Review of Smallmouth Bass Management in the Colorado River Ecosystem

Final Report

Smallmouth Bass Management Review Committee

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Executive Summary

The potential expansion of the abundance and range of smallmouth bass (*Micropterus dolomieu)* in the Colorado River ecosystem (CRe, here defined as the Colorado River between Glen Canyon Dam and Lake Mead) beyond their present distribution in the Glen Canyon Dam tailwater is of increasing concern to river and fisheries managers. The concern is motivated by the potential threat of smallmouth bass predation on native fishes, particularly to humpback chub (*Gila cypha)*, a species classified under the Endangered Species Act as Threatened, having been downlisted from Endangered in 2021. The CRe has two population centers of humpback chub, near the Little Colorado River confluence (River Mile 61) and between Havasu Creek and Pearce Ferry Rapid (RM 157 – RM 280) in west-central and western Grand Canyon (Van Haverbeke et al. 2017). The population centers of humpback chub do not overlap with the area where smallmouth bass are now becoming established. The concern of managers is that smallmouth bass might expand their range downstream into the humpback chub population centers. Decision-makers are presently considering the adoption of new management actions to reduce the risk of smallmouth bass population establishment in Glen Canyon (RM -16) and the immediate Glen Canyon Dam tailwater, thereby reducing the risk of their expansion downstream. The apprehensions of the river and fish management communities are consistent with the dominant paradigm of managers elsewhere in the Colorado River basin – that smallmouth bass poses an immediate and significant threat to humpback chub populations in the CRe. This risk of impact to humpback chub populations from smallmouth bass has motivated the proposed revision of the 2016 Long-Term Experimental and Management Plan (LTEMP) to reduce the risk of smallmouth bass establishment, including experimental water releases from Lake Powell reservoir to interrupt smallmouth bass spawning by manipulating flows and/or water temperature (https://tinyurl.com/3w88myxs). Separate from that EIS process, mechanical removal efforts for smallmouth bass have been initiated in Glen Canyon.

Glen Canyon Dam is approximately 76 RM upstream from the nearest humpback chub population center (Little Colorado River confluence near RM 61), and there is concern about a potential cascade of events if smallmouth bass become established in Glen Canyon and then expand downstream. **We think the uncertainty in predictions about smallmouth bass establishment near the downstream humpback chub population centers, and their impact on chub populations if smallmouth bass do become established, is not adequately recognized.**

Smallmouth bass invasion is a significant management concern in the Upper Basin. Federal and state agencies have dedicated large efforts to control the species' abundance. We recognize that smallmouth bass populations have expanded in the Upper Basin and may be limiting native fish recovery, including humpback chub. However, our review of smallmouth bass invasion risk in the CRe suggests key factors beyond the present focus on the role of river temperature,

should be considered when evaluating whether smallmouth bass range expansion is likely to occur into existing humpback chub population centers. The experience of smallmouth bass invasion and control in the Upper Basin may have limited relevance to the CRe, because humpback chub population centers in Grand Canyon differ in habitat, turbidity, and temperature regimes compared to the Upper Basin. Glen Canyon, including the Glen Canyon Dam tailwater, are more similar to the Upper Basin than Grand Canyon and the Little Colorado River confluence area or the west-central and western Grand Canyon. Thus, smallmouth bass invasion may be successful in Glen Canyon but not further downstream in Grand Canyon, where the largest humpback chub population centers are found.

Present management proposals in the CRe concerning smallmouth bass are partly based on the work of Eppehimer et al. (in-review), who developed a detailed model linking Glen Canyon Dam storage and operations, water temperatures, and smallmouth bass population dynamics. This model accurately predicts adult smallmouth bass catch near the dam and in Glen Canyon in 2022 and 2023. It is not clear whether the model was tested in other years of low reservoir levels and warm water temperatures, such as 2006. The authors of this model assume temperature is the limiting factor for smallmouth bass distributions and use a matrix population model parameterized by age-specific survival, fecundity, and growth information for smallmouth bass from the Upper Basin [see supplemental files in Eppehimer et al. (in-review)] to predict smallmouth bass population growth rates. These rates are predicted at three locations in the CRe (Lees Ferry, LCR confluence, Diamond Creek confluence) using predicted water temperatures under different water management and basin-level hydrologic scenarios. While it is reasonable to assume that the age-specific survival, fecundity, and growth information from the Upper Basin used in the smallmouth bass matrix model is realistic for Glen Canyon, we are uncertain whether these assumptions hold for the downstream locations due to the different environmental conditions.

Water temperature may not be the only limiting factor for smallmouth bass establishment near the Little Colorado River or in west-central and western Grand Canyon. We are unsure if the key assumption in Eppehimer et al. (in-review) of temperature as the sole limiting variable for smallmouth bass distributions fully captures the uncertainties in smallmouth bass population dynamics in Grand Canyon and downstream. The downstream river segments differ from the tailwater in Glen Canyon in temperature, turbidity, and habitat. The Colorado River near the Little Colorado River and in west-central and western Grand Canyon are also very different from the Upper Basin, where smallmouth bass populations have proliferated. For example, the Yampa and middle Green Rivers have much lower base flows and different temperature and turbidity regimes than Grand Canyon. These differences, particularly in turbidity, could be included in future modeling efforts to examine the likelihood of smallmouth bass expansion in the CRe.

Endemic fish species in the CRe evolved under river conditions with highly fluctuating river discharge and turbidity levels (Moran et al. 2018). In the post-dam CRe, overall turbidity levels have declined significantly. This change in turbidity is one reason non-native fish species have successfully invaded the CRe (Moran et al. 2018). Turbidity varies significantly over space and time in the CRe, from very low turbidity (median since 2015 < 2 FNU) levels near Glen Canyon Dam for most of the year to high turbidity throughout Grand Canyon during the monsoon season (Voichik and Topping 2014). Most centrarchids and salmonids (including smallmouth bass and rainbow trout) evolved in habitats that rarely exceed 50 FNU (similar to NTU) (Trebitz et al. 2007 as in Moran et al. 2018). Korman et al. (2021) identified how reductions in rainbow trout feeding efficiency due to episodic inputs of fine sediments from tributaries (thereby increasing turbidity) combined with higher trout densities and lower prey availability reduced trout growth in Grand Canyon downstream of Lees Ferry, which contributed to large reductions in rainbow trout abundance. Rainbow trout and smallmouth bass have similar reductions in visual reactive distances to turbidity (Sweka and Hartman 2003, Figures 2 and 4) with significant impairment at turbidity levels of ≥ 50 NTU. During summer monsoon events, turbidity levels in western Grand Canyon can exceed several hundred NTU, and turbidity levels ≥ 50 NTU occur for several months each year in the CRe downstream from the Little Colorado River. This turbidity level would likely inhibit juvenile and adult smallmouth bass feeding (Sweka and Hartman 2003), reducing the condition of adult smallmouth bass and leading to higher mortality, as observed for rainbow trout (Korman et al. 2016; Yard et al. 2016; Korman et al. 2021). Feeding impairment from turbidity has also been documented experimentally for multiple fish species, including rainbow trout and smallmouth bass (Ward et al. 2016; Ward and Vaage 2019).

We suggest that turbidity may contribute to the limitation of smallmouth bass population growth by influencing reactive distance to prey, feeding rate, growth, and survival at locations in Grand Canyon, especially in reaches that experience periods of high turbidities, including the Little Colorado River and west-central and western Grand Canyon. We recognize that Yard et al. (2011) reported a higher incidence of piscivory of native fish by rainbow trout during periods of high turbidity in the Little Colorado River inflow reach. However, these were short-term effects on native fish species, and humpback chub populations have expanded over longer periods, and rainbow trout populations have declined (Yackulic et al. 2014; Korman et al. 2021). Regular year-class failure could limit smallmouth bass population growth, thus reducing the risk to humpback chub populations in areas with high turbidity events downstream of the Little Colorado River. Over the long term, if turbidity suppresses non-native fish populations, the net effect on humpback chub populations would be positive.

The role of turbidity in influencing smallmouth bass feeding, growth, persistence, and the risk to humpback chub populations is a reasonable hypothesis that should be explored. This information can inform the risk of smallmouth bass invasion in different locations and assess the efficacy of various management options and their costs. Examination of this alternative hypothesis should be explored under a structured decision making framework. This approach is common in adaptive management programs and has been used recently in the Colorado River ecosystem to assess management policies for invasive brown trout (Runge et al. 2018).

We are not confident that mechanical removal, short-term flow spikes of dam releases, or manipulation of the temperature of releases from Glen Canyon Dam (informally known as cool mix) will substantially reduce the abundance of smallmouth bass in the tailwater, especially over the long term. The cool mix option, which requires more frequent use of jet tubes to cool water temperatures and reduce entrainment, is likely the most effective action to limit smallmouth bass abundance in Glen Canyon in the short term. However, long term application of this approach may be infeasible due to engineering and policy issues associated with prolonged used of the river outlets.

The conviction of river and resource managers to take immediate management actions is understandable, because those managers are highly risk-averse to the dangers posed by smallmouth bass populations. We urge caution in this approach.

The likelihood of smallmouth bass persisting in the Glen Canyon Dam tailwater and Glen Canyon during the next 20 years is high, even with the short-term measures (e.g., mechanical removal and designer flows) proposed in the next 3 years to prevent smallmouth bass expansion. Persistence in the smallmouth bass population in Glen Canyon will add to the number of downstream dispersing individuals in future years, leading to at least some risk to humpback chub downstream of the Little Colorado River, whether or not self-sustaining bass populations develop in those downstream areas. In terms of the likely effect of smallmouth bass predation on humpback chub in the Colorado River ecosystem, it is highly uncertain whether predation mortality would be sufficiently high to reverse ongoing humpback chub population growth near the Little Colorado River confluence or reverse the rapid and large humpback chub population expansion in west-central and western Grand Canyon. The humpback chub population expansion in the west-central and western Grand Canyon occurred when water temperatures exceeded the smallmouth bass temperature spawning threshold of 16°C for several months each year, which has occurred since at least 2012. Smallmouth bass are present in Lake Mead reservoir and have had access to the western Grand Canyon between the time when smallmouth bass were introduced in Lake Mead in the 1990s and the emergence of the Pearce Ferry Rapids migration barrier in 2008 or 2009. Notably, most of RM 240 – RM 280 (Separation Canyon to Pearce Ferry) has returned to riverine habitat since about 2004 with decreased water storage in Lake Mead, and this reach now supports high catches of humpback chub (Van Haverbeke et al. 2014). Humpback chub were also collected downstream of RM 280 (Pearce Ferry) in Lake Mead in the last two or three years. Smallmouth bass invasion is possible in this westernmost part of the CRe from a Lake Mead source population. Smallmouth bass have been irregularly captured in this region for more than a decade. Interactions between humpback chub and smallmouth bass downstream of Pearce Ferry could provide insight into how these two species interact in a reach of the Colorado River where high turbidity events are more common than Glen Canyon.

We think the various management actions being considered to control smallmouth bass recruitment are unlikely to be effective given the modest history of success of similar actions in the last two decades in the CRe. **However, if actions are undertaken, these actions should be viewed as an adaptive management process.** We also think an adaptive management program could be integrated with the ongoing experimental efforts downstream from Flaming Gorge Dam to improve learning on approaches to use dam releases to manage native and non-native fish species in the Upper and Lower basins.

If smallmouth bass population responses are similar to those seen for rainbow trout in the CRe, the results of previous studies suggest the proposed short-term management actions are unlikely to reduce smallmouth bass population expansion significantly. However, there is uncertainty in whether smallmouth bass population expansion will occur similarly to rainbow trout. Mechanical removal of rainbow trout in the CRe near the Little Colorado River did not substantively reduce their abundance or likely have any impact on the recovery of humpback chub (Coggins et al. 2011), and designer flows reduced incubation survival of rainbow trout eggs in the tailwater but did not reduce recruitment (Korman et al. 2011). Inducing smallmouth bass year-class failure is required for the proposed management actions to reduce smallmouth bass population growth in the tailwater. Interrupting recruitment for a limited number of weeks in a year is different from producing a year-class failure, and limiting recruitment may not change overall year-class strength owing to compensatory increases in juvenile survival rates when densities are reduced. In addition, short-term cold water and high-flow releases designed to reduce smallmouth bass nest success may have limited effects in the tailwater due to renesting by smallmouth bass. Longer duration cold water releases may limit smallmouth bass spawning in river reaches where the cold treatment persists. These actions may have limited effects near downstream humpback chub population centers because of natural river warming and flow spike attenuation with distance from Glen Canyon Dam.

Based on our modeling of available smallmouth bass and green sunfish removal data, **it is unlikely that mechanical removal of smallmouth bass in the Glen Canyon Dam tailwater is an effective long-term population control measure.** Due to low capture probability, a sustained high-effort mechanical removal would only substantially reduce smallmouth bass abundance if the population growth rate is very low due to limited entrainment and lack of local reproduction.

We recognize that our report differs from the dominant paradigm related to smallmouth bass in the Colorado River basin and that even suggesting this alternative paradigm will likely create disagreements among scientists and GCDAMP stakeholders. Disagreement is common in complex resource management decisions (Williams et al. 2012). The GCDAMP program has used basic elements of adaptive management (Melis et al. 2015) and structured-decision making to address other complicated issues, including management of non-native rainbow and brown trout (Runge et al. 2018) and long-term experimental planning (Runge et al. 2015). We encourage using these approaches for the current situation with smallmouth bass in the CRe.

The management implications of this report's findings depend on the time frame being considered.

<u>Considerations in 2024-2026</u>

The current predictions from the March 2024 24-month study are that Lake Powell releases are likely to exceed 16C in the summer and fall of 2024, which may result in suitable conditions for smallmouth bass entrainment and possibly local reproduction (Eppehimer et al. in-review). Different management actions will likely be taken to reduce the likelihood of smallmouth bass reproduction and expansion in Glen Canyon in light of the present risk-averse management strategy in place, including mechanical removal and one of the flow and temperature fluctuation actions proposed for Glen Canyon Dam. We recommend the outcome of these management actions be evaluated in a structured decision making framework to document the actions' motivation, trade-offs, and costs. A structured decision making approach to decisions made in 2024 could be a template for expanding similar approaches for long-term planning. For example, these types of models could be used to evaluate different long-term management options as part of the risk assessment for smallmouth bass expansion to downstream locations, including temperature, turbidity, and other factors, and then evaluate the risk of humpback chub extinction from smallmouth bass predation under different population starting points, predatory and competitive interactions, and population dynamics of humpback chub and smallmouth bass. Cost information from these different policies should also be considered as part of the decision-making process.

Considerations concerning the development of the post-2026 Guidelines

We have described uncertainties related to whether smallmouth bass invasion downstream from Glen Canyon Dam and Glen Canyon is likely to increase the extinction risk to humpback chub and other native fish in the CRe, and the efficacy of short and long-term management actions. There is scientific consensus that the future hydrology of the Colorado River basin will be drier, but there is great uncertainty about how dry that future will be. Although the Bureau of Reclamation and the Colorado River Basin States are doing their best to identify robust policies in the face of hydrologic uncertainty, the best-made post-2026 management plans may not be successful. For these reasons, management actions for the CRe should not be developed prescriptively and be static for the next several decades. We encourage the GCDAMP to "reinvigorate the implementation of adaptive management" (Runge et al. 2011) by embracing the uncertainties associated with smallmouth bass invasion in the CRe, evaluating the effects of different possible actions, and assessing how the consequences of the existing scientific uncertainties relate to these actions. As demonstrated previously for Grand Canyon (Runge et al. 2015; Runge and Bean 2020), using decision science tools "...could not remove the contentiousness from the decision, but they made transparent the nature of the trade-offs and the effects of uncertainty, allowing focused and well-informed deliberations among the decision makers and stakeholders." This is the spirit in which we hope this report is received.

Acknowledgments

Clayton Palmer assembled this panel and gave us our charge. Sadly, Clayton passed away before reading a draft of our work. The panelists have all known Clayton for decades, and Clayton made us better scientists through his inquisitive nature and tough questioning. In the early 2000s, Clayton repeatedly raised concerns that slow-moving basin-wide changes in hydrology could disrupt the best-laid plans of the GCD-AMP to use experiments to understand the effects of Lake Powell releases on resources of Grand Canyon. Those concerns seem highly applicable today. We thank Carl Walters for helpful discussions in developing critical aspects of this work and Rich Valdez for sharing his decades of sampling experiences and observations of smallmouth bass and native fish throughout the Colorado River basin.

Section 1: Introduction and situation

In 2022 and 2023, non-native smallmouth bass (*Micropterus dolomieu;* hereafter, SMB) were collected in the Colorado River downstream from Glen Canyon Dam (GCD) in the 15 miles of Glen Canyon between the dam and Lees Ferry. Based on the size structure of those collected fish, it is likely that they had been entrained from Lake Powell reservoir through the dam's penstocks and had survived passage through the dam's turbines. It is also likely that local reproduction of SMB has occurred in the river downstream from the dam. The establishment of SMB in the Colorado River ecosystem (defined as the river between GCD and Lake Mead reservoir) is a significant conservation concern to resource management agencies and stakeholders, because SMB is a predatory species whose invasion into the Colorado River ecosystem has the potential to limit the recovery of native fish species, including humpback chub (HBC). In the Upper Basin (defined as the Colorado River watershed upstream from Lake Powell), SMB invasion is hypothesized to have adversely affected native species, and SMB are considered a significant threat to native fish [USFWS 2002; Johnson et al. 2008; USFWS 2018 (HBC SSA); USFWS 2020 (CPM SSA)]. Ongoing management efforts, including mechanical removal and experimental designer flows, are being tested in the Upper Basin (Breton et al. 2015; Bestgen 2018; Bestgen and Hill 2016) to interrupt SMB spawning and increase the effectiveness of control efforts. This work is ongoing.

