the Upper Basin and 1,255 km in the Lower Basin (**Figure TA 8 Attachment 1-7**). Within the analysis area, critical habitat includes (a) the Colorado River and its 100-year floodplain from Westwater Canyon to full pool elevation, upstream of North Wash and including the Dirty Devil arm of Lake Powell, (b) the San Juan River and its 100-year floodplain from the Hogback Diversion to the full pool elevation at the mouth of Neskahai Canyon on the San Juan arm of Lake Powell, (c) the Colorado River and its 100-year floodplain from the confluence with the Paria River to Hoover Dam, including Lake Mead to the full pool elevation, (d) the Colorado River and its 100-year floodplain from Hoover Dam to Davis Dam, and (e) the Colorado River and its 100-year floodplain from Parker Dam to Imperial Dam including Imperial Reservoir to the full pool elevation or 100-year floodplain, whichever is greater.

Figure TA 8 Attachment 1-7
Critical habitat designated for the razorback sucker in 1994 (yellow highlight), and general areas defined as Upper and Lower Basins (modified from FWS 2018b)



Distribution and Abundance

Razorback sucker are endemic to the Colorado River Basin, which encompasses parts of seven western states, including Arizona, California, Colorado, Nevada, New Mexico, Utah, and Wyoming, as well as parts of Sonora and Baja California, the United Mexican States (Mexico). The Upper and Lower Basins are split at Lees Ferry, Arizona, which is 15 miles downstream of Glen Canyon Dam. Glen Canyon Dam provides physical separation between the Upper and Lower Basins. Historically, razorback sucker were widespread and common throughout the larger rivers of the entire Basin, with particularly high razorback sucker abundance in the Lower Basin near Yuma, Arizona. Razorback sucker are thought to have been uncommon in turbulent, canyon-bound reaches, with robust populations typically being found in calm, flatwater river reaches. Historical populations were affected by the construction of multiple in-stream impoundments in the early to mid-1900s, which changed riverine conditions. Razorback suckers have persisted in some resulting reservoirs but have been extirpated in others (Marsh and Minckley 1989; FWS 2018b).

Razorback sucker adults are present and spawning across much of the Colorado River Basin, but some areas have patchy distributions. Species presence is likely due to its ability to utilize both lotic (rapidly moving fresh water, like a river) and lentic (still, fresh water, like a reservoir or lake) environments. Four populations, all maintained by stocking, are currently present in the Upper Basin, including in the Green, Colorado, and San Juan subbasins and Lake Powell. Lower Basin populations currently occur in Lake Mead (and the Colorado River upstream of Lake Mead), Lake Mohave, the Colorado River between Davis and Parker dams (Lake Havasu), and the Colorado River downstream of Parker dam, and all except the first are maintained by stocking. Although other, smaller, isolated, and more localized sources of razorback sucker can be found in various ponds, backwaters, and other habitat types common to the Lower Basin, they are not included herein as not all are sampled routinely or with the specific purpose to produce abundance estimates (FWS 2018b), and they are not in the analysis area.

Ten historical records exist for razorback sucker between Glen Canyon Dam and the upper extent of the inflows of Lake Mead between 1944 and 1990 (Valdez and Carothers 1998, FWS 2018b). Razorback sucker were detected in 1990 at the confluence of the Little Colorado River, but they were thought to be functionally extirpated in the Grand Canyon (Clarkson and Childs 2000).

Until recently, the razorback sucker was considered extirpated from the Grand Canyon, with the last individual collected in 1993 at RM 39.3. However, subsequent captures of razorback suckers and flannelmouth-razorback sucker hybrids in the Little Colorado River (Marsh and Douglas 1997) and western Grand Canyon (Bunch et al. 2012; Rogowski and Wolters 2014; Rogowski et al. 2015) indicated the species' continued presence. Notably, four sonic-tagged fish from Lake Mead were detected in Grand Canyon National Park up to Quartermaster Canyon (RM 260) in 2012 (Kegerries and Albrecht 2012), and additional adults were collected near Spencer Creek (RM 246) in 2012 and 2013 (Bunch et al. 2012; GCMRC 2014). Since 2010, studies in the Lake Mead inflow and, since 2014, in the lower Grand Canyon have verified razorback sucker spawning and larval production within Grand Canyon National Park, with larvae documented as far upstream as RM 179 (Albrecht et al. 2014), and in 2019, eight larvae were captured between RM 127.3 and RM 279.0, extending the known upstream distribution by 17.5 RM compared to the previous record (Rogers et al. 2023).

However, no razorback sucker larvae have been detected since 2019, suggesting that spawning in the mainstem river may be limited or no longer occurring.

Although captures of any life stage of razorback sucker are rare in the Grand Canyon, telemetry of translocated razorback sucker has been useful for documenting use and movement since 2014 (Albrecht et al. 2014). Since 2014, six translocated fish have been detected through telemetry above RM 30 (Rogers et al. 2023). Most recently, Roger et al. (2023) reported that a translocated adult razorback sucker was detected via sonic telemetry to have been near RM 15.

The largest reservoir population of razorback sucker, estimated at 75,000 in the 1980s, occurred in Lake Mohave, Arizona and Nevada, but at present, no wild fish remain because of a lack of recruitment (Marsh et al. 2003; Marsh et al. 2005; Marsh et al. 2015). However, Lake Mohave remains an actively managed and important genetic refuge for the species where larvae are collected, reared in the hatchery, and released back into the reservoir at larger sizes to improve survival, as little to no natural recruitment occurs because of predation (Marsh et al. 2015; FWS 2018b). Razorback sucker were also often reported in the riverine Lower Basin downstream of Lake Mohave from the early 1940s through early 1980s (FWS 2018b), and presently one of the largest populations of this species occurs in the reach between Davis Dam and Parker Dam (LCR MSCP 2024).

The Gila River system encompasses the Verde, Gila, and Salt rivers and their tributaries. Razorback sucker historically occupied all larger streams in the Gila River Basin, including the Salt, Verde, and Gila Rivers (FWS 2018b). By the 1970s, the species was extirpated from the Basin, and by 1981, efforts to reestablish razorback suckers had begun, but lack of survival precluded additional work in these locations (FWS 2018b).

The decline of the razorback sucker throughout its range has been attributed primarily to habitat loss due to dam construction, the loss of spawning and nursery habitats as a result of diking and dam operations, and alteration of flow hydrology (AZGFD 2002a; FWS 2018b). For example, the 80 percent reduction in the historical distribution of this species has been attributed to the construction of Hoover, Parker, Davis, and Glen Canyon dams on the Colorado River and Flaming Gorge Dam on the Green River (Mueller et al. 2000). In addition, competition with and predation by nonnative fish have also been identified as important factors in the decline of this species (Minckley et al. 1991; FWS 2002a). In the Grand Canyon, the decline of native fish, including razorback sucker, has been attributed to multiple factors, including modifications to river temperatures and discharge patterns due to Glen Canyon Dam and the subsequent establishment of nonnative fish populations. This has led to more than 2 decades of experimental actions to understand the factors that influence the occurrence, abundance, and distribution of native fish in the Grand Canyon (Coggins et al. 2011; Mueller et al. 2000; FWS 2018b).

Razorback sucker in the Grand Canyon have been observed, and evidence of spawning has been observed in the river from at least Lava Falls through the entirety of Lake Mead, and it maintains a reproducing population in the analysis area (Albrecht et al. 2014; Kegerries et al. 2015). Currently, there is little information on the habitat use and life history needs of the species in the Grand Canyon and Lake Mead.

Habitat

The razorback sucker uses a variety of habitats, ranging from mainstem channels to slow backwaters of medium and large streams and rivers (AZGFD 2002a; FWS 2018b). In rivers, habitat requirements of adults in spring include deep runs, eddies, backwaters, and flooded off-channel areas; in summer, runs and pools, often in shallow water associated with submerged sandbars; and in winter, low-velocity runs, pools, and eddies. (FWS 2018b). In reservoirs, adults prefer areas with water depths of 1 meter (3.3 feet) or more over sand, mud, or gravel substrates. Young require nursery areas with quiet, warm, shallow water such as tributary mouths, backwaters, and inundated floodplains along rivers, and coves or shorelines in reservoirs (FWS 2018b). Captures of larval razorback sucker in the western Grand Canyon in 2014 and 2019 found the highest density of larvae in isolated pools, which comprised less than 2 percent of all habitats sampled (Albrecht et al. 2014; Kegerries et al. 2019). Similar results were found in 2015, when the highest catch of larval razorback sucker was found in isolated pools, followed by backwaters, which comprised 2.1 percent and 9.1 percent of habitats sampled, respectively (Kegerries et al. 2015). Marsh et al. (2024) provide an exhaustive review of the current concepts related to the role of isolated backwaters and their creation for this species, the need for genetic monitoring, and other pertinent management actions for razorback sucker in the Lower Basin.

Life History

Adults and immature razorback sucker are omnivorous, feeding on algae, zooplankton, and aquatic insect larvae. In Lake Mohave, their diet is dominated by zooplankton, diatoms, filamentous algae, and detritus (Marsh 1987). Razorback suckers exhibit relatively fast growth in the first 5 to 7 years of life, after which growth slows and possibly stops (AZGFD 2002a). Both sexes mature by about age 4. Spawning in rivers occurs over bars of cobble, gravel, and sand substrates during spring runoff at widely ranging flows and at water temperatures typically greater than 14 °C (57 °F) (FWS 2002a, 2018b). In reservoirs, spawning occurs over rocky shoals and shorelines. Temperatures for spawning, egg incubation, and growth of this species range from 14–25 °C (57–77 °F).

Hatching success is temperature dependent, with 100 percent mortality occurring at temperatures less than 10 °C (50 °F); optimum temperatures for adults are around 22–25 °C (72–77 °F) (AZGFD 2002a). Based on back calculation from the dates of larval collection, Kegerries et al. (2015) estimated that the onset of spawning in western Grand Canyon was in mid-February when average daily water temperatures were between 10 °C and 12 °C (50 °F and 54 °F). Spawning appeared to peak toward the end of March when water temperatures were in the range of 12–14 °C (54–57 °F), although the entire spawning period was estimated to range from mid-February to July (Kegerries et al. 2015).

Historically, this species exhibited upstream migrations in spring for spawning, although current populations include groups that are sedentary and others that move extensively (Minckley et al. 1991). Adults in the Green River subbasin move as much as 62 miles to specific areas to spawn (Tyus and Karp 1990). In Lake Mohave, individuals move 12-19 miles between spring spawning and summer use areas (Mueller et al. 2000). Kegerries et al. (2015) reported that sonic-tagged razorback suckers moved up to 224 miles within the western Grand Canyon, the Colorado River inflow to Lake Mead, and throughout Lake Mead as a habitat used by all life stages of this species.

Bonytail

The bonytail is a medium-sized fish species endemic to the Colorado River and its tributaries. The species was historically widespread and common from Mexico to Wyoming, but by the 1970s had declined to fewer than 50 known individuals (FWS 2002b). The bonytail was listed as an endangered species in 1980 because of extirpation from most of its range caused by habitat alteration (45 Federal Register 27710; April 23, 1980). Critical habitat for the bonytail (59 Federal Register 13374, March 21, 1994) includes 502 km (312 miles) of the Colorado River System. Within the analysis area, critical habitat includes (a) the Colorado River from Brown Betty Rapid to Imperial Canyon in the Lake Powell inflow, (b) the Colorado River from Hoover Dam to Davis Dam including Lake Mohave up to its full pool elevation, and (c) the Colorado River from the northern boundary of Havasu National Wildlife Refuge to Parker Dam including Lake Havasu up to its full pool elevation. The elevation at Imperial Canyon is 3,680 feet, which is within the full pool elevation of Lake Powell of 3,700 feet, the defined analysis area of this Draft Environmental Impact Statement (EIS).

A Bonytail Recovery Plan was approved in 1990 and amended and supplemented by Recovery Goals in 2002 (FWS 2002b). A Species Status Assessment has not been developed for the bonytail, and the most recent 5-Year Status Review (FWS 2024) recommended no change in the listing status. The current Recovery Priority Number of 5C is indicative of a species facing a high degree of threat, having a low recovery potential, and is listed at the species level, and there is the potential for conflicts between needed recovery actions and economic activities. The bonytail is included in the Upper Colorado River Recovery Program and the Lower Colorado River Multi-Species Conservation Program. These programs undertake management actions for the conservation of the species, including stocking, flow management, nonnative fish control, and habitat development. Despite management efforts, signs of bonytail survival in the wild remain rare, and the ecology of the species remains poorly understood (Bestgen et al. 2008). Without viable, wild bonytail populations, the species continues to rely on hatchery propagation to persist in the wild and to advance recovery efforts. The present hatchery propagation program is based on a founder population of 11 individuals (6 females and 5 males) taken from Lake Mohave between 1976 and 1978 (FWS 2002b).

Distribution and Abundance

The bonytail was historically widespread and common throughout the larger rivers and tributaries of the Colorado River System, with historical captures documented from Mexico to Wyoming (Behnke and Benson 1980; Minckley and Deacon 1991; Mueller and Marsh 2002; FWS 2002b). The first recorded bonytail from the Upper Basin was in 1889, when one specimen was captured from the Green River. Subsequent historical collections, limited largely to anecdotal and historical interviews with limited scientific documentation, indicate a once-expansive range of bonytail inhabitance (FWS 2002b). During the 1950s, bonytail populations underwent a large but poorly documented decline in abundance following extensive biotic and abiotic habitat modifications, including dam construction, water withdrawal, land use practices, and widespread introductions of nonnative fish (Miller 1961). Holden (1991) described the effects of a large-scale rotenone treatment in the upper Green River and provided insight into the rather large population of bonytail present until 1962, at the time of the treatment. Bonytail numbers were drastically reduced in the Green River following the closure of Flaming Gorge Dam in 1963 (Vanicek and Kramer 1969; Holden and Stalnaker 1975). Very few bonytail have been captured in the Upper Basin since the 1980s, including one from Desolation

Canyon of the Green River during 1979-1981 (Tyus et al. 1982), 14 juveniles and adults from Cataract Canyon during 1985-1988 (Valdez 1990), and one from Black Rocks (Kaeding et al. 1986). These were probably the last wild bonytail captured in the Upper Basin before widespread stocking of hatchery fish (Bestgen et al. 2008).

Bonytail numbers in the Lower Basin followed a similar trend of rapid decline (FWS 2002b). FWS (2002b) summarizes and references historic captures downstream of present-day Lake Powell. In the 1940s, 16 individuals were captured from the Grand Canyon (as referenced in FWS 2002b). In the 1950s, a large aggregation estimated at 500 adults was located spawning over a shelf of angular rock and gravel in Lake Mohave. Because of the low numbers of bonytail seen in the wild, the FWS and its cooperators initiated a program in 1974 to collect the few remaining wild bonytail and establish a broodstock for reintroducing the species and reestablishing wild populations (FWS 2002b). Thirty-four bonytail were captured in Lake Mohave from 1976 to 1988, and 11 of these fish (6 females and 5 males) were used to establish a hatchery broodstock, the progeny were stocked into Lake Mohave and Lake Havasu, as well as in the Upper Basin (Minckley et al. 1989; Minckley et al. 1991; FWS 2002b). From 1981 to 2013, approximately 209,500 bonytail were stocked into Lake Havasu (Humphrey et al. 2014), and from 1991 to the end of 2022, 483,224 bonytail were stocked in the Lower Basin (Marsh et al. 2024). Of these, approximately 98,172 were stocked into Lake Mohave and lakeside backwaters (the mouths of inundated desert washes separated from the reservoir by wind-generated gravel berms), 284,392 into Lake Havasu and adjacent habitats, 99,591 into the Lower Colorado River mainstem and its connectives downstream of Lake Havasu, and 1,069 into inland impoundments and isolated backwaters along the Lower Colorado River. Bonytail stockings into the Lower Colorado River or connected backwaters have produced no evidence of long-term survival, reproduction, or recruitment, while stockings into habitats free of nonnative fishes have demonstrated survival, reproduction, and even recruitment, suggesting that the future of this species may depend on isolated habitats and intensive nonnative fish management (Marsh et al. 2024).

Hatcheries in the Upper Basin produce and stock over 35,000 adult bonytail per year into Upper Basin rivers, including the Green, White, Yampa, Dolores, Gunnison, San Rafael, Price, and Colorado (Integrated Stocking Plan Revision Committee 2015). The hatcheries include Utah's Wahweap Hatchery (Big Water, Utah), the FWS's Ouray Hatcheries (Grand Junction, Colorado and Randlett, Utah), and Colorado's Native Aquatic Species Recovery Facility (Alamosa, Colorado). Bonytail have spawned naturally in ponds at Wahweap State Fish Hatchery, creating an overabundance of fish that exceeds hatchery capacity. Hence, hatchery managers released over 150,000 untagged juvenile bonytail into Lake Powell between 2016 and 2023 in addition to their traditional stocking targets (FWS 2024).

Two bonytail were captured recently in the Colorado River downstream of Glen Canyon Dam (FWS 2024). In July 2023, an untagged bonytail was found downstream of the dam in the -12-mile slough. The size of the bonytail indicates it was likely part of a 2023 stocking event from Wahweap that survived in Lake Powell and passed through the dam penstocks. A second bonytail was captured in Glen Canyon in fall of 2023 that was PIT-tagged and released alive. Additional bonytail were detected in Lake Powell. One bonytail was captured in a gill net in Lake Powell in March 2023. When sampling for larval razorback sucker in the Colorado River arm of Lake Powell in 2022, Utah Division of Wildlife Resources biologists found two larval bonytail, indicating that the species naturally reproduced in the Colorado River inflow area. Lastly, Rogers et al. (2024) summarize the capture of a single adult bonytail in good health in Lake Mead near the Las Vegas Wash inflow.

Habitat

Information on bonytail habitat use is limited because of the small numbers of individuals left in the wild and due to the extirpation of this species prior to extensive sampling of the Colorado River (Bestgen et al. 2008; Marsh et al. 2024). Early fisheries surveys indicate that bonytail were found in high gradient, gravel, riverine sections, but this observation is probably based on locations of the last few wild fish captured in the Upper Basin (Bestgen et al. 2008). Bonytail are characterized as being adapted to the swifter sections of the Colorado River, with affinity for areas of high flow and rocky habitat types (Miller 1946). Like other native fishes, backwaters and other slackwater habitats are thought to serve as important nursery areas for young bonytail (FWS 2002b). Available information suggests that adult bonytail display similar habitat affinities to those of other native fishes, with a preference for deep, fast-water sections, as well as eddy and pool habitats. Vanicek (1967) noted bonytail habitat selection coincided with habitats occupied by roundtail chub and found these species not only in pools and eddies near "fast-flowing" riverine areas but also in slower sections. Valdez (1990) reported that wild bonytail used the same habitats as humpback chub, with individuals being found in shoreline eddy habitats, boulders, and cobble, and near swift water sections in Cataract and Desolation canyons. Miller (1946) hypothesized that the unique morphological characteristics of the bonytail (long, slender body, adnate scales, and thin, powerful tail) and the humpback chub (large stabilizing dorsal hump and large falcate fins) made these species uniquely adapted to the swift currents, high turbidity, and high salinity of the Colorado River. A laboratory study by Pimentel and Bulkley (1983) showed that hatchery bonytail, when given the opportunity, selected water with high levels of total dissolved solids. Bonytail can persist in water with total dissolved solids of 4,700 mg/L, the highest tolerance reported for any of the *Gila* species of the Colorado River. However, hatchery-produced bonytail may not exhibit the same habitat selection as wild bonytail. Bestgen et al. (2008) found stocked bonytail occupying nearly all habitat types, including riffles that were thought to be too demanding for this species energetically, and likely an atypical habitat choice.

Bonytail were observed spawning over rocky substrate in a reservoir (FWS 2002b) and likely use similar habitat to spawn in both flowing and standing water (Mueller 2006). In the Lower Basin, documented successful natural reproduction in Cibola High Levee Pond indicates that bonytail spawned on shorelines associated with riprap materials (large-diameter gravel, cobble, and boulder substrates) in water 3–6 feet deep (Mueller et al. 2003). Sonic-telemetry studies have also revealed that adult bonytail prefer interstitial spaces associated with shoreline riprap during daylight hours, whereas open-water areas are more commonly used at night, possibly because of the high clarity associated with the off-channel pond. Individuals spawning in Lake Mohave displayed similar diel habitat shifts where adults were found in deeper habitats during the day, and at dark they formed congregations along shallower shoreline habitats (Mueller and Marsh 2002). Intensive telemetry surveillance showed a high degree of site-specific habitat fidelity, with individually marked bonytail consistently returning to the same cavities within the riprap-type shoreline. Young bonytail were most often associated with areas of dense overhead cover in depths greater than 1 m, and they schooled in warm, shallow areas of an oxbow pond (Mueller et al. 2003; Mueller 2006). These findings suggest that habitat for bonytail should include riprapped shoreline materials, and that bonytail are a highly cover-affiliated (likely vegetation and/or turbidity) species with one of the few specific habitat preferences that has been well documented.

Life History

The life history of the bonytail is believed to be like that of other closely related chub species that are warmwater spring broadcast spawners. Vanicek and Kramer (1969) documented the last substantial spawning of a wild, riverine population of bonytail in Dinosaur National Monument. Ripe fish were collected from mid-June through early July (1964–1966) in water temperatures around 18 °C (64 °F). Bonytail estimated between 5 and 7 years old were determined to be sexually mature (Vanicek 1967), whereas in controlled hatchery environments, Hamman (1982) found that bonytail began to mature sexually at age 2. Johnston (1999) classified bonytails as being broadcast spawners and suggested that the loss of eddy habitats due to the construction of impoundments probably contributed to the apparent reproductive failure of the bonytail and a closely related species, the humpback chub. Marsh (1985) reported that bonytail eggs are adhesive throughout the incubational period, which is thought to be an adaptive strategy to swift-moving currents of the mainstem Colorado River.

As stated previously, active spawning of a large (approximately 500 individuals) aggregate of bonytail in Lake Mohave was observed spawning over angular rock and gravel substrates near shore and in water up to 30 feet deep. Eggs were described as adhesive, and one individual female contained over 10,000 eggs, suggesting a high level of fecundity, a trait that appears to be typical for Colorado River endemic species (FWS 2002b). Even higher levels of fecundity were found in a hatchery, with individual egg production averaging over 25,000 eggs per female (Hamman 1982). Spawning bonytail in Cibola High Levee Pond were observed using shoreline riprap materials, typically in mid-April, frequently during nighttime hours, and in water temperatures ranging from 20.4-21.6 °C (68.7–70.9 °F) (Mueller et al. 2003).

Bonytail egg survival is highly influenced by incubation temperature. Hamman (1982) found 90 percent survival at water temperatures of 20–21 °C (68–70 °F), 55 percent survival at 16–17 °C (61–63 °F), and only 4 percent survival at temperatures of 12-13 °C (54-55 °F). Incubation periods ranged from 99 hours to nearly 500 hours, depending upon water temperatures. Newly hatched fry averaged 6.8 mm (Hamman 1982). This research is corroborated by Marsh (1985), who found that bonytail embryos have the highest survival rates at temperatures near 20 °C (68 °F) and indicated that newly hatched larvae averaged 6.0–6.3 mm in size. In summary, the literature and hatchery evidence show that strong reproductive output and production of young are key characteristics of this species in the absence of predators.

The bonytail's diet is reportedly comprised of a wide variety of aquatic and terrestrial insects, worms, algae, plankton, and plant debris (Mueller and Marsh 2002). This information corroborated earlier findings by McDonald and Dotson (1960) and Vanicek (1967), who also found that Colorado River chub species (including bonytail) fed omnivorously. More detailed and quantitative descriptions of the bonytail diet, including differences in stomach composition by life stage, are limited. However, bonytail stocked into Cibola High Levee Pond fed omnivorously, with adults consuming algae, vegetative material, small fish, and crayfish (various species), while the young fed near the pond surface, consuming zooplankton and invertebrates. Bonytail appear to be cannibalistic as they were observed consuming their own gametes, as well as young razorback sucker larvae (Mueller et al. 2003).

Colorado Pikeminnow

The Colorado pikeminnow is the largest freshwater fish endemic to the Colorado River System and is the largest member of the minnow family (Cyprinidae) native to North America. Historically, individuals estimated to weigh over 100 pounds and nearly 6 feet in length were reported from the Lower Basin (Seethaler 1978). The Colorado pikeminnow was included in the 1967 List of Endangered Species (32 Federal Register 4001; March 11, 1967) and encompassed in the protection of the ESA in 1973. A Recovery Plan was first developed in 1978, revised in 1989 (Upper Colorado River Basin Coordinating Committee 1989), and amended and supplemented by Recovery Goals in 2002 (FWS 2002c). The most recent recovery plan was approved in September 2023 (FWS 2023). A Species Status Assessment was completed in March 2020 (FWS 2020). The most recent 5-Year Status Review (FWS 2020) concluded that the Colorado pikeminnow is currently in danger of extinction throughout all its range because of the cumulative effects of threats, despite ongoing activities to address threats throughout the species' current distribution.

Critical habitat was designated in 1994 (Federal Register Vol. 59, No. 54, 13374-13399, March 21, 1994) and includes 1,148 miles of the Colorado River System. Within the analysis area, critical habitat includes the Colorado River and its 100-year floodplain downstream to North Wash near Hite Marina (RM 168; river miles upstream of Lees Ferry), including the Dirty Devil arm of the Lake Powell inflow. It also includes the San Juan River and its 100-year floodplain downstream to the mouth of Neskahai Canyon. At full pool elevation (3700 feet), Lake Powell inundates the lower 34 miles of critical habitat on the Colorado River (Imperial Canyon to North Wash, RM 202-168) and the lower 34 miles on the San Juan River (Clay Hills to Neskahai Canyon). A reservoir elevation below 3603 feet exposes all critical habitat upstream of North Wash on the Colorado River, and an elevation below 3600 feet exposes all critical habitat upstream of Neskahai Canyon on the San Juan River. Exposing the river channel through critical habitat may be beneficial to the Colorado pikeminnow, as historic riverine habitat provides the lotic environment preferred by the species. There is no critical habitat designated for Colorado pikeminnow in the Lower Basin.

Distribution and Abundance

Colorado pikeminnow historically occurred throughout the warmwater reaches of the Colorado River System, including the Green, Colorado, and San Juan River subbasins in Wyoming, Colorado, Utah, and New Mexico; downstream through the Colorado River mainstem in Arizona, Nevada, California, and Mexico; and the Gila River subbasin in Arizona and New Mexico. Colorado pikeminnow were largely extirpated from the Lower Basin by the 1960s following extensive dam construction, water development, erosive land use practices, and widespread introductions of nonnative fish (Miller 1961). In the Upper Basin, construction of large dams and diversions was more diffuse, leaving longer reaches of river available for habitat and unimpeded movement. Still, Colorado pikeminnow populations in the Upper Basin occur in contracted ranges and reduced abundances. Biochemical genetics show evidence of multiple stocks of Colorado pikeminnow in tributaries of the Upper Basin, with genetic panmixia indicating constant interchange of genes among the stocks or subpopulations (Morizot et al. 2002). This hypothesis of multiple interconnected stocks is supported by long-distance movements and strong fidelity of adults in contemporary subpopulations to particular tributaries and reflects the development of powerful selection mechanisms that evolved over thousands of years (Irving and Modde 2000; Tyus 1990). Hence, the contemporary populations in the Green River, Upper Colorado River, and San Juan River subbasins are likely derived from stocks with fidelity to those subbasins but with interchange of individuals.

The Colorado pikeminnow is presently found as three populations in the Upper Basin, including wild self-sustaining populations each in the Green River and Upper Colorado River subbasins, and a wild population sustained by annual hatchery stocks in the San Juan River subbasin (FWS 2023). About six million young Colorado pikeminnow were stocked from hatcheries into the San Juan River from 1994 to 2020, where the species is reproducing but with limited recruitment. Colorado pikeminnow are not stocked in the Green River or Upper Colorado River. The most recent estimates of abundance are 429 adults (95 percent confidence interval [CI], 334-561) for the Colorado River, and 885 (95 percent CI, 679-1,171) for the Green River for 2015 and 2018, respectively (FWS 2022, 2023). The most recent estimate for the San Juan River is 214 adults (95 percent CI, 162-320) for 2022, including wild and stocked fish (Schleicher et al. 2023). All ages of fish are naturally produced in the Green River and Upper Colorado River populations, and the fish in the San Juan River are comprised of a mix of naturally produced fish and stocked fish.

Colorado pikeminnow were likely extirpated from the Lower Basin by the mid-1970s, and during 1981–1990, over 623,000 juveniles were reintroduced into the Salt and Verde rivers, tributaries of the Gila River subbasin in Arizona (Hendrickson 1993), as a nonessential experimental population under Section 10(j) of the ESA. Survival of these fish was low, with few persisting, and the AZGFD stocked the remaining Colorado pikeminnow from their Bubbling Ponds Hatchery in 2018, with no plans to continue stocking into the future. With low survival of stocked fish and a lack of subsequent captures, this experimental population is considered functionally extirpated (FWS 2020). In a separate effort, the FWS recently assessed the potential recovery viability for Colorado pikeminnow in the Colorado River in Grand Canyon with the intent of stocking fish experimentally (Dibble et al. 2023), but no further action has been taken.

Colorado pikeminnow have not been documented downstream of Glen Canyon Dam since the mid-1970s (FWS 2020), and the last specimen reported from the Grand Canyon was in 1968 (Behnke 1973). In the first fish surveys of the Colorado River through Glen Canyon before dam construction, McDonald and Dotson (1960) reported only one immature Colorado pikeminnow, and in post-dam surveys, the species was considered rare (Stone and Rathbun 1968). In Lake Powell, Colorado pikeminnow have been captured in small numbers and occur as extensions of populations in the Upper Colorado River and San Juan River subbasins (FWS 2020).