An established SMB population in Glen Canyon is a conservation concern, because a Glen Canyon SMB population could act as a dispersal source into areas downstream where humpback chub (HBC) are found. The assumption that SMB would emigrate downstream is likely based on the CRe experience with rainbow trout (*Oncorhynchus mykiss*). Previous CRe research showed that when rainbow trout populations were high in Glen Canyon, rainbow trout emigrated from Glen Canyon to downstream reaches, including the LCR confluence, where they had the potential to compete with and prey upon HBC (Yard et al. 2011; Korman et al. 2016) and can impact HBC (Yackulic et al. 2018).

Currently, SMB are considered a significant threat to the native fish community in the CRe (SMB Technical Workgroup 2022; https://tinyurl.com/23dc5fy7). This perspective is based on the paradigm of the Upper Basin, that the invasion of SMB has adversely affected native fish species, and SMB is a significant threat to native fish population recovery [USFWS 2002; Johnson et al. 2008; USFWS 2018 (HBC SSA); USFWS 2020 (CPM SSA)]. SMB were purposefully introduced into Upper Basin reservoirs more than 50 years ago and were considered rare in rivers until the early 1990s when populations expanded rapidly due to multiple factors, including escapement from reservoirs and a period of low, stable flows which were likely suitable for reproduction (Breton et al. 2014; related information reviewed in Appendix 2). Ongoing management efforts, including mechanical removal of SMB and experimental designer flows to interrupt SMB spawning, are being tested in the Upper Basin (Breton et al. 2015;

Bestgen 2018; Bestgen and Hill 2016) to reduce SMB populations and increase the likelihood of HBC and other native fish recovery. A fundamental uncertainty related to management downstream from Glen Canyon Dam is whether the Upper Basin SMB paradigm applies to the CRe.

In the CRe, different experimental policies have been used to understand factors that influence HBC population recovery. These include the mechanical removal of non-native rainbow trout near a HBC population center at the Little Colorado River (LCR) confluence (Coggins et al. 2011), designer fluctuating flows from GCD to suppress rainbow trout recruitment (Korman et al. 2011, 2012), and designer stable flows to improve HBC recruitment (Dodrill et al. 2015; Finch et al. 2016). While it is unclear if or how these management actions contributed to HBC population recovery, the current total HBC population in the CRe is likely 10-20 times larger than in the early 2000s (Coggins et al. 2006; Yackulic et al. 2014; Van Haverbeke et al. 2017; Van Haverbeke et al. 2023).

Warming the mainstem Colorado River was identified as essential for HBC to spawn in the mainstem Colorado River (USDOI 1994; Trammell et al. 2002; USDOI 2008), and lab and modeling studies have suggested HBC could benefit from warm water through improvements in growth rate (Clarkson and Childs 2004; Petersen and Paukert 2005). Because of declining water storage in Lake Powell, the temperature of reservoir releases during summer and fall increased in 2004, 2005, and after 2020 (Fig. 1). Warming reservoir releases has likely driven significant ecological changes in the Colorado River ecosystem, including recovery and expansion of native fish populations (Coggins et al. 2011; Van Haverbeke et al. 2017; Yackulic et al. 2018), increased potential for range expansion of non-native warm water species (Dibble et al. 2021), and declines in cold-water non-natives (Korman et al. 2016; Korman et al. 2022). The temperature of reservoir releases directly determines water temperature in the Colorado River immediately downstream from GCD, and the extent of additional warming further downstream is primarily determined by the volume of released water. Less warming occurs when more water is released (Wright et al. 2009).

Lake Powell water storage is primarily determined by basin-scale natural runoff, consumptive use upstream from Lake Powell that reduces the amount of natural runoff that reaches the reservoir, and the amount of water released downstream. Basin-wide consumptive use has exceeded supply for most of the 21st century (Schmidt et al. 2023). Future changes in basin-wide water use and inflows, mainly prompted by more than two decades of the Millennium Drought, are difficult to predict. Still, most near-term predictions are for Lake Powell levels to remain low (Wheeler et al. 2022), increasing the risk of entrainment of SMB and other nonnative fish into a warming Colorado River ecosystem.

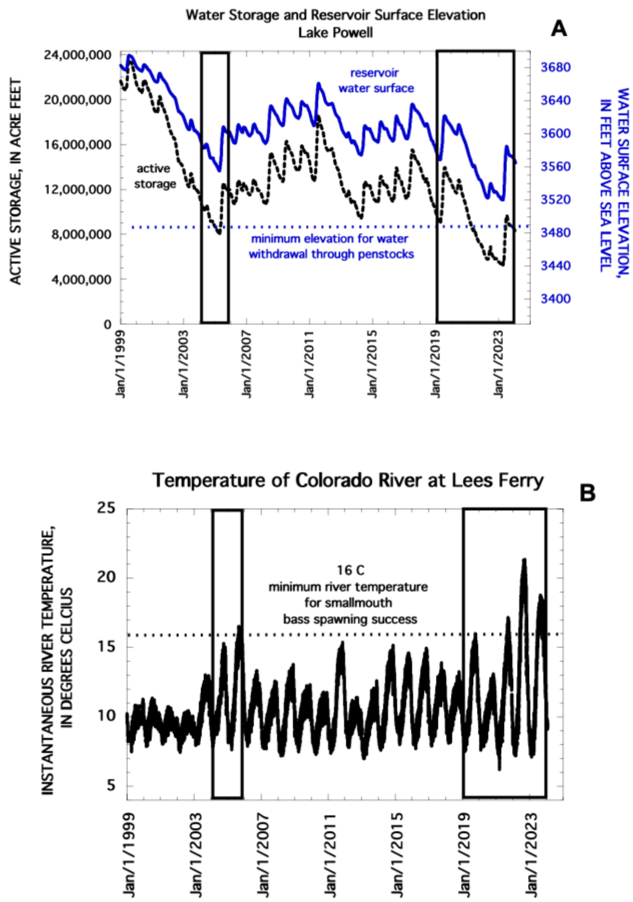


Figure 1. A. Water storage (black line) and reservoir elevation (blue line) for Lake Powell with rectangles indicating the period water temperatures in B at Lees Ferry exceeded 16C.

Section 2: A new crisis?

The SMB situation in the CRe occurs after more than two decades of research and experimental management actions to address an earlier perceived crisis between HBC and non-native trout (mostly rainbow trout and also brown trout *Salmo trutta*). Rainbow trout predation and competition with HBC were initially considered significant reasons for the decline of the HBC population in the CRe (Coggins et al. 2011; Yard et al. 2011). Subsequent research has shown that environmental conditions, including turbidity, water temperature, and prey availability driven by the concentration of phosphorous in GCD releases, drove rainbow trout abundance in Glen Canyon and near the LCR (Korman et al. 2016, 2021). These studies also showed that the effects of a 4-year mechanical removal effort near the LCR had an uncertain impact on the abundance of rainbow trout relative to these other factors associated with the warm water releases of 2005 (Fig. 1). Positive HBC responses initially attributed to decreasing trout abundance may have mostly been driven by other factors including warm water (Coggins et al. 2011; Yackulic et al. 2018). This experience is relevant to the current SMB crisis and the risk to HBC populations in the CRe and is therefore reviewed below.

Rainbow trout distribution in the CRe historically centered in Glen Canyon, where cold, low turbidity water from GCD created environmental conditions suitable for rainbow trout but not HBC (Fig. 2). Changes in GCD releases in the 1990s designed to restore sand bars and recover HBC led to increases in rainbow trout recruitment and the overall population of rainbow trout in Glen Canyon (Korman et al. 2012), which likely led to rainbow trout emigrating from Glen Canyon to the Little Colorado River (LCR) inflow reach, where they negatively impacted HBC (Coggins et al. 2011; Yard et al. 2011; Yackulic et al. 2018).

During the last decade, rainbow trout populations have declined across the CRe. Reasons for this system-wide decline include changes in rainbow trout growth, which can lead to rapid decreases in survival, maturation, recruitment, and abundance. These changes in growth were driven by changes in prey availability, turbidity-influenced foraging ability, and intraspecific competition (Mckinney et al. 2001; Korman et al. 2021).

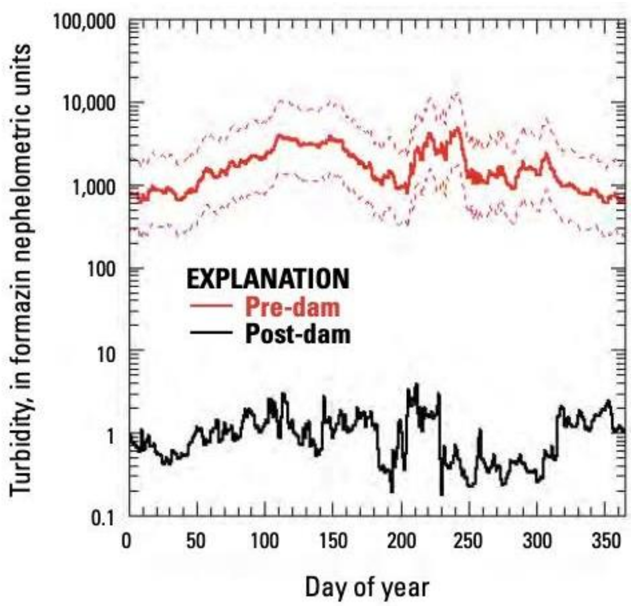


Figure 2. Graph showing geometric daily mean turbidity at the Lees Ferry gage over a year. The period of the pre-dam record (red) is 1947-1959 and was computed from daily suspended sediment data. The long-term post-dam turbidity record at Lees Ferry indicates that stream flow is typically less than 10 FNU. From Voichik and Topping (2014, Figure 12A).

Korman et al. (2021) documented differences in trout growth in five reaches of the CRe between GCD and the LCR confluence, which ranged from perennially clear in Glen Canyon to increasingly turbid in Marble Canyon (Figure 3). In addition to biotic factors, reduced trout growth and declines in abundance were likely caused by reductions in trout feeding efficiency, due to fine sediment input from the Paria River during the summer/fall monsoon season (July-October, Topping et al. 2000), and higher turbidity downstream of the LCR primarily from floods in the LCR (Figure 4). Ward et al. (2014) and Ward and Vaage (2018) used lab trials to document how turbidity as low as 25 FNU reduces the predation vulnerability of native Colorado River basin fishes to rainbow and brown trout and warm-water non-native predators, including SMB. Yard et al. (2011) documented a higher incidence of piscivory for trout during periods of higher turbidity, possibly related to increased abundance of fish prey (including native fish) during monsoon events, which transport sediment and native fish from the LCR to the mainstem. The results of these field assessments (Yard et al. 2016; Korman et al. 2021) and lab experiments support the idea that turbidity may influence site-feeding predator growth and survival and predator-prey dynamics between non-native and native fish species in the Colorado River basin. Combined, field and lab studies suggest that in the short term, turbidity could lead to higher predation on native species by non-natives. This short-term increase in piscivory could be because native and non-native species may move to shoreline habitats when turbidity levels increase or that lab studies have higher levels of predation because the fish are confined. Still, the current assessments of rainbow trout populations in the CRe demonstrate that turbidity may reduce the number of non-native species present in the long term, which benefits native fish.

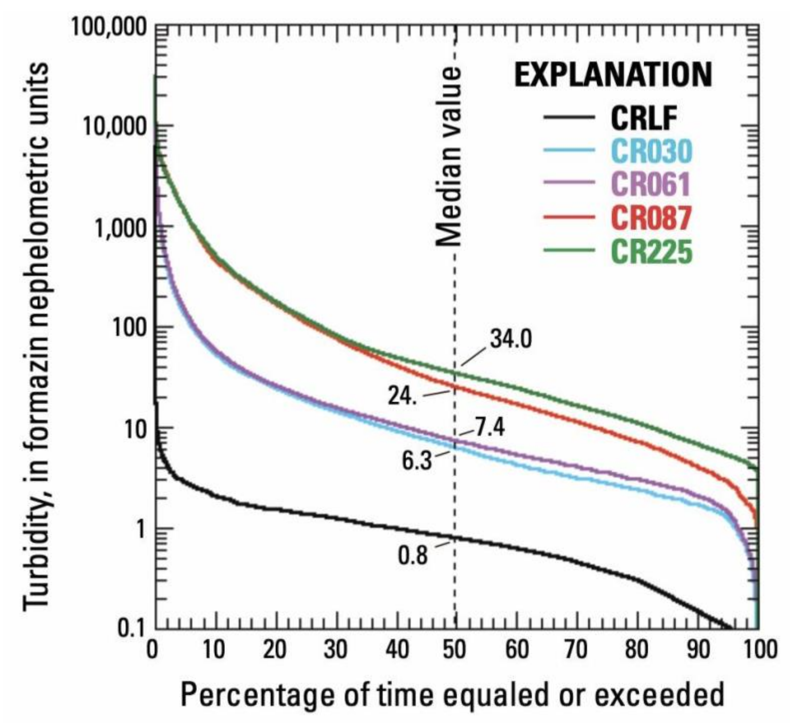


Figure 3. Graph showing the percentage of time indicated turbidity occurred between January 2008 and fall 2012 at five gaging stations throughout Marble and Grand Canyons. RM shows the locations of each gage, and the gage at RM61 is located just upstream from the Little Colorado River. During this period,

turbidity at the two gages in Marble Canyon (RM30 and RM61) exceeded 6.3 and 7.4 FNU half the time. Turbidity at the Grand Canyon gage (RM 87) and downstream from the LCR exceeded 24 half the time and exceeded 34 FNU half the time at the gage just upstream from Diamond Creek (RM 225). From Voichik and Topping (2014, Figure 13B).

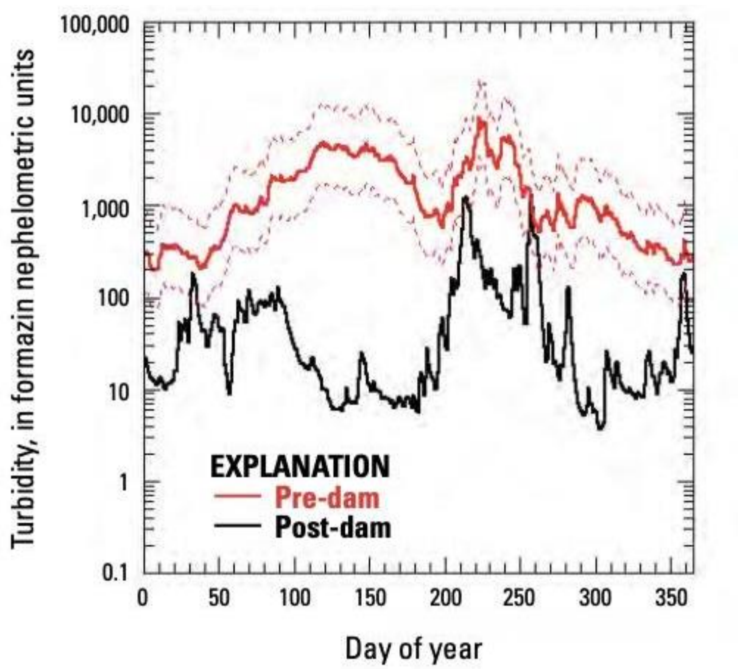


Figure 4. The graph shows the geometric daily mean turbidity at the Grand Canyon gage near Bright Angel Creek (RM 87) and downstream from the LCR for over a year. The period of the pre-dam record is 1947-1959 and was computed from daily suspended sediment data. The long-term post-dam turbidity record at this gage indicates that turbidity typically exceeds 100 FNU during the summer/fall monsoon season. From Voichik and Topping (2014, Figure 12B).

Recent work by Dibble et al. (2021) documented how changing climate and water storage decisions in the Colorado River basin may have contributed to and could further facilitate nonnative fish population expansion, including SMB, by creating thermal regimes that favor native and warm-water non-native species. Although these thermal conditions may favor native and warm-water non-native species, the native species may already be declining due to negative interactions with warm-water non-native species, including SMB, and this perspective is the dominant SMB-native fish paradigm in the Upper Basin.

Eppehimer et al. (in-review) document adult SMB catches from 2011-2023 of 0 to 3 fish in Glen Canyon (Lees Ferry), with a catch of 2 in 2022, which followed catches of 0 in 2020 and 2021. While these catches are low, they have occurred for more than a decade. There is also evidence of local SMB reproduction in a backwater-like habitat in Glen Canyon (https://tinyurl.com/4uz4ewyw). Multiple sampling efforts have captured SMB at different times and sizes, suggesting entrainment and local reproduction are occurring.