The diversion channels on Glen Canyon Dam closed in March 1963 and began forming Lake Powell. The reservoir reached a full capacity of 3,700 feet elevation on June 20, 1980, but continued to rise to a maximum level of 3,708.34 feet above sea level on July 14, 1983 (Ferrari 1988). At maximum elevation, the reservoir inundated approximately 187 miles of Glen Canyon and Cataract Canyon. As the reservoir was filling, Colorado pikeminnow were caught occasionally in nets for monitoring the fish in Lake Powell. In the spring of 1980, 45 adult Colorado pikeminnow were captured in the Colorado River inflow near Gypsum Canyon (RM 197) when the reservoir was near full pool (Gustaveson et al. 1985). Some of these fish were radio-tagged and later detected in the Colorado River near Grand Junction, Colorado, over 200 miles upstream (Kaeding and Osmundson 1989).

For the Colorado River inflow to Lake Powell, Valdez (1990) sampled Cataract Canyon from Mineral Bottom to Palmer Canyon (RM 195) in 1984-1989 and found 111 larval and YOY Colorado pikeminnow (3 percent of 4,161 total captured) in the Lake Powell inflow downstream of the Big Drops Rapid in Cataract Canyon. These fish were captured in backwaters that had formed in silty deltaic sediments as Lake Powell was receding. Continued sampling of the inflow from 1990 to 1994 (except for 1992) yielded an additional 219 larval and YOY Colorado pikeminnow (Valdez and Cowdell 1994). In 1986, the Interagency Standardized Monitoring Program began, and through that program, YOY Colorado pikeminnow were sampled in the Upper Basin but did not extend into Cataract Canyon or the Lake Powell inflow. The Colorado River inflow to Lake Powell is sampled as an occasional extension of humpback chub monitoring in Cataract Canyon (sampled in 2021, but no Colorado pikeminnow were reported; Ahrens 2022), and the numbers and distribution of Colorado pikeminnow in the inflow are not unknown.

The San Juan River Basin Recovery Implementation Program was established in 1992 to promote recovery of the endangered Colorado pikeminnow and razorback sucker while allowing water development to continue. Surveys to monitor larval fish have been conducted as part of the San Juan River Basin Recovery Implementation Program since 1998 between RM 147.9 (Shiprock, New Mexico) and RM 2.9 (Clay Hills Crossing, Utah) (Farrington et al. 2023). Small-bodied fish monitoring has been conducted from RM 180.6 (Farmington, New Mexico) downstream to RM 3.1 (Clay Hills Crossing, Utah) (Barkalow and Zeigler 2022).

A large waterfall referred to as the Piute Farms Waterfall formed in the transition zone between the San Juan River and Lake Powell and has served as an impassable upstream barrier to fish since 2001, except once for 2 weeks in late July and mid-August in 2011 (Cathcart et al. 2018). During monitoring conducted from 2002 through 2005, 22 YOY Colorado pikeminnow were collected between Clay Hills and Lake Powell from November 2004 to November 2005 (Golden et al. 2006). In 2005 and 2006, 287 and 256 juvenile Colorado pikeminnow were captured, respectively, with only two collected downstream of the waterfall (Jackson 2006; Elverud and Jackson 2007). Studies of stocked Colorado pikeminnows in the San Juan River from 2002 through 2005 (Golden et al. 2006) collected 22 young between Clay Crossing and Lake Powell from November 2004 to November 2005. These data indicate that in the early 2000s, small numbers of Colorado Pikeminnow were present near the waterfall (upstream and downstream) when Lake Powell elevations were below 3,666.5 feet.

In 2015-2017, 24 Colorado pikeminnow were captured downstream of the waterfall; most were subadults except for one adult in 2016 (Cathcart et al. 2018). In 2023, three juveniles were captured in the San Juan River arm of Lake Powell near Piute Canyon (RM -34.1) and Neskahai Canyon (RM ~32.0; Schleicher et al. 2024). Two of the fish had been stocked in McElmo Creek on October 25, 2022, about 10 miles upstream of the confluence with the San Juan River and moved over 110 miles downstream to and past the waterfall, while the third fish was untagged (Schleicher et al. 2024). The San Juan River inflow to Lake Powell, like the Colorado River inflow, is periodically sampled.

Hence, the numbers and abundances of Colorado pikeminnow in these inflows are not known with certainty.

Habitat

Colorado pikeminnow inhabit large warm rivers and tributaries of the Colorado River System and require uninterrupted stream passage for spawning migrations and dispersal of young. The species is adapted to a hydrologic cycle characterized by large spring peaks of snowmelt runoff and low, relatively stable base flows (Bestgen 2018; Elverud et al. 2020). High spring flows create and maintain in-channel habitats and reconnect floodplain and riverine habitats. Through most of the year, juveniles, subadults, and adults use relatively deep, low-velocity pools, large eddies, and deep runs that occur in nearshore areas of main river channels (Tyus and McAda 1984; Valdez and Masslich 1989; Tyus 1990, 1991; Osmundson et al. 1995). In spring, Colorado pikeminnow adults use floodplains, flooded tributary mouths, flooded side canyons, and eddies that are available only during high flows (Tyus 1990, 1991; Osmundson et al. 1995). Such environments may be particularly beneficial to Colorado pikeminnow as riverine fish that gather in floodplain serve as prey for the pikeminnow. Such low-velocity environments also provide resting areas for adults and subadults.

Adults are potadromous (highly migratory within a river basin) and move over 600 miles to rocky canyon-confined spawning areas in summer (late June and July) following spring runoff. Spawning takes place in low velocity runs less than 1 meter (m) deep on cobble, gravel substrate (Tyus and McAda 1984; Tyus 1985, 1990, 1991; Irving and Modde 2000). After hatching and emerging from the spawning substrate, Colorado pikeminnow larvae drift passively downstream to backwaters in sandy, alluvial regions, where they remain for most of their first year of life (Holden 1977; Tyus and Haines 1991; Muth and Snyder 1995; Bestgen and Hill 2016a).

Colorado pikeminnow larvae occupy in-channel backwaters soon after hatching. The greatest number of larvae occurs in backwaters that are large, warm, deep (average, about 1 foot), and turbid (Tyus and Haines 1991; Bestgen and Hill 2016a). Such backwaters are created when a secondary channel is cut off at the upper end and remains connected to the river at the downstream end. These chute channels are deep and may persist even when discharge levels change dramatically. An optimal river-reach environment for growth and survival of early life stages of Colorado pikeminnow has warm, relatively stable backwaters, warm river channels, and abundant food (Muth et al. 2000). Juveniles and adults use deep, low velocity eddies, pools, and runs, but move into flooded habitats and bottomlands during spring runoff (Tyus and McAda 1984; Valdez and Masslich 1989; Tyus 1990, 1991; Osmundson et al. 1995).

Life History

The Colorado pikeminnow is a large piscivorous minnow that spawns annually in spring, starting at 5-7 years of age and may live up to 50 years (FWS 2020). The species is adapted to warm rivers and requires uninterrupted passage for spawning migration and larval drift, and a hydrologic cycle characterized by large spring peaks of snowmelt runoff and low, relatively stable base flows (Bestgen et al. 2018; Elverud et al. 2020). Adults are potadromous and may move up to 590 miles to and from spawning sites in summer (Tyus and McAda 1984; Tyus 1990; Irving and Modde 2000).

Archaeological remains from several sites along the Lower Colorado River and its tributaries indicate that the species was historically widespread, abundant, and the largest fish in the river (Seethaler 1978; Miller 1955). The largest of six specimens from Catclaw Cave (now inundated by Lake Mohave), dating from 1100 A.D., was at least 1.7 m long (5.6 feet) (Miller 1955). Early expeditions of the Lower Basin provided the type specimen (Girard 1856) and reports of fish frequently taken near Yuma, Arizona, reaching lengths of 1.2-1.5 m (4-5 feet) (Gilbert and Scofield 1898) and *"…fish would easily weigh 100 lb. (45 kg)"* (Chamberlain 1904). The largest confirmed weights are 27 and 34 pounds for two fish from Lake Mead (Wallis 1951). The largest Colorado pikeminnow recently caught was about 930 mm TL from the Upper Colorado River (Elverud et al. 2020), and about 900 mm TL from the Green River (Bestgen et al. 2018); these fish are estimated to weigh 25-27 pounds. Contemporary sizes of adult Colorado pikeminnow are not as large as estimated for historical archaeological specimens and accounts from the Lower Basin, where the species is now extirpated. Growth and size in the Upper Basin appear limited by colder temperatures and a shorter growing season (Kaeding and Osmundson 1989). Bulkley et al. (1981) estimated the thermal preference of juvenile and adult Colorado pikeminnow as 24.6 ºC (76.3 °F) and 25.4 ºC (77.7 °F), respectively. Mean temperature of the Colorado River near Cisco, Utah, from 2006-2023 for July, which is the warmest month, is 23.6 ºC (74.5 °F) (United States Geological Survey Surface-Water Monthly Statistics for Utah), which is slightly below the preferred temperature of Colorado pikeminnow. Young Colorado pikeminnow have very specific streamflow and temperature requirements. Kaeding and Osmundson (1989) correlated lower water temperatures with reduced growth of age-0 Colorado pikeminnow and concluded that fish 45-100 mm TL would not grow at water temperatures below 13 ºC (55 °F).

Spawning of Colorado pikeminnow occurs after spring runoff (typically late June through August) at water temperatures of 18-23 °C (64–73 °F) (Vanicek and Kramer 1969; Hamman 1981). Females are broadcast spawners that attract multiple males to fertilize eggs in the water column. Average fecundity is about 66,000–77,000 eggs/female (Hamman 1986), and females broadcast adhesive eggs over cobble bars where the eggs incubate 90–121 hours at 20–24 °C (68–75 °F) (Hamman 1981; Marsh 1985). The larvae emerge from the substrate after about 5 days and drift up to 125 miles to nursery backwaters where survival is critical to recruitment (Holden 1977; Tyus and Karp 1989; Haines and Tyus 1990; Tyus 1991; Tyus and Haines 1991; Bestgen et al. 1997, 1998; Converse et al. 1999).

Nursery areas consist of ephemeral backwaters and shoreline embayments with little or no current (Tyus and Haines 1991). Bestgen (1996) reported that Colorado pikeminnow larval growth rate declines at water temperatures below 22 ºC (72 °F). As summarized in Valdez (2006 and 2014), spawning begins after spring runoff when water temperatures reach at least 16 °C, with reproduction often initiated between 16–23°C depending on the river system. Their adhesive eggs are scattered over cobble substrates and incubate in interstitial spaces, with laboratory studies showing highest hatching success at temperatures of 20–24°C and incubation times of 90–121 hours. In the wild, eggs typically incubate for about five days before hatching. The mean size of Colorado pikeminnow larvae is 7.7 mm TL, and they transition to juveniles at 35.2 mm TL, or at about 60 days of age (Snyder 1981). In the Green and Upper Colorado rivers, most of the juveniles remain in shallow, warm, productive backwaters through summer, fall, and winter, and are flushed from these habitats with the following spring runoff (Valdez and Cowdell 1994). In the San Juan River, the larvae use zero velocity habitats, including backwaters and embayments, and juveniles remain in available low velocity habitats (Barkalow and Zeigler 2022).

Adult Colorado pikeminnow are primarily piscivores starting in their first year of life, although they lack jaw, palatine, or vomerine teeth. Instead, they have pharyngeal teeth, which are modified gill rakers that process and masticate swallowed food items passing through their gullet (FWS 2002c). Young Colorado pikeminnow consume zooplankton and midge (chironomid) larvae (Vanicek 1967; Jacobi and Jacobi 1982; Muth and Snyder 1995), and piscivorous juveniles, subadults, and adults eat soft-rayed native and nonnative fish (Osmundson 1999), as well as a variety of insects and animals, including Mormon crickets (*Anabrus migratorius*) (Tyus and Minckley 1988), mice, birds, and rabbits (Beckman 1963).

Special Status Species

In addition to threatened and endangered species, the Colorado River also supports special status fish species that are not currently classified as threatened or endangered but are of conservation concern or have significant ecological importance. These species include flannelmouth sucker (*Catostomus latipinnis*), bluehead sucker (*Pantosteus discobolus*), and speckled dace (*Rhinichthys osculus)*, as described below.

Flannelmouth Sucker

The flannelmouth sucker is a medium to large river sucker (Catostomidae). It has a maximum TL greater than 600 mm (AZGFD 2001b; Rees et al. 2005). It is a long-lived species that lives up to 30 years (AZGFD 2001b). The flannelmouth sucker is categorized by the International Union for Conservation of Nature (IUCN) as a species of least concern and not a focus of wildlife conservation because it is still plentiful in the wild. The species does not qualify as federally endangered, threatened, or conservation dependent. It is a Bureau of Land Management (BLM)-sensitive species in Utah and Arizona. The flannelmouth sucker is included in the Range-Wide Conservation Agreement and Strategy that has developed and implemented conservation measures in lieu of federal listing (UDWR 2006).

Distribution and Abundance

The flannelmouth sucker is endemic to the Colorado River System. Historically, the species ranged throughout the Colorado River in moderate to large rivers in Arizona, California, Colorado, Nevada, New Mexico, Utah, and Wyoming (Bezzerides and Bestgen 2002; Rees et al. 2005). Within the analysis area, flannelmouth sucker are found in Lake Powell mostly in the Colorado River and San Juan River inflows, they are abundant in the Colorado River between Glen Canyon Dam and Pearce Ferry, they are in Lake Mead and in tributary inflows, and they are absent downstream of Hoover Dam except for a small, reintroduced population in the Colorado River downstream of Davis Dam (Mueller and Wydoski 2004). Within the Virgin River, this species may be found in the mainstem river and its tributaries downstream to at least Halfway Wash, with specimens also being captured near the Virgin River inflow of Lake Mead, in lentic habitats (Kegerries et al. 2018; Rogers et al. 2024).

In the Grand Canyon, this species is also found in the mainstem Colorado River and its tributaries, including the Little Colorado and Paria Rivers and Shinumo, Bright Angel, Kanab, and Havasu Creeks (AZGFD 2001b; Bezzerides and Bestgen 2002; Douglas and Marsh 1998; Weiss 1993). In contrast to bluehead sucker, flannelmouth sucker are only found downstream of the barrier falls in Shinumo and Havasu Creeks. During annual monitoring conducted between 2000 and 2022, flannelmouth sucker were present in all reaches of the river between Lees Ferry and the inflow to Lake Mead (Makinster et al. 2010; Fonken et al. 2023). Abundance, across all reaches measured as catch-per-unit-effort, has been increasing since 2000, especially since 2004 (Makinster et al. 2010). However, abundance had been decreasing within the reach between RM 0 and RM 179 since about 2005, while increasing downstream of RM 179. Surveys of the small-bodied and larval fish communities in the western Grand Canyon (Lava Falls to Pearce Ferry) found flannelmouth sucker present in the western Grand Canyon, accounting for over 38 percent of the total larval catch in this area (Albrecht et al. 2014).