Eppehimer et al. (in-review) developed a detailed model linking GCD storage and operations, water temperatures, and SMB population dynamics to accurately predict adult SMB catch downstream from GCD in 2022 and 2023. The authors developed a matrix population model parameterized by age-specific survival, fecundity, and growth information for SMB from the Upper Basin (see supplemental files in Eppehimer et al. (in-review)) to predict SMB population growth rates at three locations in the CRe (Lees Ferry, LCR confluence, Diamond Creek confluence) using predicted water temperatures under different water management and basin-level hydrologic scenarios at these three locations. The matrix model used in the predictions of SMB population is parameterized by SMB demographic and life history parameters from the Upper Basin, then uses water temperature predictions to forecast SMB population growth. A key assumption in this approach is that the age-specific survival, fecundity, and growth information from the Upper Basin for SMB is realistic for Glen Canyon, where turbidity is typically very low. These life-history parameters could differ further downstream in Grand Canyon, where high, seasonal turbidity is typical, including the LCR confluence and west-central and western Grand Canyon (LCR and Diamond Creek confluence) where HBC abundance is much higher than in the Upper Basin (Figure 3).

Section 3: Adaptive management, risk tolerance, and establishing competing SMB paradigms in the CRe

Within an adaptive management program, Williams et al. (2012), in the DOI Adaptive Management Applications Guide, describe the development of alternative predictive models as a way to "…represent uncertainty (or disagreement) about the resource system." Williams et al. (2012) describe how "Agreements, disagreements, and uncertainties about resource behaviors can be incorporated in models and used to guide investigations through basic research and learning-oriented management interventions." This adaptive management approach has been a core aspect of learning in the CRe as part of the Glen Canyon Dam Adaptive Management Program (GCDAMP) since 1997. Adaptive management and related structured decision making approaches provide a deliberate process for identifying uncertainties, resolving disagreements, and improving resource management related to the risks SMB populations may present to HBC, the likelihood of SMB population establishment at different locations in the CRe, and experimental management actions to improve the ability to manage CRe resources and the effects of dam operations under the Grand Canyon Protection Act.

The risks SMB poses to HBC population viability in the CRe are uncertain. Still, the dominant paradigm taken from the Upper Basin and applied to the CRe is that the risks are significant, and SMB will increase the likelihood of HBC extinction. Characterizing and understanding these risks can inform management decisions related to SMB in the CRe. For example, suppose the risk of potential impacts on HBC populations from SMB ranges from 0 to 1, with a value of 0 indicating no increase in the risk of extinction for HBC populations and a value of 1 indicating an inevitable HBC extinction because of SMB predation. In that case, stakeholders can develop management actions to reduce this risk. Absent a deliberate introduction of SMB and monitoring of the interactions between SMB and HBC populations, other information, including studies of native and non-native SMB populations, other non-native species in the CRe, and predictive models developed from this information can be used to characterize the risk and inform management actions taken to reduce risk.

A stakeholder's risk tolerance and values can influence whether to advocate a management action. For example, a risk-averse stakeholder may see the growth of SMB populations over time in the CRe as highly probable and irreversible. Therefore, the existence of any SMB in the CRe should be considered an existential threat to HBC populations. For such a risk-averse stakeholder, management actions to control SMB should be implemented immediately to minimize (or eliminate) SMB below GCD and in Glen Canyon. A more risk-tolerant stakeholder may prefer to wait until empirical evidence is available that SMB abundance is increasing or likely to increase towards a level that could negatively impact HBC before deciding whether management actions are needed. Some stakeholders may be risk averse because of their charge as an agency – such as management requirements related to the Endangered Species Act. The values of decision-makers also influence risk tolerance. For example, a stakeholder who highly values native fish conservation may perceive any increased risk of HBC extinction negatively, and this stakeholder would place high value on management actions to minimize any risk to HBC populations by SMB.

What controls SMB distribution in the CRe?

The SMB task force identified factors necessary for SMB to develop locally reproducing populations in the CRe (https://tinyurl.com/23dc5fy7) that include sufficient prey resources, infrequent high turbidity events, low-velocity gravel/cobble habitat, and suitable water temperatures. We agree that the first three factors are met in most years in Glen Canyon, and appropriate water temperatures have occurred since 2022 (Figure 1). Temperature and turbidity were previously identified as essential factors for native and non-native species, and both can be influenced by management actions (https://tinyurl.com/yc4cxesj).

Based on these factors, at least two distinct paradigms can be compared, describing the potential for SMB invasion, persistence, and expansion in the CRe. The first is a temperature-only paradigm, which assumes that water temperature is the only limiting factor in SMB expansion in the CRe. This temperature-only paradigm guides models used in Dibble et al. (2021) and Eppehimer et al. (in-review). Under this temperature-only model, SMB in the CRe, introduced via entrainment from Lake Powell, local reproduction, or other ways, are only limited in their distribution in the CRe by water temperature. All other conditions and factors identified by the SMB task force above are assumed not to limit SMB expansion.

A second paradigm, based on the factors described by the SMB task force list, is a temperature+turbidity model, which assumes that the distribution of self-sustaining SMB populations in the CRe is limited by water temperature and the frequency of high turbidity events. The temperature+turbidity paradigm recognizes the critical role of temperature but also builds on rainbow trout research in the CRe, highlighting the role of turbidity in impeding rainbow trout feeding, reducing growth, and ultimately impacting survival and recruitment. Because rainbow trout and SMB have similar visual reactive distances to turbidity (Sweka and Hartman 2003), the temperature+turbidity paradigm assumes that turbidity events identified as negatively impacting rainbow trout will also negatively impact SMB. This alternate paradigm does not mean that turbidity would limit the existence of SMB; instead, much like rainbow trout, this paradigm would suggest that temperature+turbidity combined may influence SMB populations in different locations of the CRe such that in some years (and locations) SMB populations may grow and survive when temperatures are warm and turbidity events infrequent and in other years when these conditions are not met SMB populations growth or survival will be lower.

Under the temperature+turbidity paradigm, turbidity conditions that could prevent the development of self-sustaining SMB populations are only found downstream of Glen Canyon because of turbidity inputs from the Paria River and LCR. Critically, the two population centers for HBC, near the LCR confluence and in west-central and western Grand Canyon (WGC), experience more frequent high turbidity levels likely to impair rainbow trout or SMB reactive distance to prey (~ 50 FNU). By some estimates, the west-central and western Grand Canyon population of HBC which have expanded dramatically in recent years to nearly 60,000 adult fish (Van Haverbeke et al. 2017; Van Haverbeke et al. 2023) or perhaps 100,000 (2024 GCMRC annual planning meeting), is ~ 4- to 6-fold larger than the population of ~10,000 adults near the Little Colorado River confluence (Yackulic et al. 2014). Because HBC populations are much larger now than during the early 2000s when rainbow trout predation impacts on HBC populations were of significant concern, the risk of HBC extinction in the short term (< 10 years) is lower because the humpback chub populations are larger.

The temperature-only paradigm predicts a relatively high risk to HBC below the LCR from SMB because only a single factor, temperature, is considered limiting. Stakeholders would favor this paradigm if risk-aversion to SMB were high because it would suggest the potential for invasion and persistence of SMB is high. The temperature+turbidity paradigm is more risk tolerant because two factors, temperature and turbidity, are considered limiting factors. More risk-tolerant managers may give this paradigm more consideration because it assumes that persistent sources of SMB are only likely to persist in areas used by HBC if both water temperature is high and high turbidity events are infrequent, such as the conditions below GCD. However, these conditions are less likely to be met downstream, where high turbidity events increase due to Paria and LCR inputs.

These two paradigms are models, descriptions of how the world might work, and these models can be used to assess how SMB populations may grow in different locations or how these populations may respond to management actions. Currently, the temperature-only model is described by a mathematical model in Eppehimer et al. (in-review) that predicts SMB population growth rate at three locations in the CRe (Lees Ferry, LCR confluence, Diamond Creek confluence). The matrix model used to make these predictions is informed by SMB life history parameters (see supplemental files in Eppehimer et al., "Population growth model details" section) such as adult and subadult age-specific survival, fecundity and growth from recent and ongoing studies in the Yampa, Green, and White rivers; all in the Upper Basin. These life history parameters are assumed to be the same for SMB populations at each of the three locations in the CRe where the Eppehimer et al. (in-review) model predicts SMB population growth rate (Lees Ferry, LCR confluence, Diamond Creek confluence).

Under the temperature+turbidity hypothesis, key life history parameters, including growth and survival, could differ in locations that experience frequent high turbidity (i.e., west-central and western Grand Canyon downstream of the LCR) from areas that do not experience high turbidity events (i.e., Yampa River, Glen Canyon). A revised version of the Eppehimer et al. (in-review) model that includes turbidity and other limiting factors could be used to evaluate the risk of SMB expansion in the CRe. Understanding this risk is necessary to choose whether or not to take management actions, or how extreme an action to take, to change the risk of SMB establishment or population growth in the CRe.

Support for the temperature+turbidity hypothesis

Rainbow trout and SMB are visual feeders, and because the reactive distances to prey for rainbow trout and SMB are similar (Sweka and Hartman 2003), SMB feeding could likely be impacted by periodic turbidity events downstream of the LCR. Ward and Vaage (2018) and Albrecht et al. (2017) discuss the potential for high turbidity to function as a "habitat feature," which may have facilitated juvenile HBC and razorback suckers to recruit in the Lake Powell and Mead inflows even though both lakes support large numbers of warm-water predators elsewhere in the lakes where turbidity is low.

Observations in Albrecht et al. (2017) and Ward and Vaage (2018) highlighting how turbidity may be facilitating native fish recruitment in the Lake Mead inflow or function as a type of habitat highlight the need to evaluate turbidity as a limiting factor for SMB at the LCR and in the west-central and western Grand Canyon population centers. SMB dispersing from a self-sustaining population in Glen Canyon could persist near the LCR confluence during periods of clear water, creating predation risk for HBC. Like the LCR confluence, SMB could persist in west-central and western Grand Canyon during periods without high turbidity events. However, high turbidity events occur most years downstream of the LCR (Figures 4, and 7). Under the temperature+turbidity hypothesis, the frequency and duration of both conditions will strongly influence SMB reactive distance to prey, growth, survival, and recruitment, which would drive the population growth rate of SMB in ways similar to rainbow trout (Korman et al. 2021). Under this paradigm, the abundance of SMB populations downstream of the LCR will be limited by a combination of temperature and turbidity conditions.

Water temperature in the CRe increases with distance downstream from GCD, and warming water will favor SMB and other warm-water species, including native fish (Figure 1A, Figure 5). Water temperature is identified as the probable driver for both HBC population growth and expansion in the Colorado River ecosystem (Van Haverbeke et al. 2017) and for the potential invasion and expansion of SMB (Dibble et al. 2020; Eppehimer et al. in-review). But critically, are HBC and SMB predicted to overlap in space in the Colorado River ecosystem? For example, if SMB become established in Glen Canyon, where there is no HBC population center, and SMB does not become established near the LCR or in western Grand Canyon, where there are HBC population centers, are SMB simply another non-native sportfish found in Glen Canyon? The potential for this spatial overlap is important to understand the risk SMB establishment presents to HBC populations.

SMB captures in western Grand Canyon downstream from Diamond Creek have occurred occasionally since 2005-2006 (map available slide 18 http://tinyurl.com/3zuj8x4v), and a simple graph of temperature data suggests water temperature near Diamond Creek has been suitable for SMB spawning since the late 1990s. However, SMB populations have not expanded in this reach despite occasional captures, a source population in Lake Mead (Albrecht et al. 2017), and suitable temperatures. This lack of expansion suggests other factors may have limited SMB invasion from Lake Mead or other sources in recent decades.

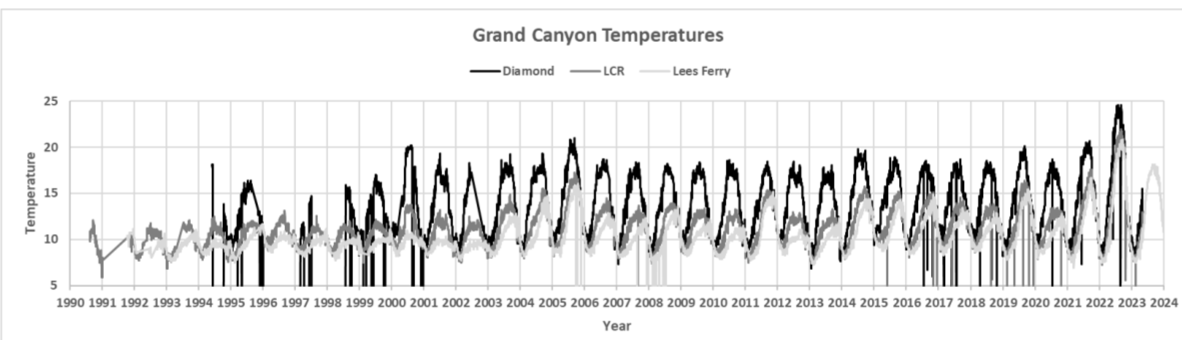


Figure 5. Water temperature at Diamond Creek, LCR, and Lees Ferry from 1990-2023.

The lack of SMB population growth in western Grand Canyon despite suitable temperature conditions, occasional catches, and a source population in Lake Mead suggests factors other than temperature may be limiting. Based on observations with rainbow trout and similarities between rainbow trout and SMB reactive distance to prey, we think turbidity should be considered as another key factor constraining SMB population growth in Grand Canyon. Increasing turbidity can lead to strong reductions in SMB reactive distance to prey

(Figure 6). This reactive distance reported for SMB is similar to the values reviewed for rainbow trout in Sweka and Hartman (2003). Turbidity can exceed these levels in the mainstem Colorado River (Korman et al. 2016; Figures 4 and 7) and in the Little Colorado River (Stone et al. 2010; Stone et al. 2018; Korman et al. 2016), and these high turbidity conditions are more frequent in west-central and western Grand Canyon (Figure 3). Feeding impairment from turbidity can affect multiple SMB life stages, such as reducing adult fish body condition leading into spawning from spring runoff-related turbidity or impairing juvenile life-stage condition, growth, and survival during the first summer of life from monsoon-related turbidity spikes. The failure of SMB populations to expand in the reaches above and below Pearce Ferry over the last decade when water temperatures have been suitable, indicates that turbidity may be affecting SMB feeding efficiency and the risk of SMB establishment. **We conclude that both temperature and turbidity factors will likely play critical roles in how SMB colonize and persist in the CRe downstream of the LCR, and thus the risk SMB present to HBC extinction. The GCD-AMP should consider the temperature+turbidity and temperature-only hypotheses as part of the adaptive management process.**

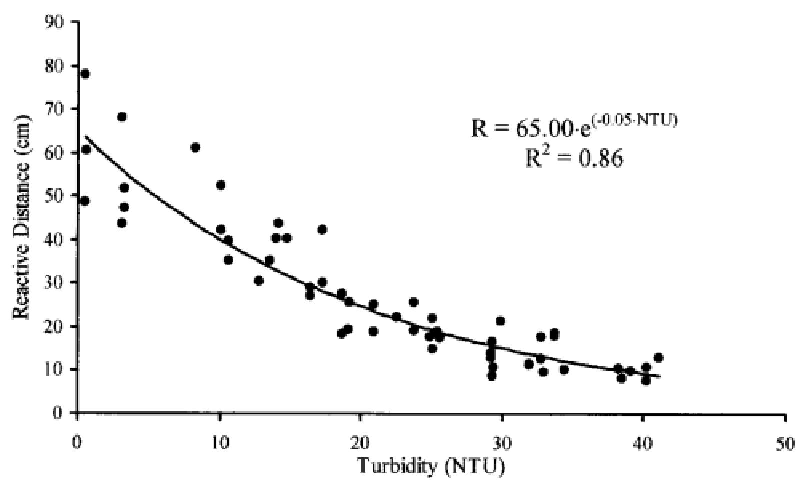


Figure 6. Graphic from Sweka and Hartman (2003) SMB experiments measuring the reactive distance to prey.

To demonstrate how often high and low turbidity conditions have occurred in recent years, we downloaded data from the GCMRC server and plotted the number of days with turbidity ≥ 50 FNU or ≥ 500 FNU each year from 2008-2023 (Figure 7). These plots indicate that there are weeks in most years when turbidity levels are high enough near the LCR and Diamond Creek to reduce SMB feeding efficiency significantly. As shown in Figures 7 and 8 (see also Appendix 4), periods of high turbidity have occurred in late summer and fall in most years since 2002, as driven by monsoonal rainfall patterns. Small SMB juveniles may be susceptible to high turbidity and low visual reactive distance to prey during summer and fall, when SMB normally grow rapidly before the winter. Monsoon season periods of high turbidity are rare in the upper basin areas where SMB have successfully invaded, and populations expanded (based on the available turbidity record on the Yampa River). This difference between the CRe and the Upper Basin implies that population responses in the Upper Basin may be poor predictors of responses in the CRe below the LCR. GCMRC sediment monitoring expertise could be integrated with ecological observations of how turbidity may influence fish reactive distance to prey to refine the frequency of these conditions observed at different locations in the CRe.