Abundance estimates using monitoring data and age-structured mark-recapture models show an increase in the abundance of age-1 (juvenile) and age-4 (adult) flannelmouth suckers in the Grand Canyon between 2000 and 2008 (Wolters et al. 2012). Abundance of age-1 flannelmouth sucker increased from about 2,500 in 2000 to about 10,000 in 2008, while abundance of age 4+ adults increased from about 10,000 to about 25,000 for this same period (Wolters et al. 2012). Other abundance estimates based on electrofishing catch-per-unit-effort for this same time period showed an increase in abundance from less than 1,000 in 2000 to about 12,000 in 2009, while the estimated abundance of age-4+ adults increased from about 2,500 in 2001 to about 31,000 in 2009 (Wolters et al. 2012). Long-term fish monitoring by AZGFD (Rogowski et al. 2018) shows flannelmouth sucker in the Colorado River from Lees Ferry to Pearce Ferry increased slowly in catch-per-unit-effort from 2000 to 2011, followed by a general decline to 2014 and an increase to 2017. Based on flannelmouth sucker recaptures in 2002, Mueller and Wydoski (2004) estimated the Reach 3 population to be 2,286 individuals (95 percent confidence interval 1,847–2,998). From 2006 through 2010, estimates ranged from 1,044–2,171 (Best and Lantow 2012). The relative abundance of adult flannelmouth sucker remained high through 2023 (Rogowski et al. 2023).

Habitat

The flannelmouth sucker is generally found in moderate to large rivers. Adults prefer deep water when not feeding (Rinne and Minckley 1991), while larvae and young are often associated with shallow, slow-moving nearshore areas such as backwaters and shoreline areas of slow runs or pools (AZGFD 2001b; Rees et al. 2005). Although it is a riverine species, the flannelmouth sucker has been collected from the inflow areas of Lake Mead (Rogers et al. 2024). In the Virgin River, both adults and subadults are found in low-velocity, nearshore habitats with large amounts of cover, such as eddies and runs over sandy bottoms. Juveniles and adults may be considered habitat generalists and can be found using pool, run, and eddy habitats.

The flannelmouth sucker has been collected from Flaming Gorge and Fontenelle Reservoirs of the Upper Basin. In the Colorado River in the Grand Canyon, subadults are found in eddies and runs over sand bottoms. In the Little Colorado River, adult and juvenile flannelmouth sucker use low-velocity, nearshore habitats with large amounts of cover during the daylight, and faster, more exposed midchannel habitats at night (Gorman 1994). Juveniles and adults may be considered habitat generalists and can be found using pool, run, and eddy habitats. Surveys of larval flannelmouth sucker in western Grand Canyon (from Lava Falls to Pearce Ferry) found the highest abundance of larvae in embayments, isolated pools, backwaters, and other low-velocity habitats (Albrecht et al. 2014).

Life History

The flannelmouth sucker is a spring broadcast spawner, where females release eggs into the water column to be fertilized by multiple males. The semi-adhesive eggs fall into crevices of cobble and gravel substrate where they incubate for about 5 days and emerge as larvae 14-16 mm TL. The larvae drift passively downstream into low-velocity nursery areas.

Flannelmouth sucker prefer water temperatures ranging from 10–27 °C (50–81 °F) and appear to prefer temperatures of about 26 °C (79 °F) (Sublette et al. 1990). Water temperatures reported during spawning activity range from 6–18.5 °C (43–65 °F) but are usually above 14 °C (57.2 °F) (Bezzerides and Bestgen 2002). In the Lower Basin, flannelmouth sucker typically spawn in March and April (Bezzerides and Bestgen 2002). Water temperature has been suggested as a primary cue for spawning in other parts of this species' range, but it does not appear to provide a spawning cue in the Grand Canyon, where relatively synchronized spawning has been reported among sucker stocks from creeks with different temperature and flow regimes (Weiss 1993; Weiss et al. 1998). In the Paria River, the timing of spawning has been correlated with the receding limb of the hydrograph (Weiss 1993).

In the Grand Canyon, flannelmouth sucker apparently spawn in certain tributaries, and fish may move considerable distances to reach spawning sites (Douglas and Marsh 1998; Weiss et al. 1998; Douglas and Douglas 2000). Tributary spawning in the Grand Canyon may be timed to take advantage of warm, ponded conditions at tributary mouths that occur during high flows in the mainstem Colorado River (Bezzerides and Bestgen 2002). Valdez and Ryel (1995) reported large concentrations of bluehead sucker and flannelmouth sucker in tributary mouths through Grand Canyon in spring as presumed spawning runs.

The body condition of flannelmouth sucker is variable throughout the Grand Canyon, but is greatest at intermediate distances from Glen Canyon Dam, possibly because of the increased number of warmwater tributaries in this reach (Paukert and Rogers 2004). Mean condition peaks during the pre-spawn and spawning periods and is lowest in summer and fall (McKinney et al. 1999; Paukert and Rogers 2004). Sucker condition in September was positively correlated with Glen Canyon discharge during summer (June–August), possibly due to an increased euphotic zone and greater macroinvertebrate abundance observed during higher water flows (Paukert and Rogers 2004).

The flannelmouth sucker is characterized by lower lips that are noticeably bulky with thick and fleshy lobes that it uses to feed along river bottoms and around structures. The species is an omnivorous benthic feeder, foraging on invertebrates, algae, plant seeds, and organic and inorganic debris (Bezzerides and Bestgen 2002; Rees et al. 2005; Seegert et al. 2014). The larvae feed primarily on aquatic invertebrates, crustaceans, and organic debris (Childs et al. 1998). As they become juveniles and adults, their diet shifts and becomes primarily comprised of benthic matter, including organic debris, algae, and aquatic invertebrates (Rees et al. 2005; Seegert et al. 2014).

Bluehead Sucker

The bluehead sucker is a medium-sized river sucker (Catostomidae). Adults may reach 300-450 mm TL in large rivers but may be smaller in tributaries; they may live from 6 to 8 years to as many as 20 years (AZGFD 2003; Bezzerides and Bestgen 2002; Sigler and Sigler 1987). The bluehead sucker is categorized by the IUCN as a species of least concern and not a focus of wildlife conservation because it is still plentiful in the wild. The species does not qualify as federally endangered, threatened, or conservation dependent. It is a BLM-sensitive species in Utah and Arizona. The bluehead sucker is included in the Range-Wide Conservation Agreement and Strategy, which has developed and implemented conservation measures in lieu of federal listing (UDWR 2006).

Distribution and Abundance

The bluehead sucker is native to the Colorado River and its tributaries from Lake Mead upstream into Arizona, Colorado, New Mexico, Utah, and Wyoming. This species is also found in the Snake River (Idaho and Wyoming), the Bear River (Idaho and Utah), and Weber River (Utah and Wyoming) drainages (AZGFD 2003; Bezzerides and Bestgen 2002).

Within the analysis area, bluehead sucker are rare in Lake Powell, common in the Colorado River between Glen Canyon Dam and Pearce Ferry, in tributary inflows in Lake Mead, and absent downstream of Hoover Dam. In the Grand Canyon, the bluehead sucker is found in the Colorado River mainstem and its tributaries, including the Little Colorado River, Clear Creek, Bright Angel Creek, Kanab Creek, and Havasu Creek (AZGFD 2003; NPS 2013b; Ptacek et al. 2005; Rinne and Magana 2002). Prior to 2014, it was also found in Shinumo Creek but was largely displaced from that system by large debris flows (Healy et al. 2014). Annual fish monitoring during 2000—2024 in the Colorado River between Glen Canyon Dam and the inflow to Lake Mead shows the bluehead sucker present throughout the mainstem river (Makinster et al. 2010; Kegerries et al. 2015; Rogers et al. 2024). This species is rare in the upper sections of Grand Canyon National Park because of cold dam releases, and increases in number near the Little Colorado River inflow and downstream with warmer water temperatures (Bunch et al. 2012). Bluehead suckers are found more often in Grand Canyon National Park with warmer dam releases.

Within Lake Mead, the bluehead sucker is found periodically in the Colorado River inflow area in limited numbers. Annual native fish monitoring conducted between 2010 and 2024 at the Colorado River inflow area of Lake Mead has produced a total of four individuals (Rogers et al. 2024; BIO-WEST unpublished data). Most contacts occur within the flowing portions of the Colorado River upstream of Pearce Ferry (Rogers et al. 2023).

Abundance estimates using monitoring data and age-structured mark-recapture models show a high variability in the numbers of bluehead sucker over time and location. The abundance of age-1 (juvenile) bluehead sucker in the Grand Canyon appears to have declined from 1990 to 1995, increased from 1995 to 2003, and then declined through 2009 (Wolters et al. 2012). Similar estimates for age-4 (adult) fish show that abundance began increasing from the late 1990s until 2005 or 2006, after which abundance also declined. The estimated abundance of age-1 bluehead sucker has ranged from 1,000 or fewer to as many as 60,000 fish between 2000 and 2009 (Walters et al. 2012). The estimated abundance of age-4+ adults during this same period ranged from about 5,000 to as many as 75,000 fish. Relatively high numbers of individuals remain in the Lower Colorado River between

Lava Falls Rapid (RM 179) and Lake Mead (Bunch et al. 2012). Bluehead sucker larvae were reported throughout the western Grand Canyon between Lava Falls and Pearce Ferry during larval fish community monitoring throughout the area (Albrecht et al. 2014). In this area, the bluehead sucker was the most abundant species in the larval fish community, comprising almost 40 percent of the total catch. Larval fish sampling from Bright Angel Creek (RM 88.5) downstream to Pearce Ferry reported that approximately 26 percent of the larval fish were bluehead sucker (Rogers et al. 2023). Longterm fish monitoring conducted by AZGFD (Rogowski et al. 2018) indicates that bluehead sucker populations in the Colorado River from Lees Ferry to Pearce Ferry increased in catch-per-unit-effort between 2000 and 2010. After this period of growth, their numbers generally declined, with some year-to-year variability, but then rebounded to previous high levels in 2016 and 2017. More recent data show that the relative abundance of bluehead sucker has exhibited a downward trend with considerable fluctuations through 2022 (Fonken et al. 2023).

Habitat

The bluehead sucker typically inhabits rivers and large streams and may be found in smaller streams and creeks (AZGFD 2003; Sigler and Sigler 1987). Riverine habitats may range from cold (12 °C [54 °F]), clear streams to warm (28 °C [82 °F]), turbid rivers. Large adults live in deep water (6 to 10 feet), while juveniles use shallower, lower velocity habitats (Bezzerides and Bestgen 2002). In clear streams, bluehead sucker stay in deep pools and eddies during the day and move to shallower habitats (for example, riffles or tributary mouths) to feed at night, while in turbid waters they may use shallow areas throughout the day (AZGFD 2003; Beyers et al. 2001). In the Grand Canyon, larval and young bluehead sucker inhabit backwater areas and other nearshore, low-velocity habitats such as eddies, embayments, and isolated pools (Albrecht et al. 2014; AZGFD 2003; Childs et al. 1998). In Lake Mead, larval and young bluehead suckers inhabit slack water, backwater, and lentic habitats (Kegerries et al. 2015).

Life History

The bluehead sucker is a spring broadcast spawner, where females release eggs into the water column to be fertilized by multiple males. The semi-adhesive eggs fall into crevices of cobble and gravel substrate where they incubate for about 5 days and emerge as larvae 12-14 mm TL. The larvae drift passively downstream into low-velocity nursery areas. This species has been reported to be as large as 500 mm TL in the mainstem Colorado River in Grand Canyon, with tributary fish being smaller (AZGFD 2003; Valdez and Ryel 1995; Walters et al. 2012). A related subspecies, the Zuni bluehead sucker, occurs in the headwaters of the Little Colorado River along with the bluehead sucker that is the same subspecies as in the mainstem Colorado River (AZGFD 2002b).

Bluehead sucker spawn in spring and summer after water temperatures exceed 16 °C (61 °F). Valdez and Ryel (1995) reported large concentrations of bluehead sucker and flannelmouth sucker in tributary mouths through Grand Canyon in spring as presumed spawning runs. Spawning in Grand Canyon tributaries occurs mid-March through June in water depths ranging from a few inches to more than 3 feet and at temperatures of 16–20 °C (61–68 °F) over gravel-sand and gravel-cobble substrates (AZGFD 2003; NPS 2013b). In Kanab Creek, spawning has been reported to occur at temperatures of 18.2–24.6 °C (64.8–76.3 °F) (Maddux and Kepner 1988). Smaller tributaries may provide nursery areas for populations of large adjacent rivers (Rinne and Magana 2002). In Lake

Mead, larvae have been captured at temperatures of 18.2–24.6 °C (64.8–76.3 °F) at the Colorado River inflow monitoring site (Kegerries et al. 2015).

The bluehead sucker is an omnivorous benthic forager with a modified lower cartilaginous jaw used as a scraping radula. It feeds by scraping diatoms, detritus, algae, invertebrates, and other organic and inorganic material off rocks and other hard surfaces (Ptacek et al. 2005). Larvae drift to backwaters and other areas of low current, where they feed on diatoms, zooplankton, and dipteran larvae.

Speckled Dace

The speckled dace is native to the western United States and is one of eight species in the genus *Rhinichthys*. It is a small fish, typically less than 76 mm TL, and has a relatively short lifespan of about 3 years (Sigler and Sigler 1987). The speckled dace is categorized by the IUCN as a species of least concern and not a focus of wildlife conservation because it is still plentiful in the wild. The species does not qualify as federally endangered, threatened, or conservation dependent. It is a BLM-sensitive species in Arizona. Possession or importation of *Rhinichthys osculus* is prohibited or regulated in Colorado and Nevada.

Distribution and Abundance

The speckled dace is the most ubiquitous fish in the western U.S., ranging from the Columbia River drainage (north to southern British Columbia) to the southern Gila River drainage of Arizona and Sonora, Mexico (FWS 2024). The speckled dace is native to the western United States and is one of eight species in the genus *Rhinichthys*.

Although this species is the most widely distributed and abundant native fish species in the Grand Canyon ecosystem, its abundance and distribution could be affected by many of the same factors that affect the abundance and distribution of the other native fish in the ecosystem, namely altered temperature, flow, and sediment regimes and predation by nonnative fish (AZGFD 2002c; Gloss and Coggins 2005).

Within the analysis area, speckled dace are commonly found along shorelines in Lake Powell as well as in the inflows of the Colorado and San Juan Rivers. They are abundant in the Colorado River between Glen Canyon Dam and Pearce Ferry and are also present in Lake Mead and its tributary inflows. Downstream of Hoover Dam, speckled dace occur in smaller numbers. In the Grand Canyon, this species inhabits both the mainstem Colorado River and its tributaries, including the Little Colorado River (Makinster et al. 2010; Robinson et al. 1995; Ward and Persons 2006). Longterm fish monitoring of the Colorado River downstream of Glen Canyon Dam since 2000 shows the speckled dace is the third most common fish species (and most common native species) in the river between Glen Canyon Dam and the Lake Mead inflow, and it was captured most commonly in the western Grand Canyon and the inflow to Lake Mead (Makinster et al. 2010). More recently, Rogers et al. (2024) showed speckled dace were the second most common native species since 2014.

Within the Virgin River, this species occurs within the mainstem and its tributaries, including the Beaver Dam Wash Creek (Kegerries et al. 2018). Long-term fish monitoring of the Virgin River since 1996 shows the speckled dace to be the third most common fish species (and most common native species) in the lower Virgin River (Kegerries et al. 2018).