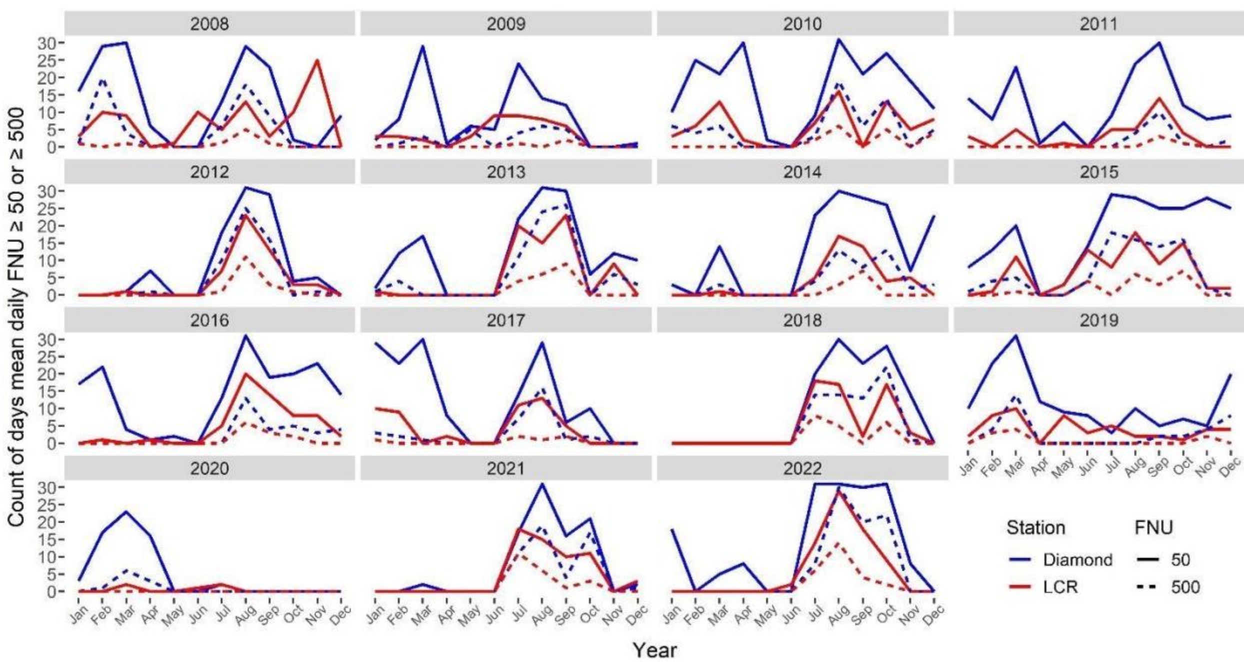


Figure 7. This figure shows the number of days each month (y-axis) the mean daily turbidity at either Diamond Creek (blue line) or the Little Colorado River (red line) is ≥ 50 FNU (solid line) or ≥ 500 FNU (dashed line) from 2008-2022. Each panel is an individual year, and then the x-axis is the months in that year. Data near the Little Colorado River are for USGS gage 09383100 (Colorado River above Little Colorado River near Desert View), and data in western Grand Canyon are for USGS gage 09404200 (Colorado River above Diamond Creek near Peach Springs). Data downloaded at https://www.gcmrc.gov/discharge_qw_sediment/stations/GCDAMP.

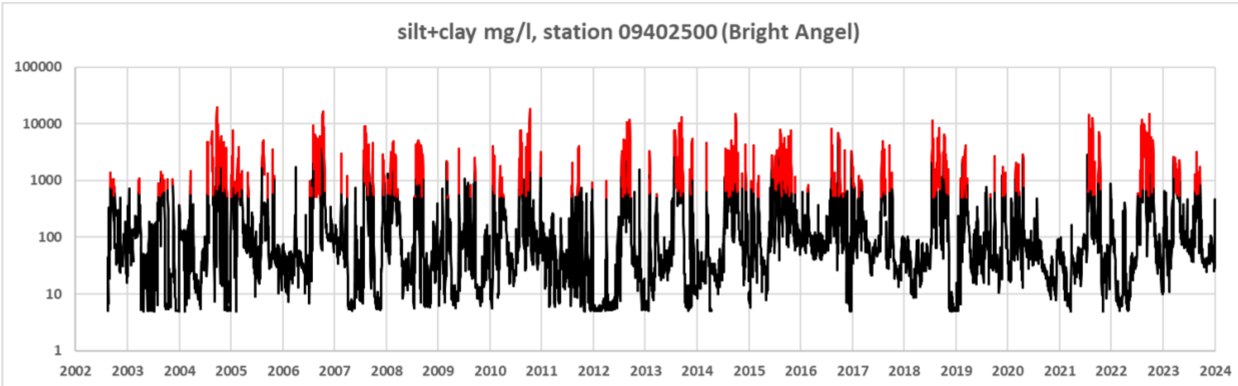


Figure 8. Daily concentration (x-axis) of silt+clay (mg/l, y-axis, log10 scale) near Bright Angel Creek (USGS Station 09402500). The black line shows average daily values from sampling every 15 minutes, and the red lines show observations greater than 500 mg/l. Data downloaded at https://www.gcmrc.gov/discharge_qw_sediment/stations/GCDAMP.

Conclusion: The risk of SMB becoming established may be very different between the Upper Basin, Lees Ferry (Glen Canyon), and the two HBC population centers in the CRE near the LCR and in west-central and western Grand Canyon. Characterizing this risk is essential to understanding the short- and long-term threat level of SMB to HBC populations in the CRe and informing short- and long-term management actions. We recognize that this is not the dominant paradigm and that even suggesting this alternative will create disagreement among scientists and GCD-AMP stakeholders. Disagreement is common in complex resource management decisions (Williams et al. 2012). The GCD-AMP program has used elements of adaptive management (Melis et al. 2015) and structured-decision making to address other complicated issues, including management of non-native rainbow and brown trout (Runge et al. 2018) and long-term experimental planning (Runge et al. 2015). Multiple assessments have reviewed the application of adaptive management for threatened and endangered species and whether the "risks of harm" associated with implementing an adaptive management program are too high to undertake. Runge et al. (2011) highlight this clearly: "...as long as the short-term risks of learning are explicitly embedded in the decision analysis, an adaptive approach will be compatible with endangered species management, and indeed, will identify the smartest course of action that accounts for management objectives in the face of uncertainty. Note that the smartest course of action, in an adaptive framework, may be *not* to pursue learning, if the risks associated with that learning are too high. But that conclusion can only be reached from full decision analysis of the problem in a context that considers the appropriate value of learning in an adaptive setting."

Section 4: Risk tolerance and the need to evaluate management actions for SMB in the CRe

Understanding the source of SMB in the CRe

Based on experience with non-native species in many places (see Appendix) and in the Upper Basin, where repeated introductions of adult SMB from reservoir populations is considered an important reason for their successful invasion (Breton et al. 2015), understanding the source of SMB can help to inform management actions to limit SMB populations in the CRe. Because the source population of SMB in Glen Canyon is entrainment from Lake Powell or local reproduction, then management options for SMB could include changes in dam operations, such as different flow options or using the river outlets to alter water temperature and impact SMB populations near GCD. However, the effectiveness of these actions is likely to decline due to discharge and temperature attenuation in west-central or western Grand Canyon.

In the early 2000s, understanding rainbow trout source populations in the CRe was identified as information needed to inform trout management decisions related to interactions between non-native rainbow trout and HBC. This work was critical to identifying (1) when rainbow trout abundance is high in the Lees Ferry reach, then more trout will immigrate to the LCR reach to find a habitat that is not occupied by rainbow trout, (2) the magnitude of immigration from upstream sources, and prey availability and turbidity conditions determined rainbow trout abundance below the LCR. These same key factors are likely relevant to SMB as well.

Evidence for SMB entrainment comes from the TRGD project and its precursor, the Natal Origins (NO) project, an intensive sampling program designed to understand the effects of GCD operations and other factors on the population dynamics of rainbow and brown trout in Glen Canyon. Currently, the project samples two 3-km reaches four times yearly (spring, fall, late fall, and winter). Reach 1A (river mile -14.3 to -12.5 mile ending upstream of the -12.5 "slough" habitat) and reach 1C (-4.3 to – 2.7 mile, upstream of the 4-mile bar to upstream of Fall Canyon) are intended to represent conditions in the upper and lower portions of Glen Canyon. Reach 1C is also sampled in the summer for five trips. The combined shoreline length of the two reaches represents ~ 25% of the total shoreline length in Glen Canyon. Overall, SMB catch from these sampling efforts has been low. The spatial distribution of SMB catch in the TRGD data shows that SMB catch is higher in reach 1A (-14 to -12.5 mile) than in 1C (at -4 mile). Within reach 1A, SMB catch is highest in the upper half of 1A, upstream of honey draw, which is ~ 1 mile from GCD. Overall, the SMB TRGD project catch data shows an upstream-to-downstream decline in SMB catch, which is also shown by the monitoring from Arizona Game and Fish Department in Glen Canyon. The TRGD data also show SMB catches are highest during fall when reservoir elevations are lowest, and water temperatures are highest. The size-frequency and catch data do not suggest these fish grow significantly over winter or that abundance accumulates over successive years when entrainment has occurred (see section on mechanical removal). Local reproduction of SMB is likely occurring in or near the -12.5 mile slough downstream of reach 1A, and adult SMB (>200 mm) have also been collected near GCD (https://tinyurl.com/4uz4ewyw).

As a low-gradient, shallow, and wide tailwater reach with very low turbidity, SMB populations will probably increase in Lees Ferry (Glen Canyon) if temperature conditions remain favorable. Glen Canyon is similar in geomorphology (shallow, wide) and turbidity to Upper Basin reaches in the Yampa and Green rivers, where SMB populations rapidly expanded in the 1990s and early 2000s following a period of low, warm, and stable river levels. This reach also supports a suitable prey base for SMB, including Gammarus and juvenile rainbow and brown trout. An established Glen Canyon SMB population may serve as a source of downstream SMB recruits into Marble Canyon and possibly the LCR inflow reach. The establishment of a reproducing SMB population is significant because if SMB are unable to establish self-sustaining populations outside of Glen Canyon, under the temperature-only hypothesis, regular inputs of recruits from Glen Canyon may support SMB in downstream locations, creating a predation risk that could lead to population declines of HBC in downstream reaches, even if local SMB recruitment is not occurring. Under the temperature-turbidity hypothesis, the persistence of SMB populations near the LCR inflow reach is less likely because of the effects of high turbidity events impacting SMB in similar ways as observed for rainbow trout (Korman et al. 2021).

Section 5: SMB invasion and potential risks to HBC populations in the CRe have motivated the identification of SMB management alternatives

The potential for SMB to successfully invade and establish viable populations in the CRe and for these populations to expand, interact with, and negatively impact HBC populations is a significant management concern in the CRe that has motivated response actions by members of the GCD-AMP. One action that has been tested is mechanical removal for SMB (example from Glen Canyon National Recreation Area, https://tinyurl.com/4uz4ewyw), and mechanical removal has been ongoing in the Upper Basin for decades (see Appendix). Additionally, six short-term management alternatives to reduce the risk of establishment and population expansion of SMB in the CRe are described here (https://tinyurl.com/2p2ea6nd). These management actions are part of a supplemental EIS to the 2016 Long-Term Experimental and Management Plan Final Environmental Impact Statement (LTEMP ROD), which includes no action, disrupting SMB spawning through releasing water through the penstocks and river outlets (cool mix), cool mix with flow spikes to disrupt SMB spawning (up to 45,000 CFS), short-duration cold shock using the river outlets, cold-shock with flow spikes, and non-bypass alternative focusing on large changes in river stage to disrupt spawning of SMB, with the intent of causing year-class failure and reducing the rate of population growth. These and other short-term management alternatives will likely inform the long-term options to be included in the post-2026 EIS for GCD.

Section 6: Learning from the past – mechanical removal, fluctuating flows, and effects of temperature on non-native rainbow trout

Finding: The past actions used to control rainbow trout in the CRe are similar to the currently proposed actions to control SMB. Neither mechanical removal nor increased flow fluctuations to reduce rainbow trout population year-class strength proved helpful as management options. Changing the GCD flow regime – increasing hydropeaking and releasing controlled floods to disrupt the spawning of a rainbow trout population in Glen Canyon has affected trout populations, but not in the ways anticipated and not necessarily to the benefit of native species. Prior experience with rainbow trout and uncertainty over SMB population responses is similar to earlier uncertainties in the GCDAMP program related to the efficacy of non-native fish management actions, which were examined as part of an adaptive management program. While the experimental actions of the adaptive management program may not have delivered the desired outcome, this approach resulted in adaptive learning and likely better river management policies (Melis et al. 2015).

Non-native fish management has been a significant component of Grand Canyon fisheries programs for over 20 years. In January 2003, the GCDAMP began a multi-year experimental program to evaluate policies to recover HBC by improving rearing conditions in the mainstem Colorado River. At that time, there was significant concern related to HBC population viability and how the complicated interactions among non-native rainbow trout and brown trout, river temperature caused by cool water releases from Lake Powell's hypolimnion, and hydropeaking had impacted HBC populations and limited population recovery (Coggins et al. 2005). In the early 2000s, three management actions were considered that might help the juvenile native fish population in the mainstem Colorado River: (1) reducing non-native fish abundance, primarily rainbow trout, (2) changing hydropeaking patterns, and (3) developing technologies to increase water temperatures released by GCD thought to be too cold for native fish spawning and juvenile growth and survival in the mainstem.

Coggins et al. (2011) assessed how native and non-native fish species responded to mechanical removal near the LCR confluence from 2003 to 2006. They demonstrated that rainbow trout populations declined during this period and that the decline was primarily not caused by mechanical removal. The decline in rainbow trout abundance was slightly greater in the removal reach than in the control reach (where rainbow trout were not removed).

Korman et al. (2011) examined the effectiveness of increasing hourly discharge fluctuations from GCD during winter (called "nonnative suppression flows") to reduce rainbow trout recruitment in Lees Ferry. They found that flow fluctuations dewatered 25-50% of the redds (trout spawning nests). Still, the overall recruitment of age-0 rainbow trout was not reduced due to compensatory improvement in the survival of the remaining juveniles. The rainbow trout population then increased in 2008 following an early March High Flow Experiment (HFE) of 42,800 ft^3/s that lasted three days, with the increase in rainbow trout likely due to the presumed "cleaning" of the tailwater's gravel bed during the flood, thereby increasing food availability for young trout after the recession of the flood. A similar response was not observed after the spring 2023 flood, 1.5 months after the 2008 HFE, when piscivorous brown trout were much more abundant, when water temperature was much higher, and when dissolved oxygen levels were low. The effects of spring HFEs on rainbow trout in Glen Canyon is likely highly dependent on a range of conditions, many of which have changed since the first spring HFE observed in 2008.

Rainbow trout research in Glen, Marble, and eastern Grand Canyon near the LCR confluence (Korman et al. 2016; Korman et al. 2021) documented the factors that had caused a system-wide decline in rainbow trout between 2012 and 2015; those factors included reduced immigration of young trout from the Glen Canyon tailwaters upstream from Lees Ferry and lower prey availability, higher turbidity, and warmer water which reduced survival rates for older rainbow trout. Temperature and sediment regimes were similar between 2003 and 2006, suggesting that mechanical removal had a marginal effect on reducing trout populations during this period. Korman et al. (2016, 2021) also demonstrated that the rainbow trout population near the LCR (1) was driven by immigration from upstream, (2) had substantially lower apparent survival rates than fish upstream of the LCR confluence, and (3) had declined systemwide substantively reducing their potential interaction with HBC population. Despite the well-documented dynamics explaining the system-wide decline in trout during the last decade (Korman et al. 2016, 2021), rainbow trout are still classified as a high threat to the Colorado River ecosystem in a recent agency assessment (http://tinyurl.com/5fxvryn9).

We conclude that during the last decade, rainbow trout populations downstream of GCD declined for multiple reasons -- reduced prey supply, sustained periods of high turbidity downstream from Lees Ferry, and warmer water temperatures (Korman et al. 2021; Yard et al. 2023) and that mechanical removal of rainbow trout played a minor role in the decline of rainbow trout. During this same decade, HBC populations have increased.

Section 7: Evaluating the potential for electrofishing mechanical removal to reduce the risk of SMB establishment in Glen Canyon

FINDINGS: Our findings suggest that the mechanical removal of SMB below GCD and in Glen Canyon is unlikely to stop SMB population growth in the short or long term. However, mechanical removal can reduce or the SMB population growth rate if capture probabilities and removal effort are high, and when entrainment from Lake Powell and local reproduction are low.

Mechanical removal of SMB in Glen Canyon using boat electrofishing is one of the methods being evaluated by GCD-AMP stakeholders to reduce the risk of establishment in the Colorado River downstream of GCD. The removal approach, implemented in 2022 and 2023, involves repeatedly electrofishing the shoreline of Glen Canyon and removing all SMB and other undesirable warmwater non-native fish captured. The approach's efficacy in controlling the

SMB population using mechanical removal in Glen Canyon depends on three factors. First, the proportion of a population within a discrete section of shoreline, say a 250 m length, removed by a single pass of electrofishing effort, must be high enough to deplete the population at a site. This proportion is typically called capture probability in the fisheries and mark-recapture literature. Second, a substantive portion of the entire shoreline available for SMB to rear and reproduce must be electrofished repeatedly if the at-a-site capture probability is low, which it typically is when sampling in a large river. This may require high levels of effort, such as weekly multi-boat trips, for a significant proportion of the year. The annual removal rate will increase with the average capture probability across sites, the proportion of the total shoreline sampled for each trip, and the number of trips per year. Finally, the annual removal rate has to be high enough to overwhelm the population growth rate. For SMB in Glen Canyon, this includes the effects of both entrainment and local reproduction (successful spawning and juvenile recruitment) downstream of GCD. If the population growth rate (often termed lambda, λ) exceeds the annual removal rate, the population will grow even with ongoing removals. In a worst-case scenario, a high-effort removal program would have no tangible effect on the trajectory of the SMB population if the population growth rate is high relative to the removal rate.