Habitat

The speckled dace is found in a variety of habitats, including large rivers, cold, fast-flowing mountain streams, ponds, and warm intermittent desert streams and springs (FWS 2024). It occurs in rocky runs, riffles, and pools of headwater streams, creeks, and small to medium rivers, typically in waters with depths less than 1.6 feet (AZGFD 2002c), and it rarely occurs in lakes (Page and Burr 1991). Valdez and Ryel (1995) reported finding speckled dace in gravel/cobble fans at arroyos and side canyons of the Colorado River through Marble and Grand Canyons.

Life History

The speckled dace is a small, short-lived minnow that spawns in spring. Adults mature at 2 years of age and are 100-130 mm TL. Spawning occurs in spring when a single female broadcasts 200-500 adhesive eggs over the gravel stream bed, and the eggs are fertilized by multiple males. Speckled dace are omnivorous, feeding on filamentous algae and other plant material, insect larvae and other invertebrates, as well as fish eggs (Seegert et al. 2014). Its young are mid-water plankton feeders (Sigler and Sigler 1987). Speckled dace can spawn twice a year, once in spring and again in late summer when water temperatures are 21–29 °C (AZGFD 2002c).

High-Risk Nonnative Fish Species

Nonnative fish species have been established in the Colorado River by various means for over a century. Starting in the mid to late 1800s, federal and state agencies distributed nonnative common carp (*Cyprinus carpio*) and channel catfish to rural communities along the Colorado River as a food source. Then, with dam construction in the early 1900s, state agencies began to introduce a variety of warm-water and cold-water species for recreational sports fisheries and as forage to support those fisheries. Changes in flow, temperature, sediment, turbidity, salinity, and physical and biological habitat components altered the river's ecological functions and the collective fish communities (Miller 1961). Understanding the past and present condition and status of fish in the analysis area of this Draft EIS is essential to understanding the potential impacts of proposed actions. The following are descriptions of nonnative fish species found in the analysis area.

Rainbow Trout

The rainbow trout is a member of the Salmonidae family, and the life history of this species is extremely variable, depending on location, environmental conditions, and genetic strain (Scott and Crossman 1973). It is one of the most popular and widespread cold-water fish in North America. It is native to cold-water tributaries of the Pacific Ocean in North America and is extensively cultured in hatcheries for stocking as a sportfish throughout the United States. Angling for rainbow trout is regulated by state game and fish agencies that limit the number of fish taken daily. It is the main sportfish in the 15-mile Lees Ferry sub-reach designated by the AZGFD as a Blue-Ribbon Rainbow Trout Fishery. The species can attain lengths ranging from 30 to 60 cm (12 to 24 inches) and can reach weights of 1.5 to 3 kg (3.3 to 6.6 pounds), although larger individuals have been recorded.

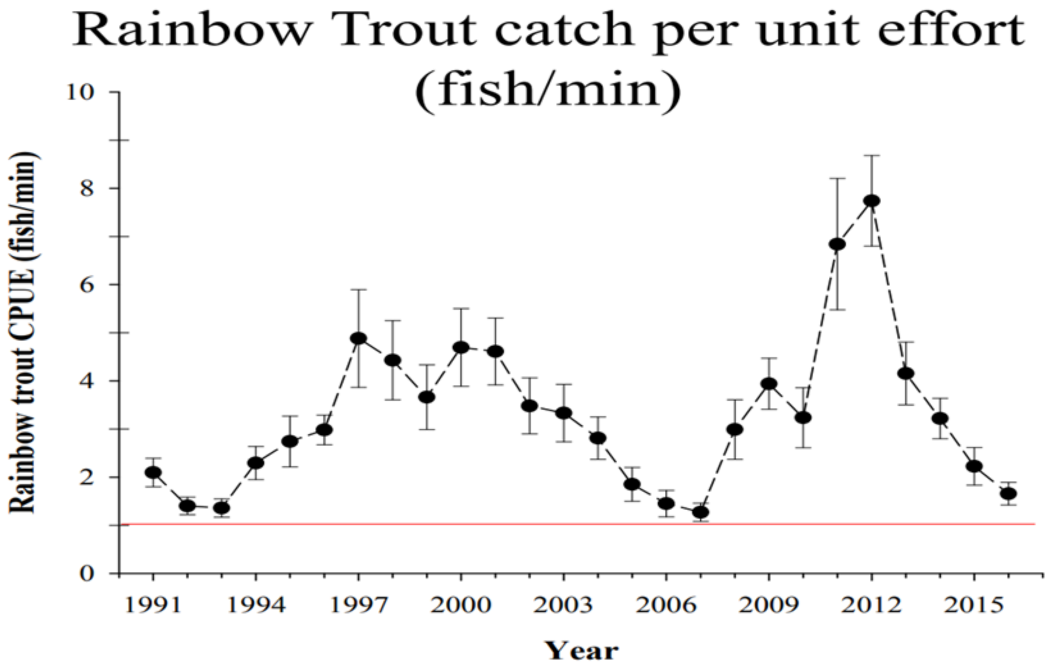
Distribution and Abundance

The rainbow trout was originally introduced into tributaries of the Grand Canyon in 1920-1940 by NPS, including Bright Angel Creek, Tapeats Creek, Havasu Creek, Clear Creek, and Phantom Creek (Runge et al. 2018). With the completion of Glen Canyon Dam in 1963 and the filling of Lake Powell, hypolimnetic dam releases provided a cold, clear, and productive environment for trout in the tailwater. The AZGFD, in cooperation with NPS, began to stock cold-water species in the Lees Ferry sub-reach to establish a recreational fishery. Rainbow trout were stocked in this reach in 1964–1998, and the fish survived successfully, but high fluctuating dam releases precluded most natural reproduction. In 1991, dam operating criteria were modified to increase minimum flows and reduce daily variability in discharge associated with hydropower production. These changes in dam operation allowed the rainbow trout to reproduce and recruit naturally, and stocking the species in the Lees Ferry sub-reach ceased in 1998. Except for localized spawning in and near the confluence of some downstream tributaries (for example, Nankoweap, Clear, Bright Angel, Shinumo, Tapeats, Deer, and Havasu Creeks), most rainbow trout reproduction in the mainstem Colorado River downstream of Glen Canyon Dam occurs within the Lees Ferry sub-reach. Rainbow trout have also been stocked into Lake Mead, Lake Mohave, and Reach 3 of the Lower Colorado River.

Rainbow trout recruitment and population size within the Lees Ferry sub-reach appear to be largely driven by dam operations (McKinney et al. 1999; McKinney et al. 2001; Makinster et al. 2011; Wright and Kennedy 2011; Korman et al. 2011; Korman et al. 2012). The decline in abundance observed in rainbow trout from 2001 to 2007 (**Figure TA 8 Attachment 1-8**) was attributed to the combined influence of increased trout metabolic demands due to warmer water releases from Glen Canyon Dam, together with a static or declining food base, periodic dissolved oxygen deficiencies, and high numbers of the invasive New Zealand mud snail (*Potamopyrgus antipodarum)* serve as a poor food source for fish (Cross et al. 2011).

From 2008 through 2010, the relative abundance of rainbow trout in the Lees Ferry sub-reach increased to near historic high levels (**Figure TA 8 Attachment 1-8**). The highest catch rates in 2011 and 2012 were likely because of warmer releases from Glen Canyon Dam and an input of nutrients from a changed reservoir elevation of Lake Powell (Rogowski et al. 2023). This was after a fall steady flow experiment and equalization flows in 2011 that boosted food base production in the Lees Ferry sub-reach and contributed to greater growth and an increase in abundance of rainbow trout. The population declined tenfold by 2016 due to a combination of lower recruitment and reduced survival of larger trout, which were likely driven by changes in nutrients and invertebrate prey availability (Yard et al. 2023). Survival rates for trout 8.8 inches or longer in 2014, 2015, and 2016 were 11 percent, 21 percent, and 22 percent, respectively, lower than average survival rates between 2012 and 2013. Abundance would have been threefold to fivefold higher had survival rates for larger trout remained at the elevated levels estimated for 2012 and 2013.

Figure TA 8 Attachment 1-8
Mean (±2 Standard Error) Electrofishing Catch Rates of Rainbow Trout in the Glen Canyon Reach, 1991–2016



Source: Rogowski et al. 2023.

Growth declined between 2012 and 2014 due to reduced prey availability, which led to poor fish condition by fall of 2014 (approximately 0.9–0.95, fish condition). Poor condition in turn resulted in low survival rates of larger fish during the fall of 2014 and winter of 2015, which contributed to the population collapse. Korman et al. (2012) found that recruitment of rainbow trout in Lees Ferry was positively and strongly correlated with annual flow volume and reduced hourly flow variation, and that recruitment increased after two of three high-flow releases related to implementation of equalization flows. In Glen Canyon, large recruitment events driven by high flows can lead to increases in the population that cannot be sustained due to limitations in prey supply. In the absence of the ability to regulate prey supply, flows that reduce the probability of large recruitment events can be used to avoid boom-and-bust population cycles (Korman et al. 2017).

Habitat

Adult rainbow trout in streams typically use deep pools, eddies, runs, and overhanging banks for resting, and they feed in open riffles. Juveniles use shallow habitats with instream and overhead cover, with recently hatched fry using shallow shorelines (Sigler and Sigler 1996). In the Lees Ferry reach of Grand Canyon, Korman et al. (2011) found the proportion of the age-0 population in low-angle shorelines, which are potentially more sensitive to flow variability, declined from 70 percent in June to 20 percent in November as fish grew and made an ontogenetic habitat shift to deeper habitat, often along steep angle shorelines, which were less sensitive to flow variability. This study demonstrated how flow variability from daily hydropower production can affect age-0 rainbow trout along low-angle shorelines.

Life History

The rainbow trout can reach a maximum size of nearly 50 pounds and typically matures between 3 to 5 years, with females producing 500-3,161 eggs in streams and 935-4,578 eggs in lakes (Sigler and Sigler 1996). Females prepare individual nests or redds in gravel riffles or runs where they deposit eggs that are fertilized by multiple males. Spawning typically takes place in spring at 10–15 ºC, although spawning in the Colorado River downstream of Glen Canyon Dam can occur as early as November and as late as June, with a peak in redd counts from late February to early March (Avery et al. 2015). A strong seasonal pattern in relative condition was seen for rainbow trout in Lees Ferry and was driven by seasonal changes in growth in length and weight (Yard et al. 2015). Body condition was usually at a minimum in January and increased rapidly in spring to attain a peak by July. The decline in condition from July to September was relatively small in clear conditions compared with the other reaches influenced by high levels of turbidity (Korman et al. 2016).

The rainbow trout has a variable diet and will forage throughout the water column on various items (Sigler and Sigler 1996), depending on the fish's genetic strain. In general, the fish's diet changes with the seasons, and generally the species forages near the bottom or near structure (Scott and Crossman 1973). In smaller lake systems, rainbow trout are primarily insectivorous, foraging on Dipetera (e.g., chironomids), Ephemoptera (mayflies), Plecoptera (stoneflies), Crustacea (e.g., amphipods such as *Gammarus* sp.), etc. (Scott and Crossman 1973).

Brown Trout

The brown trout is native to Europe, North Africa, and Western Asia, and it has been introduced as a sportfish into streams, rivers, and lakes around the world. The species was first imported into the United States from Germany in 1883 by the U.S. Fish Commission (Mather 1889). Since then, the species has been stocked in virtually every state of the country, including the seven states of the Colorado River System. It was first introduced into Grand Canyon National Park in 1919 by NPS to provide recreational fishing opportunities for park visitors (Runge et al. 2018). Brown trout were introduced into Shinumo Creek, Garden Creek, and Bright Angel Creek in the 1920s and 1930s (Carothers and Minckley 1981), and they were last stocked in the Grand Canyon in Bright Angel Creek in December of 1934, where a reproducing population became established (Runge et al. 2018).

The brown trout is one of four nonnative fish species in Glen Canyon National Recreation Area and Grand Canyon National Park identified by NPS as a species presenting a "(1) Very High" level of threat to native fish populations (NPS 2021). The other species with this threat level include smallmouth bass, walleye, and flathead catfish (*Pylodictis olivaris)*. With warmer releases from Glen Canyon Dam as seen in 2004, brown trout can disperse and pose a potential threat to rainbow trout in the Lees Ferry reach and to humpback chub in downstream aggregations (Runge et al. 2018).

Distribution and Abundance

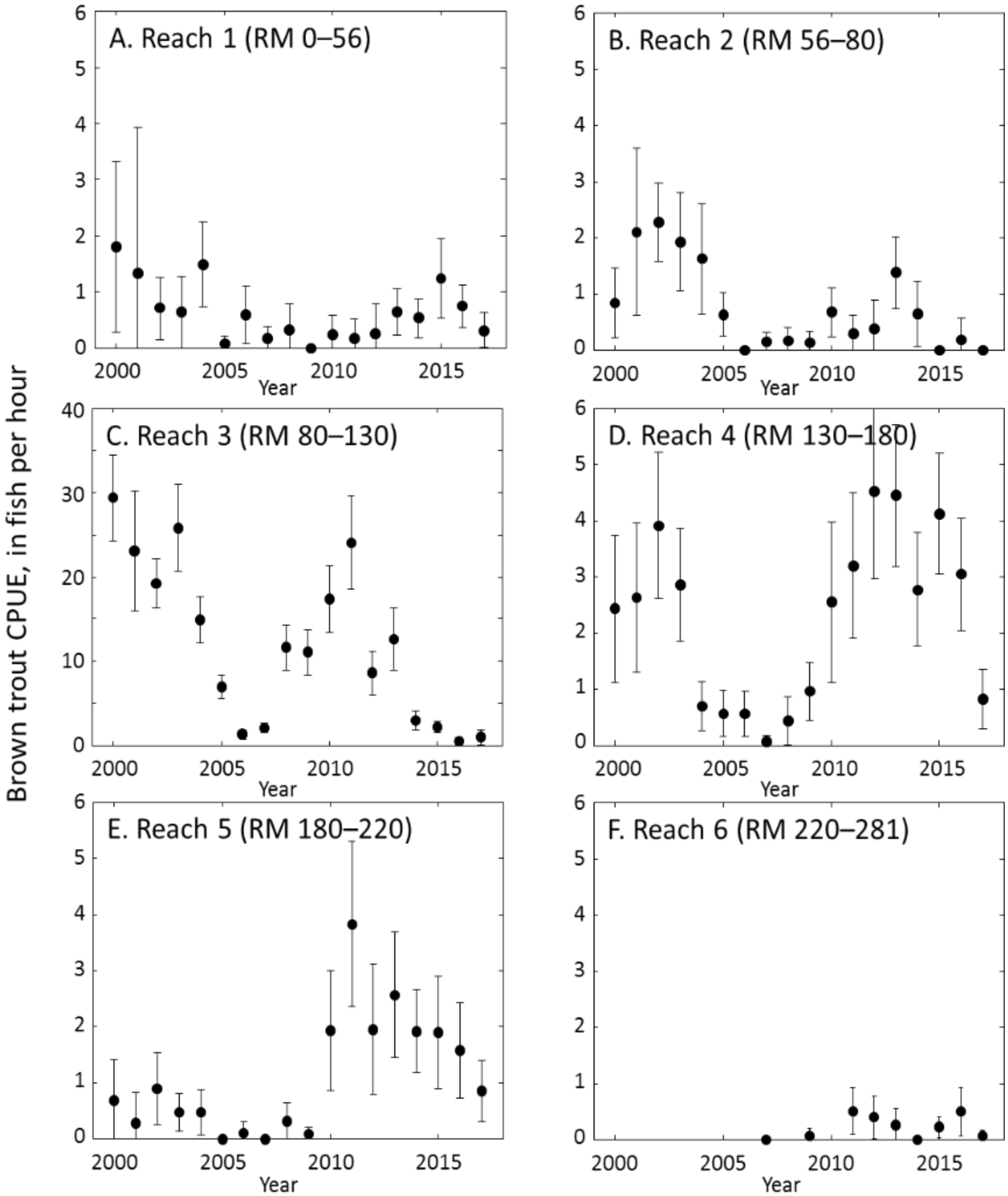
As with rainbow trout, brown trout are not native to the Colorado River and were stocked in Grand Canyon in the first half of the 1900s. Brown trout are no longer stocked in the Colorado River downstream of Glen Canyon Dam and are now found primarily in Lees Ferry and in and near Bright Angel Creek, which supports a naturally spawning population (Reclamation 2011).

Electrofishing catch rates of brown trout have been monitored annually starting in 2000 in six reaches of the Colorado River from Lees Ferry (RM 0) to Pearce Ferry (RM 281) (**Figure TA 8 Attachment 1-9**). Catch rates in most reaches declined from 2000 to 2006 and then increased to the highest levels in 2011-2013 (Makinster et al. 2010; **Figure TA 8 Attachment 1-9**). Because brown trout in the Grand Canyon spawn primarily in tributaries (e.g., Bright Angel Creek [RM 88] and Shinumo Creek [RM 109]), recruitment and abundance may be more affected by conditions in the tributaries than in the mainstem. Overall, catch rates in all reaches declined after about 2015.

Bright Angel Creek was identified as a potential site for translocation of humpback chub to start a second population in Grand Canyon (Valdez et al. 2000). Before translocations of fish could begin, the NPS determined the need to remove as many brown trout as possible from the creek. A trout control project, using a combination of a fish weir trap and electrofishing, was implemented by the NPS to benefit native species in Bright Angel Creek and humpback chub in the Colorado River. Control measures were implemented in winters 2006–2007, 2010–2011, 2011–2012, 2012–2013, 2013–2014, and 2014–2015 under the 2006 and 2013 EAs and a Finding of No Significant Impact (FONSI; NPS 2006, 2013d). Concurrent with intensive mechanical suppression of invasive salmonids, the predominant stream-wide composition of the fish community in Bright Angel Creek shifted from trout (65 percent) in 2012 to native fish (greater than or equal to 77 percent) in 2015 (Healy and Schelly et al. 2020). By the end of the study in 2018, totals of 43,665 brown trout and 7,824 rainbow trout were removed, and native fish represented 97 percent of the fish community. Population estimates for brown trout steadily declined between 2012 and 2018 from a high of 13,829 to a low of 1,315, resulting in a 91 percent reduction by the 2017–2018 sampling season (Healy and Schelly et al. 2020).

Brown trout in the Grand Canyon have shown an ability to move considerably in the Colorado River, sometimes from one tributary to another. Some brown trout captured in Bright Angel Creek were originally tagged in other parts of the Colorado River, as much as 25 miles from Bright Angel Creek (Reclamation 2011). Small numbers of brown trout are also found in other locations within the Grand Canyon, including in the vicinity of the Little Colorado River confluence, where they pose a threat to humpback chub (Valdez and Ryel 1995). A telemetry study documented brown trout moving both up and downstream from Lees Ferry. Detections included one adult brown trout moving from Lees Ferry to just downstream of the Little Colorado River confluence (Schelly et al. 2021). An indication of the relative abundance of brown and rainbow trout in the vicinity of the Little Colorado River is provided by the numbers captured using electrofishing during trout removal efforts (Coggins 2008). Of 23,000 nonnative fish captured as part of removal from 2003 to 2006, totals of 19,020 were rainbow trout and 470 were brown trout (Reclamation 2011). All brown trout removed during these efforts were in the mainstem.

Figure TA 8 Attachment 1-9
Mean Catch Per Unit Effort (CPUE) of Brown Trout Captured during Electrofishing Surveys on the Colorado River between Lees Ferry and Pearce Ferry for Reaches 1–6 (plots A–F), 2000–2017



Data from AZGFD. Figure from Runge et al. (2018).

The brown trout population at Lees Ferry historically consisted of a small number of large fish supported by low levels of immigration from downstream reaches. Over the period 2014–2016, the number of brown trout captured during routine monitoring in the Lees Ferry reach began to increase (**Figure TA 8 Attachment 1-10**). Brown trout were observed spawning near the -4-mile bar in Glen Canyon during October and November of 2014 and 2015, and an increase in age-1 brown trout was observed in 2015 and 2016, likely the result of local spawning and recruitment (Korman et al. 2015). Management agencies and stakeholders questioned whether this increase in brown trout represents a threat to the rainbow trout sport fishery, as well as to downstream humpback chub aggregations. This population of brown trout showed signs of sustained successful reproduction and recruitment of locally hatched fish into the spawning class, based on analysis with a new integrated population model (Runge et al. 2018). The proximate causes of this change in status are a large pulse of immigration in the fall of 2014 and higher reproductive rates in 2015–2017, but the ultimate causes of this change are not fully clear.

Figure TA 8 Attachment 1-10
Index of Brown Trout Abundance (mean fish caught per minute of electrofishing) in Lees Ferry, 1991-2023

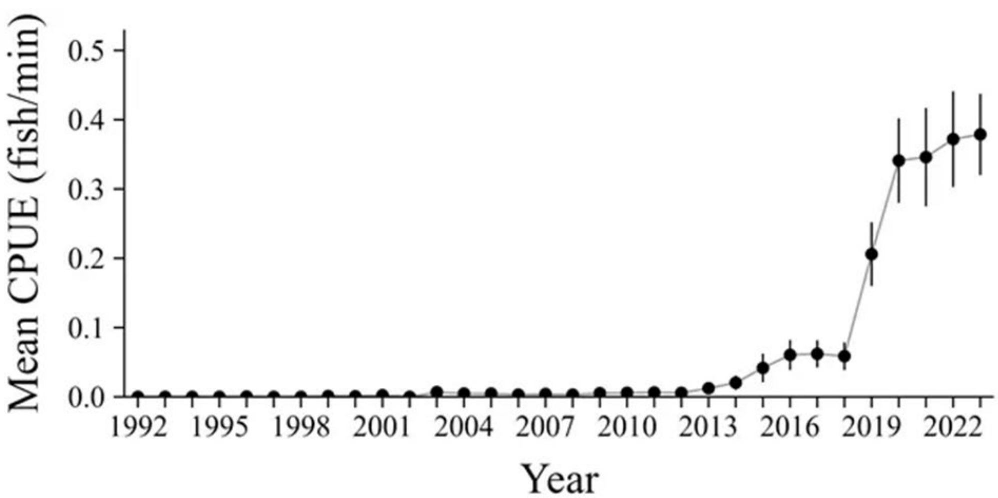


Figure from AZGFD (Rogowski 2023)
Note: The closed circles show the mean value; the whiskers show the 95-percent confidence intervals.

The pulse of brown trout immigrants from downstream reaches in fall 2014 may have been induced by three sequential high flow releases (HFEs) from the dam in November of 2012–2014 but may also have been the result of a unique set of circumstances unrelated to dam operations. The increase in reproduction may have been the result of any number of changes, including an Allee effect, warmer water temperatures, a decrease in competition from rainbow trout, or fall high-flow releases. Correlations over space and time among predictor variables do not allow for a clear inference about the cause of the changes.

There are interventions that may be effective in moderating the growth of the brown trout population in the Lees Ferry reach (Runge et al. 2018). Across causal hypotheses, it is predicted that removal strategies (e.g., a concerted electrofishing effort or an incentivized take program targeted at large brown trout) could reduce brown trout abundance by approximately 50 percent relative to status quo management. Reductions in the frequency or a change in the seasonal timing of high-flow releases from Glen Canyon Dam could be more effective, but only under the causal hypotheses that involve effects of such releases on immigration or reproduction. Brown trout management flows (i.e., dam releases designed to strand young fish at a vulnerable stage) may be able to reduce brown trout abundance to some degree, but are not forecast as the most effective strategy under any causal hypothesis.

A cost-effective strategy was identified that involves rewarding anglers monetarily for every brown trout removed from the river, as compared to intensive mechanical removal (i.e., electrofishing) (Runge et al. 2018). An incentivized harvest program was implemented in the Lees Ferry reach on November 11, 2020. For the next 5 to 6 years, eligible anglers would be offered a reward of at least $25 per brown trout over 6 inches in length removed from the Colorado River between Glen Canyon Dam and the mouth of the Paria River. This reward may vary with the seasons or be adjusted annually, but will typically be in the $25-$33 range. The usual reward is $33 per brown trout and $15 for each PIT tag.

Incentivized harvest has been ongoing for over three years (2020-2023) and has increased substantially in the last year. The adult brown trout population declined significantly in 2023, a significant portion of which can be attributed to incentivized harvest. From November 2020 through December 2023, anglers removed 5,597 brown trout and received $423,190 in payouts (NPS 2024)

The number of brown trout greater than 150 mm TL appeared to moderate after the incentivized harvest was implemented in late 2020, but the numbers increased to the highest level of over 11,000 fish in 2022 and declined to fewer than 6,500 fish in 2023 (**Figure TA 8 Attachment 1-11**). This indicates that the incentivized harvest of brown trout is working to reduce the numbers of fish, but the effects are confounded by several factors, including survival, recruitment, and environmental conditions.

Figure TA 8 Attachment 1-11
Estimated Numbers of Brown Trout larger than 150 mm TL in the Lees Ferry Sub-reach, 2000-2023

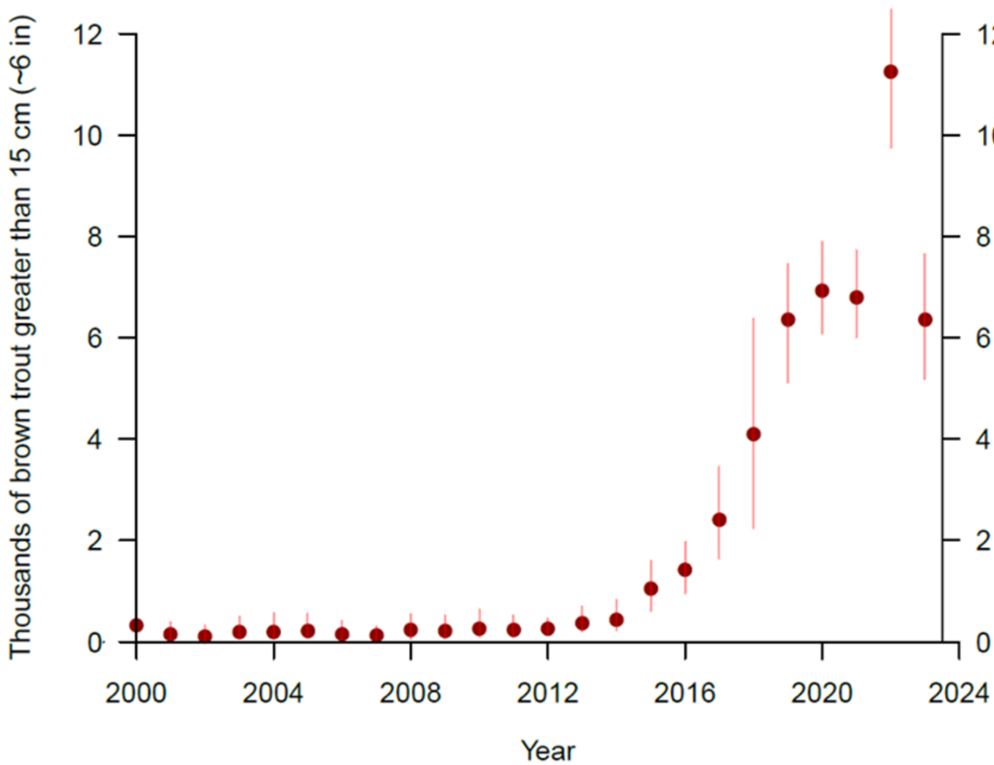


Figure from GCMRC Power Point (Yackulic et al. 2024)
Note: The closed circles show the mean value; the whiskers show the 95-percent confidence intervals.

Habitat

The brown trout is a cold-water fish species that inhabits streams, rivers, and lakes, and can migrate to and from salt water. Adults in streams typically use areas with overhead cover and deep pools, eddies, runs, and overhanging banks for resting, and they feed in open riffles. Juveniles use shallow habitats near cover, and recently hatched fry use shallow shorelines (Sigler and Sigler 1996). Brown trout prefer cover, and they can tolerate and feed at high temperatures and in higher turbidity than other salmonids, such as rainbow trout (Sigler and Sigler 1996). The upper limiting temperature for brown trout is 27.2 °C (81 °F) (Sigler and Sigler 1996). Preferred temperatures for spawning, egg incubation, and growth are 7–14 °C, 8–20 °C, and 11–20 °C, respectively (Valdez and Speas 2007).

In Bright Angel Creek, brown trout use a variety of habitats that include pools, riffles, runs, and cascades. Brown trout from the mainstem Colorado River are attracted to tributaries like Bright Angel Creek by the availability of gravel riffles used as spawning habitat. Limiting access to those habitats by placing a weir at the mouth to intercept upstream adult spawners significantly reduced the numbers of brown trout in the stream, along with ongoing mechanical removal (Healy and Schelly et al. 2020).

Habitat use in the mainstem Colorado River through Grand Canyon is not well known. Adults appear to have relied on seasonally warmed tributaries, such as Bright Angel Creek, to spawn in an otherwise cold mainstem. With warmer releases from Glen Canyon Dam starting in 2004, brown trout increased in numbers by migrating to the Lees Ferry sub-reach and by apparent successful reproduction and recruitment (Runge et al. 2018). Brown trout appear to be using the same habitats for resting, feeding, spawning, egg incubation, and nurseries as rainbow trout in the Lees Ferry sub-reach, except that brown trout spawn in the fall (October-November) and rainbow trout spawn in early spring (February and early March).

Life History

The brown trout can reach a maximum size of nearly 40 pounds and typically matures between 2 and 3 years of age, with females producing 200–6,000 eggs in streams (Sigler and Sigler 1996). Females prepare individual nests or redds in gravel riffles or runs where they deposit eggs that are fertilized by multiple males. Brown trout initiate spawning in the fall with the onset of decreasing water temperatures in the range of 7.2–10 ºC (45–50 ºF), usually from late October to December. The eggs hatch in 41 days at a constant water temperature of 10 ºC (50 ºF) (Sigler and Sigler 1996).