We analyzed SMB and Green Sunfish (GSF) data collected from the Trout Recruitment and Growth Dynamics (TRGD) project in Glen Canyon between June 2022 and November 2023. We use these data to estimate the capture probability of SMB and GSF. We include the latter species because it has a similar size distribution to SMB but is much more abundant, thereby providing a more reliable estimate of capture probability than can be derived using SMB data. The increasing abundance of GSF in Glen Canyon may also concern managers, warranting a better understanding of the effects of a removal program on this species. We estimate the annual removal rate of GSF and SMB under different levels of effort and effects on population trajectories under a range of assumed population growth rates.

Methods

The TRGD project and its precursor, the Natal Origins (NO) project, is an intensive sampling effort to understand the effects of GCD operations and other factors on the population dynamics of Rainbow Trout and Brown Trout in Glen Canyon. In its current form, the project samples two 3-km reaches four times per year (spring, fall, late fall, winter). Reach 1A (river mile -14.3 to -12.5 mile with its downstream end located just upstream of the slough) and 1C (-4.3 to – 2.7 mile, upstream of 4-mile bar to upstream of Fall Canyon) are intended to represent conditions in the upper and lower portions of Glen Canyon. Reach 1C is also sampled in the summer for five trips. Each reach is broken up into 24 250-m shoreline sites (12 on river left and 12 on river right). Thus, a total of 6 km of shoreline are sampled in each reach per trip. The combined shoreline length of the two reaches represents ~ 25% of the total shoreline length in Glen Canyon.

During a trip, all sites in each reach are sampled twice (two passes of electrofishing effort over two consecutive nights). All RBT, BNT, and flannelmouth suckers ≥75 mm are PITtagged and released. Warmwater non-natives, such as SMB and GSF, are removed as directed by permitting requirements and Grand Canyon fish sampling protocols. The number and size of SMB and GSF caught at each site are recorded for each pass, and the catch across the 24 sites sampled in each reach are summed for each pass.

The change in the total catch of SMB (or GSF) across the 24 sites on passes 1 and 2 within a reach can be used to estimate capture probability (proportion of fish caught per pass of electrofishing effort) and abundance using a “depletion” model. This model is best understood by simple equations describing how catch and abundance change during successive passes. Prior to the first pass, the population size in a reach can be represented by variable N_0, the unfished abundance. Given a capture probability p, catch on the first pass (C_1) is calculated as,

$$C_1 = No*p$$

Thus, the abundance after pass 1 will be,

$$N_1 = N_0 - C_1$$

The catch on pass 2 and the remaining population after pass 2 is calculated as,

$$C_2 = N_1*p$$

$$N_2 = N_1 - C_1$$

The extent to which the population in the sampling reach is depleted (i.e., N_0-N_2) depends on the capture probability. If p is high, catch on the first and second passes will be larger and the 2 passes of removal effort will result in a larger depletion of the population compared to the situation where p is low. Note that this model assumes capture probability does not change across passes. As described below, this is not likely to be true but variation in capture probability cannot be reliably estimated when only a limited number of passes are conducted or catches are relatively low

Multiple pass depletion experiments often suggest higher initial capture probability for the first pass than for subsequent passes (see e.g. Speas et al 2004; Peterson et al. 2004). So it is likely that the results presented below are actually biased upward due to such initial high capture rates of the most vulnerable individuals. Variation in environmental conditions (flow, wind, rain), equipment, or crew, can result in differences in capture probability among passes. Individual variation in fish behaviour and habitat selection can also result in differences in p among passes. For example, individuals located in habitats where they are more vulnerable to capture by boat electrofishing (e.g., in shallow water closer to shore) are more likely to be caught on the first pass, resulting in relatively fewer of these more vulnerable individuals available for capture on the second pass. In this situation we would expect p to be lower on the second pass. Not accounting for this dynamic can result in an overestimate of p for the population as a whole and an underestimation of the unfished population size, N_0.

As another warning about possible overestimation of capture probabilities for SMB, the estimates actually apply only to the proportion of the SMB population that is resident or holding close enough to shore for the fish to be vulnerable to electrofishing. SMB may well move offshore into deeper eddies and pools during winter, in order to conserve energy when food availability is lower. Anecdotally, fishermen know to expect such behavior of smallmouths in rivers (see e.g., https://tinyurl.com/6sxx6htm). Low catches in 2022 and 2023 in winter samples could also be due to such vulnerability changes, rather than to high mortality rates.

Depletion models are fit to the data by assuming that the observed catches on each pass are random variables drawn from a statistical distribution, such as the binomial,

$C_{ip} \sim$ binomial (p, N_{ip-1})

where C_{ip} represents the catch on a pass(ip=1) or 2 (ip=2), and N represents the abundance prior to the pass (N_0 if ip=1, N_1 if ip=2). The statistical model recognizes that there is sampling error in the data. Consider a box containing 100 balls, of which 10 are marked. Blindfolded students are then asked to draw 10 balls from the box and to calculate their estimate of the proportion of marked balls in the box given knowledge of the total number of balls. This proportion would be equivalent to the capture probability described above. The true proportion of marked balls in the box is 0.1 (10/100). Across the students, we would expect that on average, 1 of the 10 balls drawn from the box will be marked, and the average of the estimated proportion across students would thus be 0.1. However, it would not be surprising if some students calculated a proportion of zero (if they were unlucky and did not draw any marked balls), while others would draw more than one ball and estimate a $p > 0.1$. The binomial sampling distribution is used to calculate precision in the estimate of this proportion (capture probability). The precision will depend on the number of marked balls that are drawn. In context of the two-pass depletion experiment described above, the total number of balls represents the unknown number of fish in the site, and p represents the capture probability from a pass of effort. When catch from each pass is relatively small, because capture probability or abundance is low, there will be high uncertainty (low precision) in the estimate of capture probability.

A critical assumption of the binomial error distribution is that each trial is independent. In the example of marked balls in a box, this assumption holds if the marked balls in the box are well-mixed with the unmarked balls. However, if the marked balls are clustered in one part of the box, the assumption of independent trials no longer holds, and the precision in the estimate of the proportion will be overestimated. This same logic holds for the capture of SMB in a 250-m long shoreline site. If SMB in a site are clustered due to variation in habitat quality or other factors, we would expect higher sampling error in the catches and hence higher variation in the estimate of capture probability than determined from the binomial sampling distribution. Owing to this issue we assumed that sampling error is more realistically represented by a Poisson distribution.

$C_{ip} \sim$ poisson $(p*N_{ip-1})$

where the product in brackets represents the expected catch on a given pass. This distribution will result in uncertainty estimates for p that are similar to those from the binomial if C is high but will result in greater uncertainty in p when C is low (either because p or N is lower).

We estimated p and N_0 for each trip and reach combination where the total catch of a given species across two passes exceeded zero. We restricted the analysis to eight trips conducted between September 2022 and November 2023 when GSF and SMB catches were significantly higher compared to earlier trips. Higher catches result in more precise estimates of capture probability. Trip-reach cases with no catch on either pass contain no information about capture and were therefore excluded. Reach 1C was sampled on 8 trips between June 2022 and November 2023, while 1A was sampled on 6 trips because it was not sampled during the summer for logistical reasons. This sampling regime results in a maximum of 14 estimates of capture probability for all trip-reach combinations. Estimates of p and N_0 for each trip-reach combination are represented by p_i and N_{0i}, where i is an index for each trip-reach combination.

We fit independent and hierarchical Bayesian (HBM) depletion models to the data. The independent model assumes the estimates of p_i are independent of each other and depend only on the data from each trip-reach case. This model does not require additional assumptions, but estimates of p can be highly uncertain when catches are low, which is the case for almost all SMB estimates and for some GSF estimates. To address this data sparsity issue, a hierarchical version of the model assumes the p's across the trip-reaches are exchangeable. That is, we assume the p's are random variables arising from a common normal distribution (in logit space) whose mean and standard deviation are estimated,

$$\text{logit}(p_i) \sim \text{normal}(mu, sd)$$

The estimated mean of this capture probability hyper distribution and its uncertainty are used to represent the expected capture probability of future removal efforts. The mean is essentially a weighted-average of trip-reach specific estimates with the weights determined by the precision of each estimate. In trip-reach cases when the data on capture probability are informative (high catch), the independent and HBM estimates will be similar. When this is not the case, the trip-reach estimates will be shrunken towards the mean of the hyper distribution and have less of an effect on mu and sd in the equation, above compared to cases with more information about p_i.

We applied our model to data for rainbow trout (RBT), brown trout (BNT), GSF and SMB data. There were 14 estimates of capture probability for RBT, BNT, and GSF across all trip-reach combinations, but only 11 for SMB as there were a few cases where there was no catch on either pass. We fit depletion models to RBT and BNT data to compare the depletion model-based estimates of capture probability to the more reliably determined capture probability estimates from mark-recapture modeling. This analysis, in conjunction with comparisons of the two estimators of capture probability in the literature, is used to calculate an adjustment to SMB and GSF depletion-based capture probability estimates. Adjusted and unadjusted SMB capture probability estimates are then used to estimate annual removal rates under different levels of removal effort.

Results

<u>Capture Probability</u>

Catch of GSF during TRGD trips began increasing in the summer of 2022, with catch in 2023 considerably higher than in 2022 (Fig. 1). The average capture probability across trip-reaches from independent estimates was ~ 0.2. Estimates for individual trip-reach combinations generally had large uncertainty, especially for cases when catch was low. The trip-reach capture probability estimates from the hierarchical model were slightly more certain owing to the additional information from the exchangeability assumption. As expected, in cases when the data were sparse, the trip-reach p estimates were shrunken towards the estimated mean of the p hyper distribution. The mean of the p hyper distribution was 0.09 with 95% credible intervals of 0.0 – 0.22.

SMB in Glen Canyon are still relatively rare, resulting in very low total catch in the TRGD reaches (Fig. 2). There is very limited information about capture probability for SMB owing to low catches, which results in high sampling error in the model. As a result, most of the 95% credible intervals for the capture probability estimates from the independent model ranged from about 0.01 to 0.6. The mean of the hyper distribution for capture probability distribution from the HBM was 0.11, similar to the value for GSF, but much more uncertain (0.01 – 0.34). That is, the expected capture probability per pass could be as low as 1% to as high as 34%. Owing to the sparse data, there was considerable shrinkage in most trip-reach estimates of p towards the mean. Our best estimate for average capture probability for SMB of 0.11 is close to the estimate of 0.12 derived from the NPS depletion experiments in 2022 (B. Healy, pers. comm.).

Assuming estimates of capture probability are unbiased (more on this below), average GSF abundance across trips based on the HBM was~ 3,900 and 390 in reaches 1A and 1C, respectively. GSF abundance was ~ 10-fold higher in the reach closer to the dam, located upstream of the slough (Fig. 3a). Average abundance of SMB was 182 and 28 in 1A and 1C, respectively. SMB abundance in 1A was ~ 6-fold higher than in 1C (Fig. 3b). GSF abundance was 21-fold higher than SMB abundance in 1A, and 14-fold higher in 1C.

Estimated capture probabilities for RBT and BNT were much higher compared to those for GSF and SMB (Fig.'s 4 and 5). Owing to much larger sample sizes, the precision of capture probability estimates was also higher, resulting in less statistical shrinkage. The mean of the hyper-distributions of capture probabilities for RBT of 0.21 (0.13-0.29) and for BNT of 0.32 (0.25-0.39) were relatively well determined. A large majority of RBT and BNT fish captured since June 2022 have fork lengths greater than 275 mm. The mark recapture-based estimates of capture probability, based on a robust-design open population model, have been approximately 0.02 and 0.06 for RBT and BNT for the ≥ 275 mm size class, respectively (using a similar model to Korman and Yard 2017). The depletion-based estimators of capture probability presented here are ~ 10- and 5-fold higher than the more reliable recapture-based estimators, respectively. This indicates that the depletion-based estimators of capture probability for larger trout are substantively overestimated.

The estimate of upward bias in depletion-based estimates of capture probability for trout may not apply to GSF and SMB, because they are much smaller (Fig.'s 6 and 7) than the size of trout that were captured. It is likely that the upward bias in depletion-based estimates of p is caused by heterogeneity in habitat use. Smaller fish are more likely to use shallower water and be closer to shore and therefore may have lower variation in habitat use (e.g., some using shallow and some using deeper water) compared to larger fish. Korman et al. (2009) compared depletion- and mark-recapture based estimates of capture probability for small RBT (41-78 mm) in Glen Canyon. Estimated capture probabilities based on depletion experiments were ~ 2-fold higher than those based on closed population mark-recapture experiments. To account for the likely positive bias in depletion-based capture probability for GSF and SMB, we reduced the mean estimates of ~ 0.1 to 0.05. We used the lower 95% credible interval for the hyper-distribution of capture probability as a lower bound.

Efficacy of Removal

We calculated the proportion of the shoreline in Glen Canyon sampled during a 4-night electrofishing trip based on the following assumptions:

1. A total of 30 minutes is required to electrofish a 250-m site, process the captured fish, and setup for the next site. This estimate is in part based on the recommended time to 'slow-shock' a site to optimize capture probability for small fish (16 minutes).
2. There are 6 hours per night to sample. This estimate is optimistic, especially for summer trips when nighttime begins at ~ 9 pm (thus sampling would be completed by 3 am).
3. Two boats are used on each trip. This results in 24 250 m sites sampled each night.
4. A sampling trip consists of sampling 4 nights in a 7-day week (rig on Monday, sample Monday-Thursday nights, derig on Friday).

There are 224 250-m sites in Glen Canyon. Under assumptions 1-4, 96 sites would be sampled on a trip, equivalent to 43% of the total shoreline length of Glen Canyon. Assuming an average capture probability per site of 5% leads to an estimate that 2.1% of a targeted fish population in Glen Canyon would be removed during a weekly trip (Prem_trip=0.05 * 0.43). The annual removal rate per year (Prem_yr) can be calculated based on the number of trips conducted per year (Ntrips),

Prem_year = 1 – (1-Prem_trip)^Ntrips.

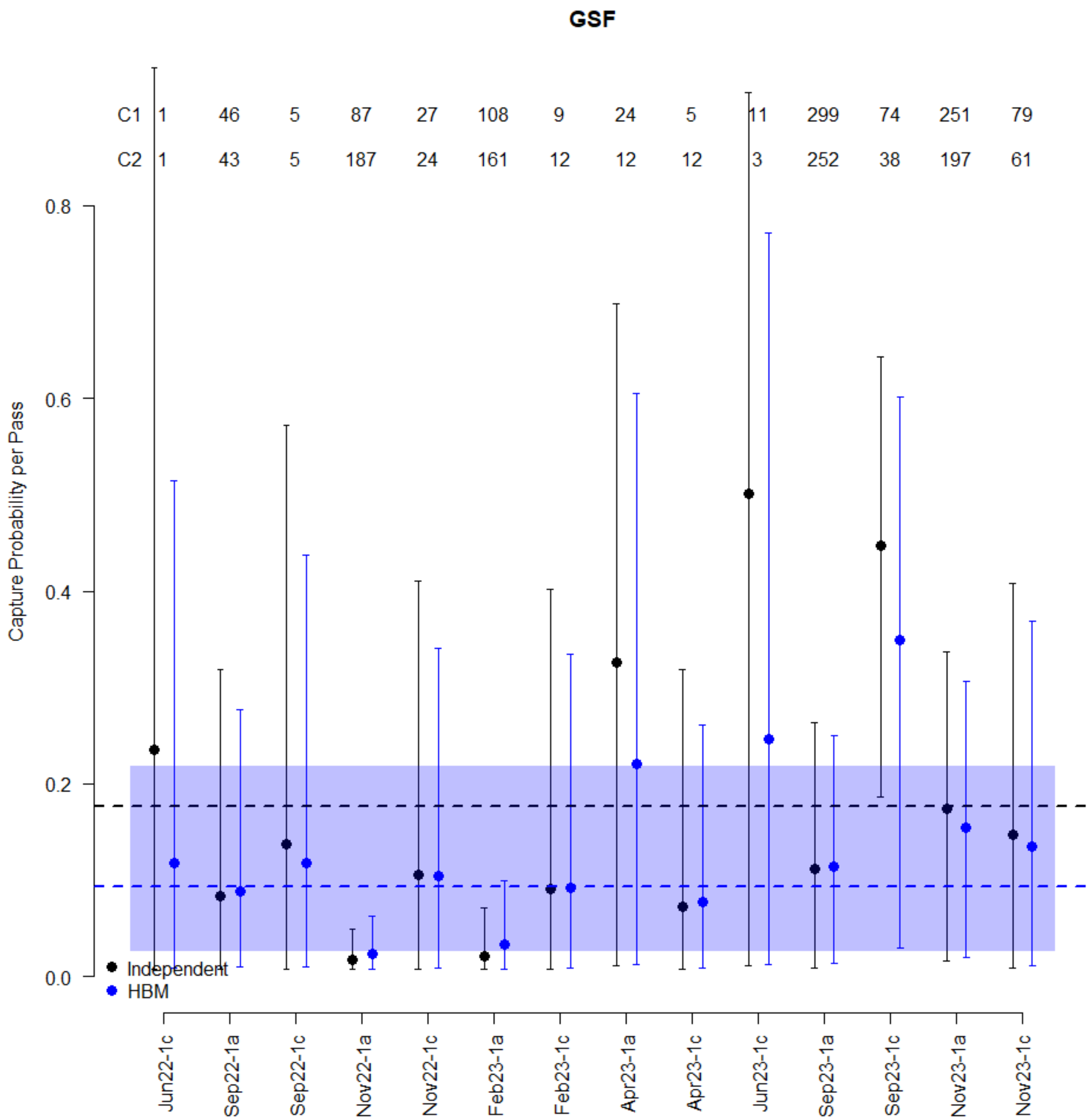
We calculated the annual removal rate under different assumptions about the average capture probability per site and effort. We considered a low-effort scenario with 4 trips per year (1 trip/month for June, July, August, and September) with two boats per trip, a moderate effort scenario with 16 trips per year (2 trips/month * 8 months) with two boats per trip, and a high effort scenario with 16 trips per year but with four boats per trip.

The percentage of a fixed population removed in a year was ~ 8, 29, and 50% for low, moderate, and high-effort scenarios given our best adjusted estimate of average site-specific capture probability of 5% (Table 1). The annual removal percentage increased to 16, 50, and 67% with a site-specific capture probability of 10% for these three scenarios (the ~unadjusted GSF/SMB estimates). At the lower 95% credible interval of the mean of the SMB capture probability hyper distribution (1%), the annual removal percentages were 2, 7, and 13% for low, moderate, and high-effort scenarios.

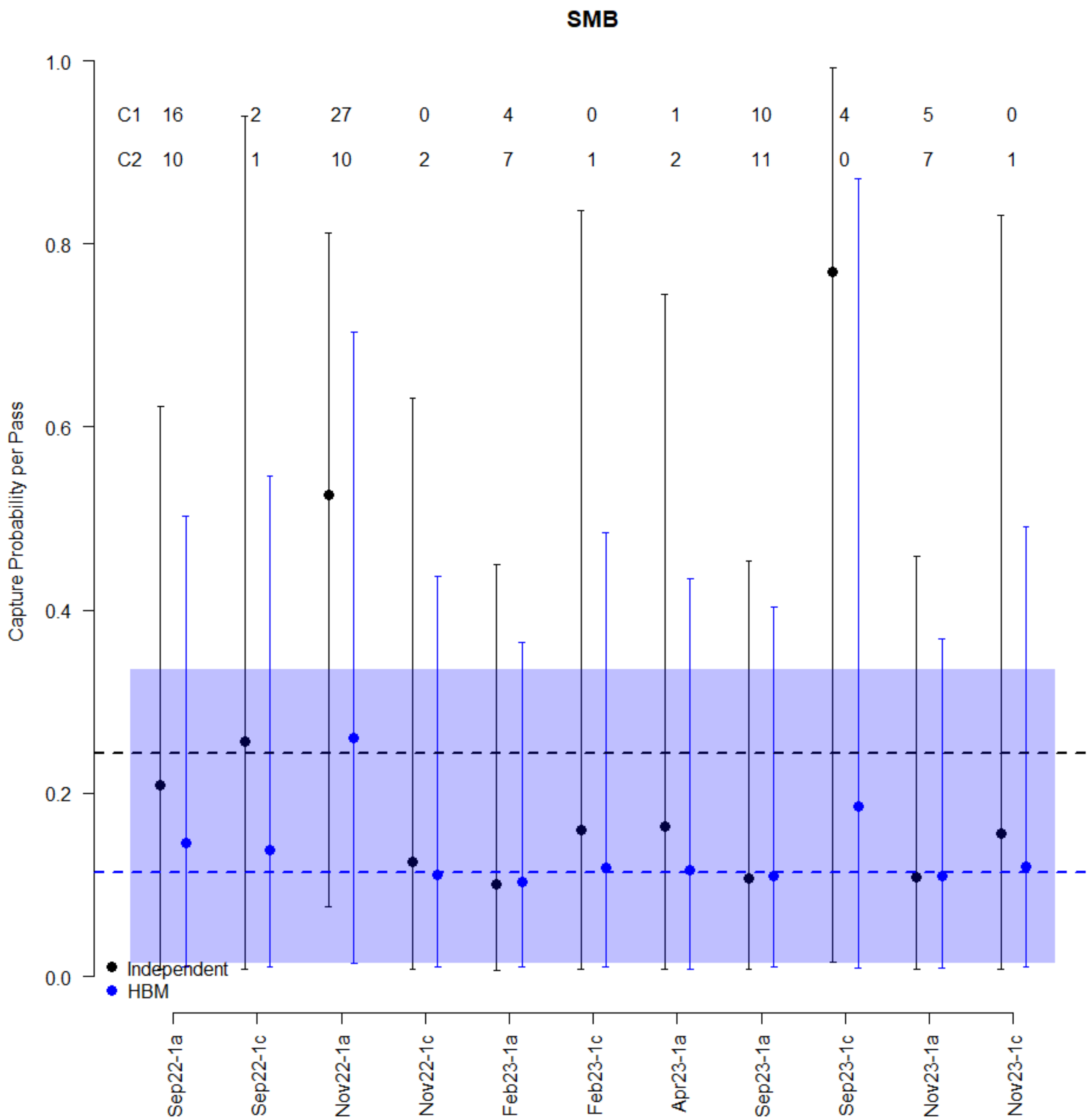
The effect of these annual removal rates on the trajectory of an SMB population in Glen Canyon depends on the annual population growth rate (Table 1d). For example, a moderate-effort removal program implemented during a five-year period would deplete the population to 0.18 of its initial size assuming the population growth rate is zero (no entrainment or local reproduction). Such a static population would only be depleted to 0.17 of its initial abundance over 5 years under a more pessimistic capture probability of 1% with a medium level of removal effort. **The effects of mechanical removal are unlikely to stop SMB population growth if a positive population growth rate is assumed, but mechanical removal can slow SMB population growth if capture probabilities are relatively high, effort is high, and local reproduction or entrainment is low.** For example, at an annual growth rate of two, (the population would double every year), a population would expand by 13-fold over five years under a 10% capture probability at low effort, and expand by ~ 6 fold based on a capture probability of 5% with moderate effort. On the other hand, using the lowest capture probability (1%) coupled with a low-effort removal program results in 39-fold population increase over five years. Removal efforts become increasingly futile with increases in the population growth rate. However, in the absence of knowledge of the future population growth rate for SMB, it is possible that removal could limit the rate of increase. **It is very likely that the benefits of removal efforts to control the trajectory of SMB abundance in Glen Canyon will be modest at best. Because of the low current catch-rates of SMB, measuring their response to any management actions will be very difficult.** The proportion of fish captured for a given sampling effort is very poorly determined. Thus, a halving or doubling of the catch could reflect a meaningful population change or could reflect sampling error. For this reason, there may be opportunities to learn about SMB by increasing studies of GSF. Questions that could be considered include how a nest-building centrarchid with higher populations, such as GSF, responds to proposed management actions as a proxy for SMB. Such studies would be a relatively low-cost effort as it would only necessitate analyzing existing data.

Table 1. Estimates of the percentage of a static SMB or GSF population removed per year based on different levels of removal effort and average capture probability at-a-site (top 4 rows). The remainder of the table shows the proportional change in population size after 5 years (relative to an initial value of 1) of removal under different levels of population productivity (lambda, representing the annual magnitude of entrainment and local reproductive inputs).

	Average Capture Probability/Site		
Removal Effort	10%	5%	1%
% Population Removed/yr (assuming no entrainment or reproduction)			
Low (4 trips/yr * 2 boats)	16.1	8.3	1.7
Moderate (16 trips/yr * 2 boats)	50.4	29.3	6.6
High (16 trips/yr * 4 boats)	76.2	50.4	12.9
Proportional Change in Population after 5 Yrs			
Lambda=0 (no population growth)			
Low effort	0.42	0.65	0.92
Moderate effort	0.03	0.18	0.71
High effort	0.00	0.03	0.50
Lambda = 2 (population doubles each year)			
Low effort	13.40	20.8	39.4
Moderate effort	0.96	5.7	22.8
High effort	0.02	0.96	16
Lambda = 2.5 (population increases by 2.5-fold each year			
Low effort	40.6	63.3	89.6
Moderate effort	1.9	16.3	69.4
High effort	0.08	2.93	49

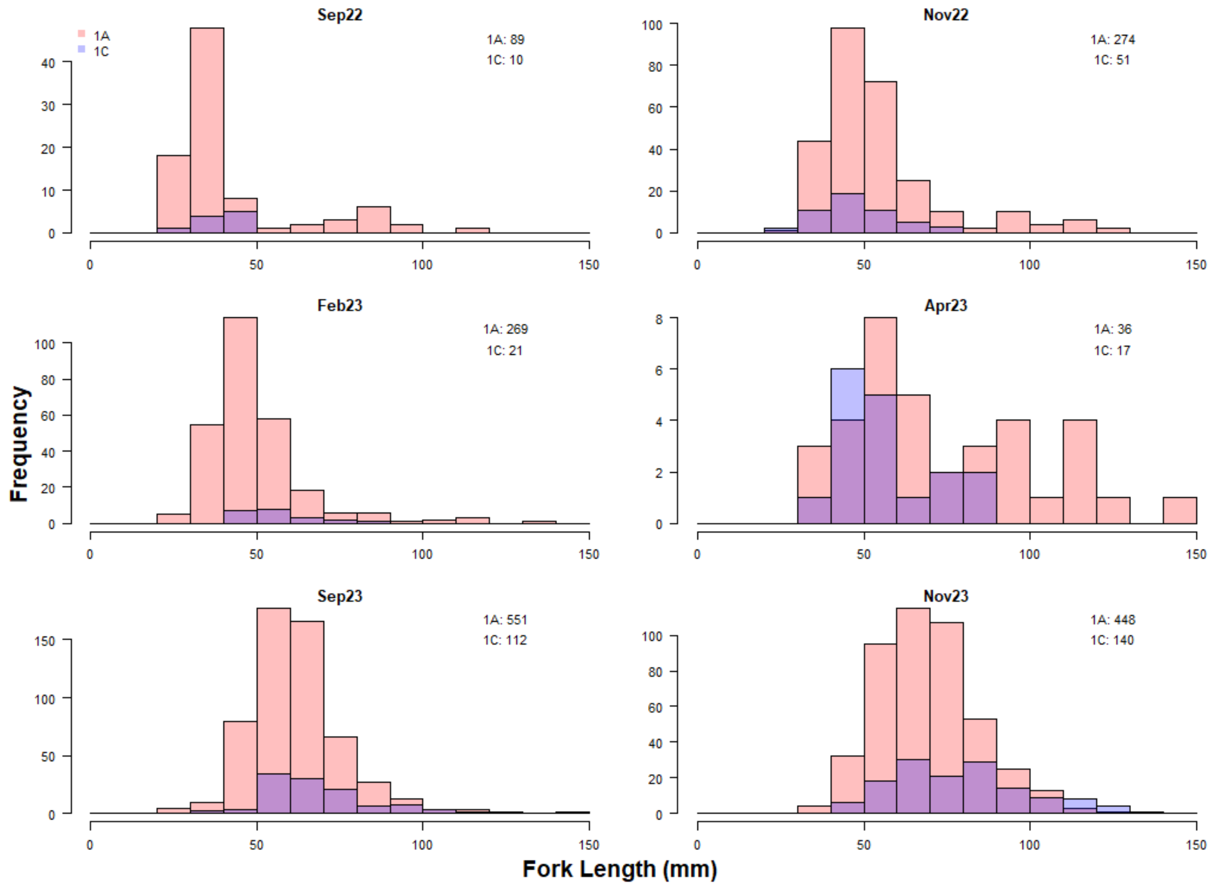


Removal Figure 1. Capture probability estimates by TRGD trip and reach for Green Sunfish (GSF). The black points and black error bars represent the mean and 95% credible intervals of independent estimates of capture probability, while the blue points and error bars represent estimates from the Hierarchical Bayesian model HBM). The horizontal black and blue dashed lines represent the mean of the independent estimates, and the mean of the hyper distribution of capture probability for the HBM. The shaded blue box represents the 95% credible interval of the mean of the capture probability hyper distribution. The text at the top of the plot shows the catches on passes 1 (C1) and 2 (C2).



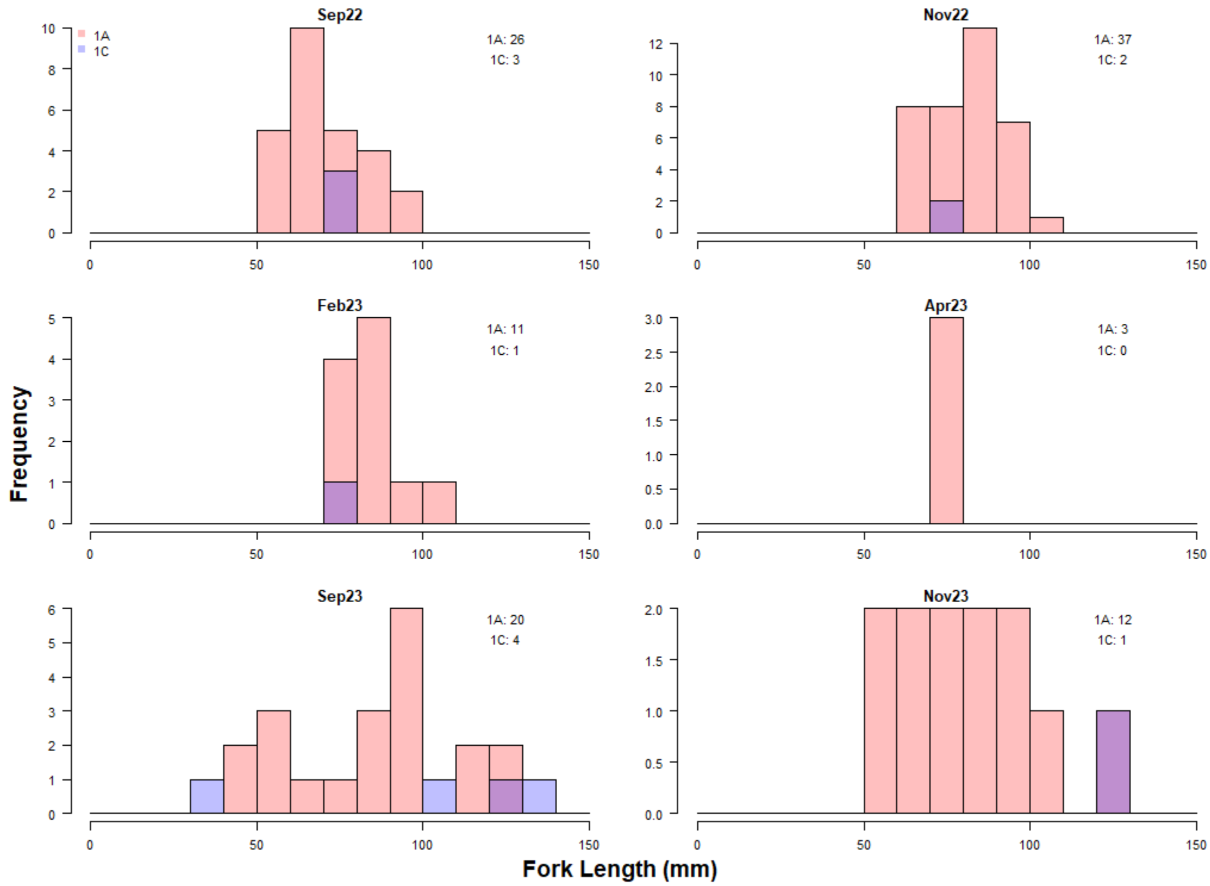
Removal Figure 2. Capture probability estimates by TRGD trip and reach for Smallmouth Bass (SMB). See caption for figure 1 for additional details.