The brown trout has a largely carnivorous diet with younger fish feeding on aquatic and terrestrial insects, as well as mollusks, frogs, and rodents, and older fish feeding primarily on fish and large crustaceans like crayfish (Sigler and Sigler 1996). This propensity for piscivory is why NPS identified brown trout as a "(1) Very High" threat level to rainbow trout and native fish in Glen Canyon National Recreation Area and Grand Canyon National Park (NPS 2021). As sub-adults and adults, brown trout are 17 times more likely to eat other fish than rainbow trout, meaning that even small numbers of brown trout can impose a significant predator threat to native fish (Yard et al. 2011).

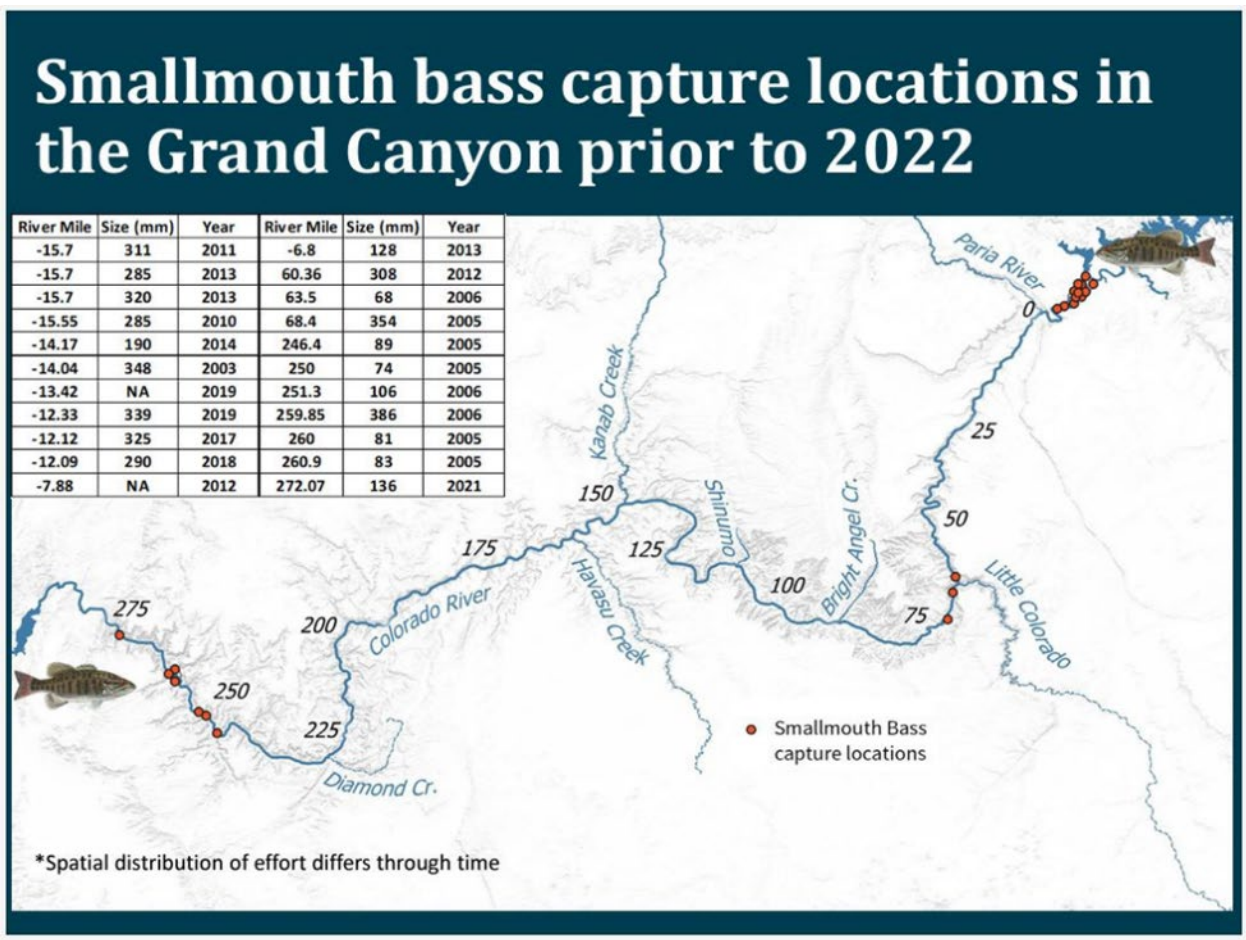
Smallmouth Bass

The smallmouth bass is considered the most problematic nonnative fish species in the Upper Basin in terms of negative impacts on native fish populations and endangered fish recovery (Bestgen et al. 2008, 2018; Breton et al. 2015). Based on results of a bioenergetics model, Johnson et al. (2008) ranked smallmouth bass as the most problematic invasive species in the Upper Basin because of their high abundance, habitat use that overlaps with most native fish, and ability to consume a wide variety of life stages of native fish. Expanded populations of piscivores such as smallmouth bass are a major impediment to conservation actions aimed at recovery of the four endangered fish in the Upper Basin (FWS 2018a, 2020). Smallmouth bass also pose a threat to desirable sportfish, such as rainbow trout. The smallmouth bass was introduced into Lake Powell in 1982 and has periodically been detected downstream of Glen Canyon Dam, where it poses a threat to native, endangered, and threatened species, such as humpback chub (Smallmouth Bass Management Review Committee 2024).

Distribution and Abundance

The smallmouth bass is native to eastern North America west of the Appalachian Mountains, but it has been widely introduced throughout the United States as a sportfish and for nuisance fish control. The smallmouth bass was introduced into reservoirs of the Upper Basin starting in 1967 (Pettengill et al. 1983). It was stocked into Lake Powell in 1982 and has been detected sporadically in the Colorado River downstream of Glen Canyon Dam since early 2000. Prior to 2022, there were records of 22 individuals being caught between Glen Canyon Dam and Pearce Ferry (**Figure TA 8 Attachment 1-12**). Capture locations show three concentrations of fish, including at the Lees Ferry reach downstream of the dam, near the confluence of the Little Colorado River, and in the newly exposed channel at the inflow to Lake Mead. The likely origin of these fish is passage through the dam, moving down the Little Colorado River from upstream reservoirs, and moving upstream from Lake Mead, respectively. Greater numbers of smallmouth bass have been captured in the Lees Ferry reach in 2022 and 2023, which have either passed from Lake Powell through the dam penstocks or have been recently spawned in the area. Most of the smallmouth bass in the Lees Ferry reach have been caught in and near the -12-mile slough (located 12 miles upstream of Lees Ferry), a small side channel of the Colorado River that can entrain and warm water released from the dam (GCMRC 2014). Prior to 2022, the bass were mostly large sub-adults and adults, suggesting that these fish survived passing through the penstocks from Lake Powell. Starting in 2022, many of the bass were smaller, indicating that these fish were produced locally, probably in and around the -12-mile slough.

Figure TA 8 Attachment 1-12
Smallmouth Bass Capture Locations in the Grand Canyon Prior to 2022



Source: GCMRC 2014

Within the analysis area, smallmouth bass are found along rocky shorelines in Lake Powell as a sportfish, in the Lees Ferry reach, near the Little Colorado River, and in the western Grand Canyon; they are also a popular sportfish along rocky shorelines in Lakes Mead, Mohave, and Havasu, and suitable riverine and lake habitats to the SIB.

Habitat

The smallmouth bass inhabits large rivers, streams, lakes, and ponds. Within its native range, it is most abundant in pools of streams that consist of a substantial proportion of riffle habitat, clean, rocky, hard bottoms, and gradients of 0.5-5.0 m/km. In large rivers and lakes, smallmouth bass tend to congregate over hard, rocky bottoms, where currents are present. Presently, smallmouth bass occur in the mainstem Colorado River, in the Verde River, and throughout the Salt River below about 2,200 m elevation (Reclamation 2018).

In late summer and autumn during low-flow, adults use habitats that consist mostly of low-velocity runs or pools separated by shallow, higher-velocity riffles. Substrate is typically a mix of boulder, cobble, gravel, and sand in low velocity areas, and cobble and gravel in riffles. Young life stages use shallow riffles with cobble and gravel substrates. Spawning takes place in backwaters and isolated pools that are created by cutoff high-flow side channels and contain cobble, gravel, and fine-grained substrate (Bestgen and Hill 2016b).

Life History

The smallmouth bass is a large member of the sunfish family (Centrarchidae) that can reach sizes of about 69 cm (27 inches) and 5.4 kg (12 pounds). Smallmouth bass spawn in spring, usually mid-April to July, depending on geographical location and water temperature. The species is highly sensitive to flow and water temperature, which greatly influence reproductive success (Winemiller and Taylor 1982; Peterson and Kwak 1999). Large fluctuations in water level can affect the reproductive success of smallmouth bass (Pflieger 1997; Montgomery et al. 1980). Ideal spawning conditions include one or more substantial rises in water level a week or two prior to nesting (Pflieger 1997) and relatively stable water levels while nesting is in progress (Watson 1955; Pflieger 1997). Rising water can flush nest areas with cold water, causing nest desertion and halting embryo development (Watt 1959; Montgomery et al. 1980). Falling water levels can drive guarding males off, limit water circulation around eggs, and increase predation, resulting in lower reproductive success (Neves 1975; Montgomery et al. 1980).

Spawning is initiated when male smallmouth bass construct nests in low velocity stream habitat, usually channel margin backwaters or cutoff secondary channels, by fanning fine sediment to reveal the gravel and cobble substrate. Nests are built in low velocity areas because eggs and weak-swimming larvae will otherwise be swept away by even relatively slow water currents 5-22 cm/sec (Larimore and Duever 1968). Females are attracted to nests by males, and eggs are deposited and fertilized over the surface of the substrate. Fertilized eggs are semi-adhesive and develop on the surface of the substrate as the male guards them from predators. After hatching, larvae lie on the bottom for several days, and after 1-2 weeks rise from the substrate and gain buoyancy, while still being guarded by the male. Those vulnerable eggs and weak-swimming fry are the life stages most susceptible to changes in flow rates that may sweep eggs and fry away and reduce their survival (Bestgen and Hill 2016b).

Mature females produce 2,000–15,000 golden yellow eggs. Males may spawn with several females on a single nest. On average, each nest contains about 2,500 eggs, but nests may contain as many as 10,000 eggs. Eggs hatch in about 10 days if water temperatures are in the mid-50s (°F), but can hatch in 2–3 days if temperatures are in the mid-70s (°F). Males guard the nest from the time eggs are laid until fry begin to disperse, a period of up to a month.

Temperature is one of the most important factors limiting the distribution of smallmouth bass (Bestgen 2018). Faster growth rates of adults are generally associated with higher summer temperatures. Faster growth rates occur in southern reservoirs, resulting in a shorter life span than in northern regions. In summer, smallmouth bass inhabit warmer shoreline areas of large lakes in the north and deeper, cooler waters in the south. Growth does not begin until water temperatures reach 10–14 °C (50–57 °F). Field data indicate that adults prefer temperatures of about 21–27 °C (70–81 °F) in summer. Smallmouth bass have been reported "sunning" themselves in pools with water temperatures of about 26.7 °C (80 °F) in summer (Edwards et al. 1983). When temperatures drop to 15–20 °C (59–68 °F), adults seek deep, dark areas. At about 10 °C (50 °F), bass become inactive and seek shelter. At 6–7 °C (43–45 °F), most smallmouth bass are beneath the rock substrate, with few remaining on top. The lower lethal temperature is near freezing. Bass will congregate around warm springs in winter when available.

Fry seem to be especially vulnerable to flood conditions and fluctuating water levels (Larimore 1975). A rapid drop in water level may trap them in areas where they will become desiccated (Montgomery et al. 1980). A stream rise of only a few inches, and consequent increase in water velocity, can displace advanced fry newly risen from the nest (Webster 1954). Most fry remain in shallow water (Doan 1940; Forney 1972), although some can be found at depths of 4.6–6.1 m (15–20 feet) (Stone et al. 1954; Forney 1972). Fry 20–25 mm (0.79–1 inch) in length cannot maintain themselves in current velocities faster than 200 mm/sec (Larimore and Duever 1968). An increase in turbulence during flood conditions creates conditions with which smallmouth fry appear unable to cope (Webster 1954). Fry cannot tolerate and are displaced at high turbidities (2,000 Jackson Turbidity Units) combined with an increase in water velocity, but they will not be displaced at moderate turbidities (250 Jackson Turbidity Units) (Larimore 1975). Low water temperatures during flood conditions will reduce fry swimming ability (Larimore and Duever 1968).

Bestgen and Hill (2016b) studied patterns of smallmouth bass reproduction in partially regulated and unregulated reaches of the Green River and the unregulated Yampa River of the Upper Colorado River Basin. Patterns of reproduction in the Yampa River contrasted with regulated or partially regulated Green River reaches, where smallmouth bass reproduction occurred later, and sometimes well after water temperatures reached the threshold 16 °C (61 °F). In the regulated reach of the Green River, successful hatching did not occur until relatively low and stable baseflow levels were reached, noting that high streamflow in 2011 delayed hatching. They postulated that stable baseflows were required for smallmouth bass so that spawning habitat was available and suitable for successful reproduction; spawning at higher flows may have been attempted, but could not be determined.

The existence of established smallmouth bass populations in Lake Powell, combined with the passing of epilimnetic water from the lake through Glen Canyon Dam to the Colorado River downstream, creates a conduit for repeated introductions of smallmouth bass downstream of Glen Canyon Dam. This complicates efforts to minimize the risk of smallmouth bass establishment and expansion in the river downstream of the dam. While directed removal efforts can take place in areas where smallmouth bass have been identified, introductions of smallmouth bass from Lake

Powell could continue unless deterrents are developed. A selected option for releases from Glen Canyon Dam is currently being implemented and evaluated as directed by the Secretary of the Interior's designee (LTEMP SEIS, Reclamation 2024). The cool mix option was selected as the most likely to minimize spawning by smallmouth bass. This option involves releasing water from Glen Canyon Dam as a mixture of cold water from the deeper river outlet works (bypass valves or jet tubes) with warmer water through the penstocks that create cooler water conditions to prevent smallmouth bass from reproducing and establishing downstream of Glen Canyon Dam. This release takes place when a mainstem water temperature of 16 °C (61 °F) is reached near the confluence of the Little Colorado River (RM 61), the temperature at which smallmouth bass spawning behavior begins for introduced populations in the Upper Colorado River Basin (Bestgen and Hill 2016b). The cool mix releases were conducted under the authority of the 2024 LTEMP SEIS and reflect approved experimental treatments under LTEMP for the Grand Canyon and not as part of this Draft EIS.

Green Sunfish

The green sunfish is a medium-sized member of the sunfish family (Centrarchidae) with a maximum length of about 275 mm TL and 960 g (Sigler and Sigler 1996). Although not native to the Colorado River, it was common in Glen Canyon before dam construction in 1958-59 (McDonald and Dotson 1960) and shortly after dam construction in 1967-1968 (Stone and Rathbun 1968). It became rare in the mainstem Colorado River after cold dam releases began in the 1970s, except at the mouths of seasonally warmed tributaries like the Little Colorado River and Kanab Creek (Valdez and Ryel 1995). The numbers began to increase in the mainstem with warmer dam releases starting in 2004, and in 2015-2017, green sunfish were found spawning and successfully reproducing in the -12.5-mile slough (Ward 2015; NPS 2018). Evidence of reproduction was found in Kanab Creek (Healy and Ward et al. 2020) and at RM 243 (Smallmouth Bass Management Review Committee 2024). The green sunfish is a small but effective predator of small fish, and a reproducing population in the Grand Canyon poses a risk to native fish populations. The green sunfish is one of six nonnative fish species designated by the NPS as a "(2) High" threat level for native fish in Glen Canyon National Recreation Area and Grand Canyon National Park (NPS 2021).