a) Green Sunfish

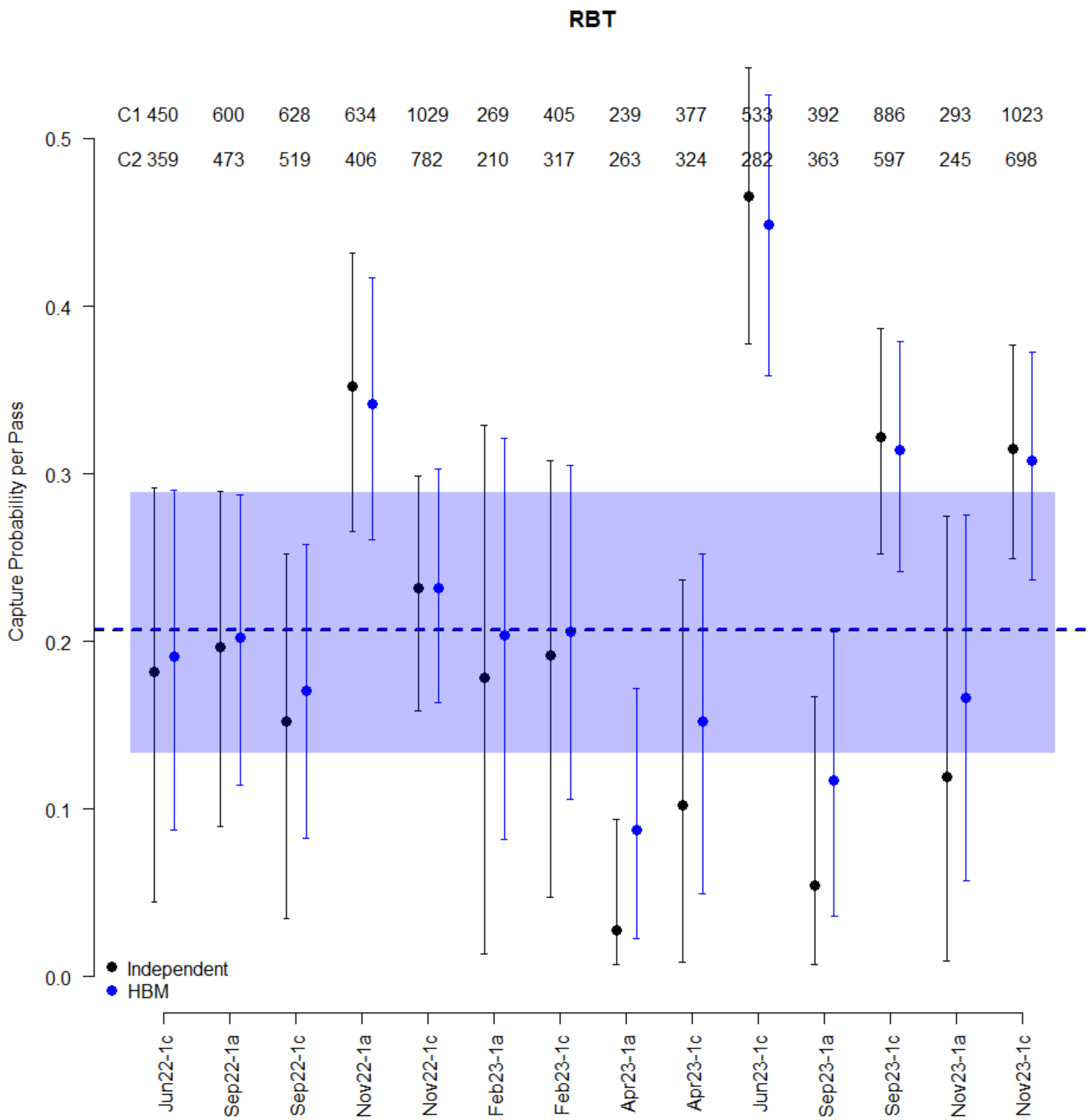


Removal Figure 3. Comparisons of Length-frequency histograms of Green Sunfish (a) and SMB (b) between reaches 1A and 1C on trips when both reaches were sampled. Text in the upper-right of each panel shows the total catch for by reach for each trip.

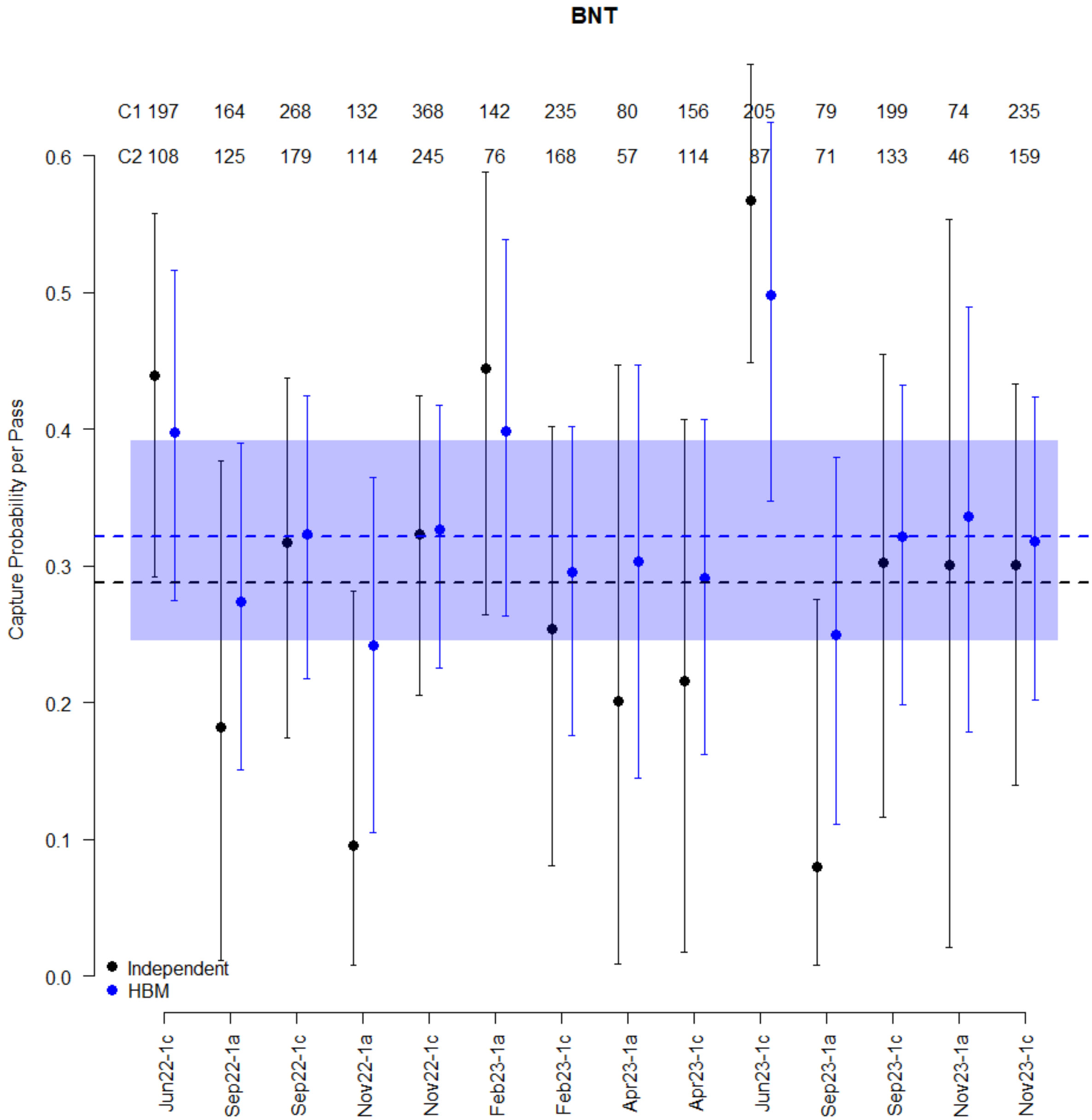
b) Smallmouth Bass



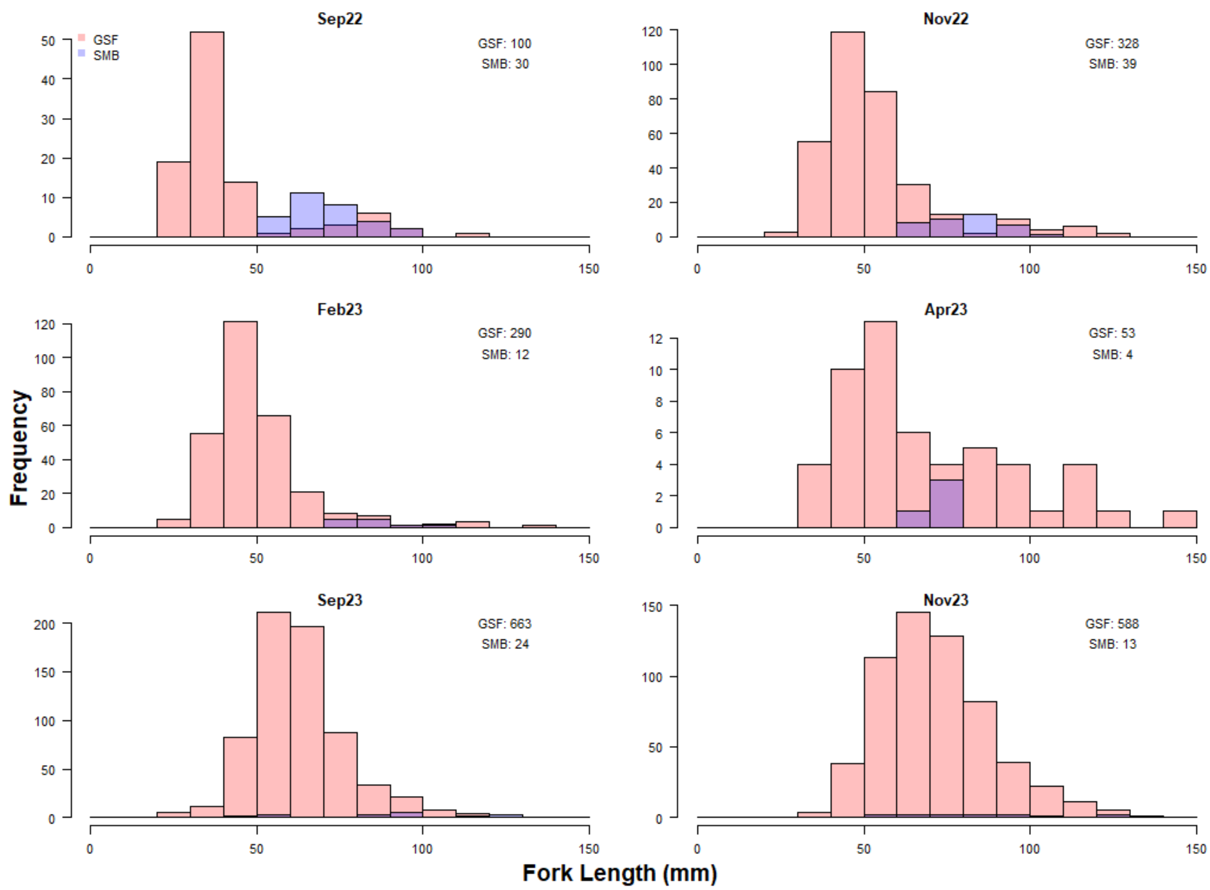
Removal Figure 3. Continued from above.



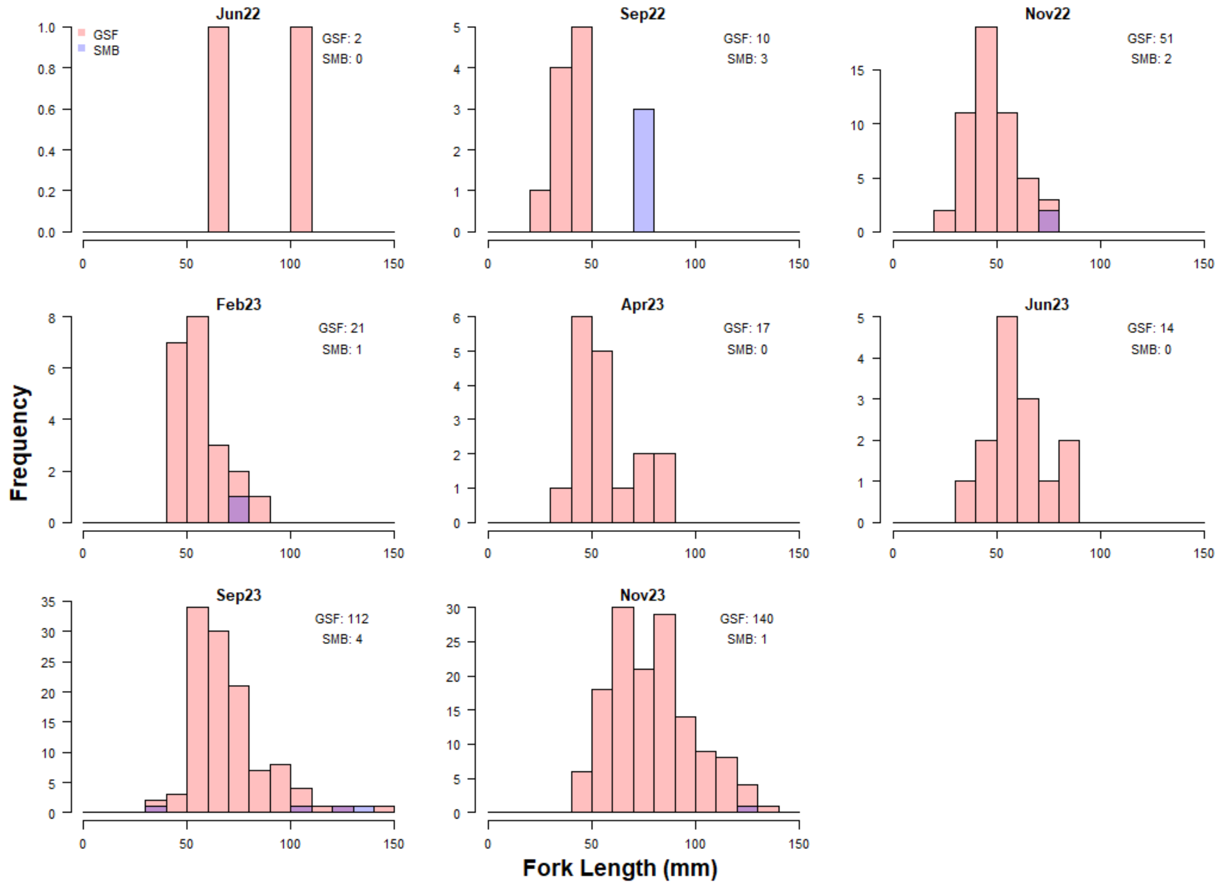
Removal Figure 4. Capture probability estimates by TRGD trip and reach for Rainbow Trout (RBT). Unlike SMB and GSF data, most RBT caught on pass 1 are marked and release. To meet assumptions of a depletion model consistent with the one used for SMB and GSF, RBT catches on pass 2 do not include fish that were marked on pass 1 and recaptured on pass 2. In addition, catch on pass 2 does also not include the estimate fish not marked on pass 1 that were caught on pass 2. See caption for figure 1 for additional details.



Removal Figure 5. Capture probability estimates by TRGD trip and reach for Brown Trout (BNT). See caption for Removal Figure 4 for additional details.



Removal Figure 6. Length-frequency histograms of Green Sunfish (GSF) and Smallmouth Bass (SMB) across TRGD trips using combined catches from reaches 1A and 1C. Data from June 2022 and 2023 trips are not included because only reach 1C was sampled on these trips. Text in the upper-right of each panel shows the total catch for each species.



Removal Figure 7. Length-frequency histograms of Green Sunfish (GSF) and Smallmouth Bass (SMB) across TRGD trips using data from reach 1C which was sampled in June as well as the other months when 1A was sampled. Text in the upper-right of each panel shows the total catch for each species.

Section 8: Using flow fluctuations to reduce non-native fish recruitment – SMB potential responses

FINDING: The efficacy of using short-duration high reservoir releases from GCD (up to 45,000 CFS are proposed) or the non-bypass alternative of large river stage changes (low and then high releases) to reduce SMB abundance via nest disruption is uncertain. This is because it is unknown whether interrupting spawning will lead to year-class failure and lower overall SMB recruitment. More information is needed to evaluate a similar program for fluctuating water releases at Flaming Gorge Dam. Fluctuating flows to reduce rainbow trout recruitment downstream from GCD were ineffective at reducing year-class strength.

SMB create nests in gravel substrate, spawn at these sites, and remain at these nests while eggs hatch and emerge as larvae. Adjusting reservoir releases to disrupt SMB nesting activities in the Glen Canyon tailwater follows similar programs already implemented at Flaming Gorge Dam. Bestgen and Hill (2016) described how changes in releases from Flaming Gorge Reservoir could be used to alter SMB reproduction. This program builds on field observations from native and non-native SMB populations that highlight how small increases in near-bed velocity at nests can displace eggs and larvae, resulting in mortality. Bestgen and Hill (2016) examined SMB spawning in the Yampa and middle Green rivers using otolith analysis. They documented the timing of SMB hatching and related abiotic conditions on both rivers. Hatching generally occurred about two to three weeks after peak flows in June and July after water temperatures had reached 16C. This work led to a test of the utility of short duration high flow releases from Flaming Gorge Dam to disrupt SMB spawning in the middle Green River (downstream from the Yampa River) in 2021 and 2022 (Bestgen et al. FY 2022 report http://tinyurl.com/4yh2v5ym; presentation with 2021 results http://tinyurl.com/3pu98ppw). Preliminary results from the 2021 release suggested that SMB spawning was interrupted based on catch rates in different gears (page 55, Figure 10; http://tinyurl.com/4yh2v5ym). However, whether these short-duration high-flow releases led to reductions in SMB year-class strength overall (http://tinyurl.com/3pu98ppw and http://tinyurl.com/4yh2v5ym) is unknown. This latter condition would have to happen for flow modifications to be an effective SMB population control method.

Stakeholders could consider multiple approaches for short-duration high-flow releases from GCD. The Upper Basin approach would use high, steady flows to trigger SMB spawning in the Glen Canyon tailwater, followed by short-duration, even higher releases to disrupt spawning through nest abandonment caused by higher near-bed velocities. An alternative approach would modify weekly load-following releases to include both short-duration high flows and longer-duration somewhat lower flows to create the changes in velocity hypothesized to be necessary to interrupt recruitment via nest abandonment.