Distribution and Abundance

The green sunfish is native to east-central North America, from southwestern New York, west of the Appalachian Mountains, south to Georgia, Alabama, west and south to Texas, northeastern Mexico, and New Mexico (Sigler and Sigler 1996). It was first introduced into the Colorado River System with various other sportfish after Lake Mead began forming in 1935. It was also introduced by state game and fish agencies as a mixture of sportfish throughout lakes and streams of the system, and is considered a sportfish in Lakes Powell, Mead, Mohave, and other waters of the Lower Basin. It is abundant in tributaries of the Grand Canyon, including the Little Colorado River (Stone et al. 2018). As summarized by Valdez and Ryel (1995), small numbers of green sunfish were found in the Grand Canyon in 1970-73, 1975, 1977-78, 1984-86, and 1990-93. Only one adult green sunfish was captured during the 2000 Low Steady Summer Flow experiment (Trammell et al. 2002), and no green sunfish were reported in the 2016 System Wide Monitoring Project from Lees Ferry to Lake Mead (Rogowski et al. 2017).

The lower levels of Lake Powell starting in about 2000 have enabled some lake fish to become entrained in the dam penstocks and enter the Colorado River downstream. Lower lake levels have also resulted in warmer dam releases that have enabled warmwater fish like green sunfish and smallmouth bass to spawn downstream of the dam. Although green sunfish have been present in the river since dam construction, their numbers have remained low, and they have not been considered a threat to native fish (Valdez and Ryel 1995). However, numbers began to increase because of dam entrainment and local reproduction, and the NPS decided to act. In November 2015, the NPS found evidence of reproduction by green sunfish in a slough located 3 miles downstream of Glen Canyon Dam (-12-mile slough, 12 miles upstream from Lees Ferry), and the slough was treated with rotenone (a chemical piscicide) to eradicate green sunfish. Additional green sunfish were discovered in the slough in 2016 that were believed to be passing through the dam penstocks from Lake Powell, and in 2018, the Bureau of Reclamation requested temperature reduction options to cool the water in the slough so that green sunfish and other invasive species (e.g., smallmouth bass) could not spawn. A series of temperature reduction options were developed (Greimann and Sixto 2018), but have not been fully implemented. A second rotenone treatment was applied to the -12-mile slough in August 2023.

Numbers and lengths of green sunfish captured in the Lees Ferry reach have been recorded and analyzed as part of the Trout Recruitment and Growth Dynamics project in Glen Canyon between June 2022 and November 2023 (Smallmouth Bass Management Review Committee 2024). These data were used to estimate the capture probability of green sunfish and smallmouth bass. The number of green sunfish is greater and provides more reliable statistics for two sub-reaches, reach 1A (RM -14.3 to -12.5 mile with its downstream end located just upstream of the slough) and 1C (-4.3 to -2.7 mile). The average green sunfish abundance was ~3,900 and 390 in reaches 1A and 1C, respectively. Green sunfish abundance was ~10-fold higher in the reach closer to the dam (1A), located upstream of the slough. The average abundance of smallmouth bass was 182 and 28 in 1A and 1C, respectively. Smallmouth bass abundance in 1A was ~ 6-fold higher than in 1C. Green sunfish abundance was 21-fold higher than smallmouth bass abundance in 1A, and 14-fold higher in 1C. Higher numbers of green sunfish from the sub-reach closer to the dam indicates that the fish were passing through the dam from Lake Powell, but the presence of small fish is evidence of local reproduction (**Figure TA 8 Attachment 1-11**).

Within the analysis area, green sunfish (and other sunfishes) are found along shorelines in Lake Powell, in the Lees Ferry sub-reach, near the Little Colorado River, and in the western Grand Canyon. They are also a popular sportfish along shorelines in Lake Mead, Lake Mohave, and Lake Havasu, and most likely wherever there is suitable riverine and lake habitat downstream to the SIB.

Habitat

Green sunfish inhabit small, warm streams, ponds, and shallow areas of lakes. They occur over a variety of habitats and in a variety of covers, including emergent vegetation, woody debris, and rocks. Green sunfish can tolerate a range of water temperatures from cool (18–21 ºC) to warm (30–32 ºC) conditions.

Green sunfish spawn in spring in water 0.61-2.44 m deep at temperatures of 18.9 ºC. They can spawn multiple times until the temperature reaches about 28 ºC. Males construct oval nests near rocks, logs, clumps of grass, or other debris, and escort individual females to deposit eggs over the nest. Females produce 2,000-26,000 eggs, and the eggs incubate 3–5 days at 23 ºC. More than one female may deposit eggs on the nest, and the male defends the nest aggressively (Sigler and Sigler 1996).

Life History

Green sunfish live up to about 10 years of age and can reach sizes of 275 mm TL and 300 g. They have high reproductive survival and are highly predaceous starting with the earliest life stages. They feed on small fish and can dominate small ponds when introduced into a bluegill (*Lepomis macrochirus*)/largemouth bass (*Micropterus salmoides*) fishery. They also consume aquatic and terrestrial insects, crustaceans, and amphibians, and they can hybridize with other species of sunfishes of the genus *Lepomis* (Sigler and Sigler 1996). These aspects of the life history of the green sunfish make it a high threat to local native fish populations.

Flathead Catfish

The flathead catfish is a large North American freshwater catfish species. Adult flathead catfish can reach lengths of between 90 and 120 cm, with some individuals growing even larger. The species have been captured weighing 23kg to 45 kg (TPWD 2024). Flathead catfish are gray to brown with dark brown-black mottling that may extend ventrally onto the belly. The undersides of the head and body are white to yellow. Their nasal and maxillary barbels are dark, and their chin barbels are white to yellow. The head is broadly flattened, with a projecting lower jaw. The tail fin is only slightly notched, not deeply forked (Pflieger 1997; Ross 2001; Fuselier 2014; TPWD 2024).

Distribution and Abundance

The flathead catfish is native to the United States, with its original distribution covering the Mississippi River Basin and Great Lakes region (TPWD 2024). They are widely distributed in the Mississippi River valley, ranging from South Dakota and Pennsylvania south to the Gulf Coastal Plain. It also occurs in large rivers of the Gulf of Mexico Basin, from the Mobile Basin west to the Rio Grande (Minckley and Deacon 1959; Hubbs et al. 2008; Glodek 1980). Over time, flathead catfish have been introduced to many other water systems across the country, sometimes leading to ecological challenges due to their predatory nature and competition with native species, including in the Lower Colorado River Basin. Due to its desirability as a sport fish, it has been widely stocked outside of its native range (Young and Marsh 1990).

Due to its large size and palatable flesh, flathead catfish have been introduced into parts of Arizona, California, Florida, Idaho, New Mexico, Oregon, and South Carolina (Moyle 1976). The first introduction in the southwest was to the Gila River system in New Mexico or Arizona sometime prior to 1950 (Young and Marsh 1990). In 1962, the AZGFD stocked flathead catfish at six locations along the lower Colorado River (Bottroff et al. 1969). Subsequent introductions from 1968 to 1975 occurred in the Salt, Verde, and San Francisco rivers of Arizona. Since these initial introductions, flathead catfish have spread to inhabit waters throughout the lower Colorado, Gila, Salt, and Verde rivers in Arizona. Population size varied with discharge in the lowermost Colorado River and was estimated to range from 155 to 259 flathead catfish per km when discharge was at its maximum in 1986 (Young and Marsh 1990).

Habitat

Flathead catfish prefer habitats in large rivers and streams, as well as reservoirs and lakes with slow-moving or still waters; they favor areas with deep pools, undercut banks, and abundant cover, such as submerged logs, rocks, and debris (Fuselier 2014; Pflieger 1997; TPWD 2024). They occupy a variety of stream types while avoiding high gradients or intermittent flow. It is not likely to be found in headwater creeks (Pflieger 1997). These environments offer both protection and an ample food supply, primarily consisting of live prey such as other fish, as well as crustaceans. Flathead catfish are known to be nocturnal predators, often hunting during the night and relying on their keen sense of smell and taste to locate prey in the dark, turbid waters they inhabit (Ross 2001; Fuselier 2014; TPWD 2024). Optimal water temperatures for flathead catfish range from 25 to 30ºC, but they can tolerate a broader range (TPWD 2024).

Life History

Flathead catfish have a relatively slow growth rate compared to other freshwater fish, with individuals taking several years to reach maturity. They can live for 12 years or more, with some documented cases of individuals living over 20 years (Fuselier 2014; Pflieger 1997; TPWD 2024). Spawning typically occurs in late spring to early summer when water temperatures reach approximately 20 to 26 °C (Fuselier 2014; Pflieger 1997; TPWD 2024). During this time, male flathead catfish prepare nests in secluded, sheltered areas, such as hollow logs, cavities under rocks, or man-made structures like submerged pipes or riprap. Once the nest is prepared, the male attracts a female to the site, where she can lay up to 100,000 eggs, depending on the size of the female. After fertilization, the male takes on the role of guarding the nest from predators (Ross 2001; TPWD 2024).

Flathead catfish are of particular concern as they are voracious predators and pose a substantial threat to native fishes. This species appears to be expanding its range from Reaches 4/5 and upstream within Reach 3 (potentially due to temperature), and as such, it is routinely monitored by FWS personnel. This species had been captured nearly as far upstream as the area between Needles and Fort Mohave in the Lower Colorado River (near Boundary Cone Road crossing) (Rasset and Love-Chezem 2024).

The rapid dispersal and population growth rates of introduced flathead catfish, coupled with their obligate carnivorous feeding habits, have raised concerns among ichthyologists and management agencies. The FWS has ranked flathead catfish as its highest priority among invasive animal species in the southeastern United States (Brown et al. 2005).

Acronyms and Abbreviations

°C	degrees Celsius
°F	degrees Fahrenheit
AZGFD	Arizona Game and Fish Department
BLM	Bureau of Land Management
CI	confidence interval
CPUE	Catch Per Unit Effort
EIS	Environmental Impact Statement
ESA	Endangered Species Act
HFE	High-Flow Experiment
IUCN	International Union for Conservation of Nature
km	kilometers
Lower Basin	Lower Colorado River Basin
m	meters
Mexico	United Mexican States
mm	millimeters
NPS	National Park Service
PIT	passive integrated transponder
RM	river mile
FWS	U.S. Fish and Wildlife Service
SIB	Southerly International Boundary
TL	total length
YOY	young-of-year

Glossary

Analysis Area	Specific geographic boundary and timeframe used to evaluate the potential environmental effects of a proposed action
auspices	Sponsorship, guidance, or protection provided by an organization or authority
backwaters	Areas of slow-moving or stagnant water connected to a river or stream
bioenergetics model	A mathematical model that quantifies the balance between energy intake from food consumption and energy release through metabolism, activity, waste, and reproduction
broadcast	A reproductive method in which eggs and sperm are released freely into the water column for external fertilization
cascades	Steep, high-energy sections of a stream or river characterized by boulder-dominated beds and turbulent flow
critical habitat	Areas officially designated under the Endangered Species Act as essential for the recovery and survival of endangered or threatened species
congeneric	Belonging to the same genus

creel census	Data collected from anglers to estimate catch, harvest, and success rates
debris flow	A fast-moving mass of water-saturated soil, rock, and organic material that travels downslope under the influence of gravity
eddy	Localized area of recirculating flow
deltaic	Anything related to, formed by, or characteristic of a river delta
downlisting	Changing a species' legal status under the Endangered Species Act from "endangered" to "threatened"
endemic	Refers to a species that is native to and restricted within a defined geographic area
extirpated	Describes a species that no longer exists in a localized zone
fecundity	The reproductive capacity of an organism
fidelity	The tendency of an organism to return to or remain in a specific location or habitat type
fry	The early free-swimming life stage of fish that begins after the yolk sac is absorbed and the individual starts active feeding
hydropower operations	The management of reservoir storage and water releases through turbines and spillways at hydroelectric facilities to produce electricity
impoundment	A man-made basin or reservoir created by a dam, berm, or other structure that holds or stores water
inundated	Covered or submerged by water, either seasonally or permanently
Interagency Standardized Monitoring Program (ISMP)	A general description for cooperative monitoring efforts between government agencies, rather than a single program
larval drift	The downstream transport of newly hatched fish by river currents
Lower Colorado River Multi-Species Conservation Program (LCR MSCP)	A comprehensive program for the protection of 27 species and their habitat in the Lower Colorado River Basin
mainstem	The principal or primary channel of a river system
mark-recapture	A population estimation method in which individual organisms are captured, marked, and released, then later recaptured to determine population loss
meristically	Anatomical traits used in describing or identifying species
nursery habitat	Habitat that provides favorable conditions for the survival and growth of juvenile life stages

oxbow pond	A U-shaped or crescent-shaped waterbody formed when a river meander becomes cut off from the main channel
penstock	A pipe that moves water under pressure from a reservoir or intake structure to turbines in a hydroelectric system
piscivory	The feeding behavior of eating fish
PIT-tagged	Describes an organism that has been implanted with a small microchip that contains a detectable code by a specialized scanner
progeny	Offspring or descendants produced by an organism
radio tagged	Describes an organism that has been equipped with a radio transmitter that emits detectable electromagnetic signals
radiotelemetry	A tracking method that uses radio transmitters and receivers to locate and monitor the movement and behavior of organisms
recruitment	The addition of new individuals to a population or to a specific life stage
recurrent channels	Secondary or alternative flow paths within a river or floodplain that periodically carry water during certain flow conditions
redd	A nest created by a female fish where she deposits and buries her eggs in gravel
refugia	Areas that provide shelter or protection for species or communities during periods of environmental stress
ripe fish	A sexually mature fish whose reproductive organs are fully developed and capable of releasing eggs or sperm
rotenone	Naturally derived fish toxin used to eliminate unwanted or invasive fish species from waterbodies
silt-laden	Containing or heavily mixed with fine particles
slack water	Water with little to no current velocity often occurring along channel margins or in backwater areas
slough	A slow-flowing or stagnant side channel, backwater, or wetland area associated with a river, floodplain, or estuary
sonic tagged	Describes an organism equipped with an acoustic transmitter that emits sound pulses detectable by monitories
stocking	Intentional release of fish into a waterbody to supplement natural populations
stratified releases	Controlled discharge of water from specific depths within a reservoir to manage temperature, oxygen levels, or water quality
stressor	Any chemical, physical, or biological factor that can cause adverse effects on an organism, population, or ecosystem

talus	Sloping accumulation of loose rock fragments that collects at the base of a cliff or steep slope due to mass wasting processes
telemetry	The remote detection and transmission of data from tagged organisms or monitoring devices to receivers for analysis
total dissolved solids	A measure of all inorganic and organic substances dissolved in a volume of water, including minerals, salts, and organic matter
translocation	Intentional movement and release of organisms from one location to another

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