We consider any designer flow experiment to have highly uncertain outcomes because controlling SMB year-class strength effectively would need to cause year-class failure. SMB have demonstrated the ability to re-nest following nest disruption and demonstrate high recruitment compensation. These adaptations can promote successful recruitment even when spawning events during the reproductive season are interrupted (Pflieger 1975). In addition, key SMB and GSF spawning habitats in Glen Canyon, such as the slough at -12 mile, maintain relatively slow velocities until river discharge is very high, and these habitats are resistant to dewatering unless discharge is low.

Section 9: Cooling GCD water release temperatures to reduce SMB recruitment

Findings: We consider the effectiveness of short-duration cold water releases from GCD (cold shocks) to reduce SMB abundance via nest interruption to be uncertain. The cold shock may disrupt spawning but not lead to year-class failure, again due to re-nesting and compensatory survival. For the downstream areas (near and below Diamond Creek), natural downstream warming could reduce the effectiveness of this experimental action.

The cool-mix alternative assumes that maintaining water temperatures ≤ 16C (or 15.5C), which is the temperature at which SMB spawning behavior has been observed to be initiated in the Upper Basin, will prevent SMB from spawning (Eppehimer et al. in-review). This approach depends on temperature being the limiting factor for SMB spawning behavior, and the behavioral response of SMB in Glen Canyon and the CRe will be the same as in the Upper Basin. If these conditions are met, then cool-mix will likely reduce the risk of SMB spawning in Glen Canyon and elsewhere these thermal conditions are met. SMB spawning has been reported for a wide range of temperatures (12.8-21C within the USFWS SMB Risk Assessment, https://tinyurl.com/5n6v485f ; citing information from the USFWS SMB HSI, referencing Turner and MacCrimmon 1970). SMB could find isolated warmer water temperatures than the release temperature at GCD as nearshore warming has been documented at the slough in Glen Canyon. Nearshore warming was also documented during the 2000 low-summer steady flow experiment (https://tinyurl.com/mrya2hb2) but the nearshore warming varied between day and night and by location in the CRe. Natural warming could reduce the effectiveness of this experimental action for backwater habitats, tributary inflows, and downstream areas (near and below Diamond Creek).

A combination of cold-water releases and flow fluctuations would be an attempt to reduce SMB recruitment through multiple actions, including physical changes in water velocity and temperature, which could impact SMB eggs and larvae and induce behavioral changes in adult SMB related to cold water (i.e., nest building not initiated or nests abandonment). We are uncertain if combining these treatments is more likely to achieve the desired result of reducing SMB recruitment. Because SMB catch near GCD and Glen Canyon is low, accurately measuring a response to these experimental actions through monitoring will be difficult.

Eppehimer et al. (in-review) provide a detailed assessment of this flow option under the temperature-only hypothesis, including predictions on SMB entrainment, temperature of GCD releases under different reservoir levels, downstream river warming, and predicted SMB population growth at Lees Ferry, LCR, and Diamond Creek confluences.

An experimental approach would be to develop an adaptive management experiment using the selective water withdrawal device at Flaming Gorge to create cold water treatments and evaluate SMB spawning responses to these treatments. There are existing SMB monitoring programs below Flaming Gorge (Bestgen 2018; https://tinyurl.com/3pu98ppw) as part of research efforts to assess how spike-flows and other designer water releases impact SMB spawning patterns. These studies provide a baseline of SMB catch rates, the spatial distribution of spawning, and spawning chronologies, so there is potential to test actions proposed at GCD, such as the cold-spike or cool mix at a location in the Colorado Basin where SMB catch rates are higher. This type of experiment would create more of a basin-scale approach to learning the best strategies to managing the effects of specific temperature manipulations on fish populations while working to meet power and water obligations.

Section 10: Other management options that are discussed with long time horizons (> 5 years)

Reduce water release temperatures by maintaining a higher water surface elevation in Lake Powell, thereby lowering entrainment and the temperature of water withdrawn through the penstocks.

Findings: Maintaining Lake Powell at higher levels would also lower water release temperatures and reduce SMB entrainment.

Eppehimer et al. (in-review) also evaluate a strategy to manage reservoir release temperatures by maintaining Lake Powell at higher levels so the penstocks withdraw water from the colder hypolimnion. This action would also reduce the risk of entrainment of SMB through the penstocks because SMB typically live in the higher parts of the water column. Lake Powell provides half of the total storage of the Lake Mead-Lake Powell reservoir system that collectively guarantees sufficient storage during extended drought to maintain water-supply deliveries to the Lower Basin and Mexico. In the past, water storage has been divided approximately equally between Lake Mead and Lake Powell, and a plan to emphasize storage in Lake Powell would necessitate re-evaluating basin-wide reservoir operating policies. During an extended drought, such as between 2002 and 2004 and between 2020 and 2022, it might not be possible to maintain sufficient water storage in Lake Powell such that releases could always be < 16C. In this case, any SMB in the CRe would likely have suitable water temperatures to spawn. But, as discussed, we must determine if water temperature is the only limiting factor for SMB populations in the CRe.

Another approach to increase reservoir elevations in Lake Powell to reduce entrainment of SMB and temperatures below GCD is to move monthly water volumes within a year from the winter to the summer. Entrainment and warm water releases only occur in the summer and fall when the reservoir is stratified, and penstocks draw from the warmer epilimnion or metalimnion. By reducing monthly release volumes in the winter and early spring and instead releasing these volumes at the end of the water year, it may be possible to maintain higher reservoir levels from late spring through mid-summer when entrainment and reproduction of SMB occurs.

We recognize that there is no way to guarantee sustained long-term high-water levels in Lake Powell because the fate of Lake Powell is determined by watershed-scale runoff, ongoing effects of a warming climate, basin-wide negotiations about the magnitude and locations of reductions in consumptive water use, and management policy related to whether water storage should emphasize Lake Powell or Lake Mead. Many policy options are being considered in the ongoing development of the Post-2026 Interim Guidelines for Operations of Lake Powell and Lake Mead. These options could be considered part of interim operations under an adaptive management plan while the feasibility and costs of longer-term options are evaluated and compared within a structured-decision making exercise.

Entrainment barrier

Findings: If a zero-tolerance policy for SMB below GCD is adopted, then controlling SMB entrainment from Lake Powell would be required. Temporary solutions using engineered curtains may reduce entrainment and force cooler water to enter the penstocks. These short-term solutions may need to be used while longer-term solutions are developed.

Non-native fish in the forebay of Lake Powell currently pass GCD into the Glen Canyon tailwaters via the penstocks and turbines. Entrainment has increased as the reservoir level has declined. Small to medium-bodied fish will likely survive entrainment and passage through the turbines (Svoboda 2022). A 2022 Bureau of Reclamation report evaluated various methods to reduce SMB entrainment from Lake Powell (Svoboda 2022). These methods include:

- Measures targeting the forebay of the dam, such as net and metal barriers or non-physical barriers like air bubbles, CO_2, or electricity, to prevent fish from entering the penstock. These methods can be used alone or in combination.
- Withdrawing water from deeper in the reservoir would avoid taking in non-native fish generally found in the upper part of the water column. Implementing this strategy would necessitate long-term use of the river outlets.
- Using energy dissipating valves on reservoir penstock outlets to induce mortality of entrained fish. This strategy has been used at other dams.

- Using operational modification to increase mortality rates of fish passing through the turbines. The Frances Turbines currently used at GCD likely kill most, but not all, entrained fish. While operational modification may increase mortality, it is unlikely to be completely effective.
- Carbon dioxide, electricity, or UV light can be used in the penstock tube to kill entrained fish. The potential efficacy of this method at GCD is uncertain.
- Implementing measures to kill non-native fish that survive passage through the penstock. Such methods could include manual removal using nets, mechanical removal (discussed above), and physical and non-physical barriers to guide entrained fish to a removal location.

The efficacy and feasibility of these methods at GCD are uncertain because of the size and scale of the structures needed, as well as possible effects on power generation Svoboda (2022). Svoboda (2022) evaluated various methods to prevent SMB from escaping from Lake Powell based on the success of these strategies elsewhere and their potential efficacy at GCD. Svoboda (2022) concluded: "*There is no clear alternative that can fully eliminate non-native fish escapement from Glen Canyon Dam. Based on the review of applicable projects, a forebay barrier net or multi-stimulus barrier may be the most likely alternative to limit impacts to power production, operations, and recreation while maintaining a reasonable level of entrainment protection. Deeper water withdrawal through the river outlet works or installation of an energy dissipating valve on the penstock pipe will also likely limit passage of non-native fish, but considerable impacts to power production are expected.*"

Section 11: Using SDM approaches to evaluate management alternatives

There is a history within adaptive management programs of using structured-decision making approaches to inform management decisions when the management decisions have tradeoffs among resources or values among stakeholders. Runge et al. (2018) demonstrate this process for brown trout in the CRe and van Poorten and Beck (2021) demonstrate a structured-decision making process for invasive SMB in a small lake. As a first step, existing models developed for use in the CRe, including Runge et al. (2018) and Eppehimer et al. (in-review), could be modified to include interactions between HBC and SMB. Then the predictions from these models related to SMB and HBC response to each of the six management actions being considered to control SMB, could be combined with cost information on these actions. This could serve as the basis for a formal structured-decision making process to evaluate tradeoffs between each alternative. This exercise would likely identify knowledge gaps that could be addressed through new research efforts or revealed through continued monitoring over time as CRe conditions change.

A critical issue in developing an effective adaptive management program to inform SMB decision making in the CRe is that SMB catches are very low in Glen Canyon, this makes it very difficult to detect a response in the SMB from a management action. At the same time, the HBC populations in the CRe are higher than observed at any other point in the long-term monitoring program. The Upper Basin paradigm would suggest that SMB populations could expand rapidly, leading to changes in HBC populations, including increased extinction risk. However, until “something happens,” an adaptive management program will be challenging to implement. This difficulty in learning in an adaptive management framework exists because if SMB catches remain low, it is not clear if that is because of some management action at GCD. If SMB populations do not expand near Diamond Creek in the future is that because of a management action at GCD, or is it due to some factor limiting SMB expansion downstream of Glen Canyon? Additional monitoring in the far western Grand Canyon downstream of Pearce Ferry into the Lake Mead inflow where HBC juveniles are caught and there are existing warm-water non-native fish populations (including SMB; Albrecht et al. 2018) could provide some information at least on whether SMB and juvenile HBC overlap in areas with more frequent high turbidity. Overall, the low SMB catch below GCD and Glen Canyon will make assessing responses to management actions difficult. This difficulty arising from low SMB catch below GCD and the challenges this creates for understanding whether or not a management action was successful. This is one reason experiments in the Upper Basin, where SMB catch is higher, may improve understanding of how SMB may respond to management actions such as cold-water releases, that could inform management actions below GCD.

“Sweet spot” management of the CRe

Over a 20-year time horizon, surprises will continue to emerge from the CRe (Melis et al. 2015), which will create new challenges and learning opportunities, and likely highlight the simple reality that no single management approach can improve all river resources valued by stakeholders (Schmidt et al. 1998). One surprise relevant to the current SMB situation is the increase in water temperature in the CRe over the last 20 years and the transition from significant policy efforts focused on identifying ways to warm the mainstem CRe to promote HBC recovery through improved juvenile fish survival (USDOI 1994; Trammell et al. 2002; USDOI 2008), to now identifying ways to cool the CRe water temperatures as a way to minimize risk of impact from a warmwater non-native SMB who could impact HBC through predation on juveniles. Melis et al. (2015) describe a “sweet spot” in terms of modest downstream CRe water temperature increases to promote native fish recovery (including HBC) while not leading to large numbers of warmwater non-native species which could impact native fish. Entering the current period where post-2026 GCD operational planning is underway and calls to reduce consumptive demand are increasing (Wheeler et al. 2022), perhaps retrospective analyses of native and non-native fish population responses to management actions such as trout suppression flows, mechanical removal, and warming water coupled with more detailed assessments of the complex dynamics related to water temperature releases at GCD related to changes in reservoir elevation and episodic high runoff years within a “millennium drought” can better identify whether such sweet spots exist. If this sweet spot exists, this could be a target to manage a system as dynamic as the CRe, with many different values from individual stakeholders within the current or future governance structure.

Key long-term considerations

1. Over the last two decades of intensive fish work in the CRe, basin-scale hydrology has shown to have a much larger influence on fish populations than management actions for native or non-native species.
2. This does not mean that management actions should not be taken, but instead, we suggest management actions be evaluated within the directional scale of the Colorado River basin, where the combined effects of aridification and increased consumptive demands will only further tax the water availability in the basin.
3. This additive effect of changing climate and increased consumptive demands will increase the likelihood that legal and political concerns will drive water releases from GCD and decisions related to storage in Lake Powell.

-END-

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Appendix 1

Mechanical removal efforts to manage SMB management in the Upper Basin

The threat of SMB to Upper Basin native fishes has led to a large-scale mechanical removal effort by the Upper Colorado River Endangered Fish Recovery Program (UCRRP; http://tinyurl.com/39j9dvjc)[1] designed to reduce the abundance of SMB to promote the recovery of native species (removal efforts summarized in Appendix A; see Figure A-1 for removal locations). SMB was first introduced into the Upper Basin by the Utah Division of Wildlife Resources (UDWR) and Colorado Parks and Wildlife (CPW) as a sportfish and to control nonnative invasive fish species. We summarized the history of SMB stockings in the Upper Basin to document the timeline of SMB being released into the Upper Basin (Summarized in Appendix B;). A critical point in this timeline is the expansion of SMB populations in number and area within the riverine portion of the Upper Basin since the 1990s, possibly due to changes in reservoir releases, consumptive use of surface and groundwater, and climate-driven changes in precipitation and drought (Udall et al. 2022) which have led to more stable flows and seasonally modified thermal conditions.

Concurrent with these increases in SMB populations, there have been increasing concerns about how SMB and native fish interact. In the Upper Basin, declines in native fish populations, primarily the endangered Colorado pikeminnow, appear to be correlated with the increase in SMB abundance. Breton et al. (2014) provide a synthesis of other reports that identify significant declines in native fish densities in the Yampa River from 1998-2004, coincident with a large increase in SMB abundance from 2000-2004. Reasons for the SMB population increase include favorable spawning conditions during several years of low-flow conditions (warm water, little flow fluctuations) and an influx of SMB from Rifle Gap Reservoir during a reservoir release. How SMB populations may have influenced other native fish, including HBC is unknown. A summary of SMB and HBC status for each of the six HBC populations recognized in the 2018 Humpback Chub SSA can be found in Appendix B. One of the five HBC populations in the Upper Basin, Dinosaur National Monument, including the Yampa River, is considered extirpated (Valdez et al. 2021). While the proximal cause of the extirpation of this population is difficult to determine, both basin-scale processes and SMB and channel catfish predation have been

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[1] SMB is native to the St. Lawrence River and Great Lakes, Hudson Bay (Red River), and Mississippi River basins from southern Quebec to North Dakota and south to northern Alabama and eastern Oklahoma; Atlantic and Gulf slope drainages from Virginia to central Texas (Page and Burr 1991).

identified as contributing causes. An ad hoc team of scientists (Valdez et al 2021) concluded that:

> *"The cause(s) for the decline and functional extirpation of the humpback chub in the Yampa River is unclear. The last few specimens were observed at a time when river discharge was exceedingly low, with a concurrent proliferation of smallmouth bass (Haines and Modde 2007). In August of 2002, the lowest daily flow in 100 years occurred for the Yampa River at 1.8 cfs near Maybell; a flow of 2 cfs had previously been recorded in 1934. It is believed that the low flow of 2002 was the stressor that eliminated an already declining population of humpback chub in the Yampa River. Possible reasons for decline are the compounded effects of low flow and nonnative predaceous and competing fishes, including channel catfish and smallmouth bass, although other native fish species have persisted, including the roundtail chub (Smith 2019)."*

Flows in the CRe are a several orders of magnitude larger than in the Yampa River. Critically, the Upper Basin including the Yampa River and Green River below Flaming Gorge appear to have much lower turbidity levels than the CRe downstream of the LCR confluence. As a result, the expansion of SMB in the Yampa may not be an appropriate model for the CRe.

Appendix 2: History of SMB stockings in the Upper Basin

Green River Subbasin

- 1967: SMB were stocked in Flaming Gorge Reservoir on the upper Green River by UDWR as a sportfish and to control invasive Utah chub (Pettengill et al. 1983; Mullner and Hubert 1993; Teuscher and Luecke 1996). Utah chub is native to the Snake River and Bonneville Basin and first appeared in Flaming Gorge in 1964 (Holden 1991).
- 1977: SMB were stocked in Starvation Reservoir on the Strawberry River by UDWR as a sportfish and to control invasive Utah chub (Wolf et al. 2022). SMB continued to be stocked into Starvation Reservoir through 1984, where they have persisted without supplemental stocking.
- 1978: SMB were stocked by Colorado Division of Wildlife in Elkhead Reservoir, on Elkhead Creek, a tributary of the Yampa River (Hawkins et al. 2009).

Upper Colorado River Subbasin

- 1972: 6,325, 3-inch SMB were stocked in Rifle Gap Reservoir by Colorado Parks and Wildlife as a sportfish (CPW 2015); SMB were transferred illegally by anglers to Ridgeway Reservoir, Highline Lake, and other reservoirs.

- 1973: 1,800 SMB fingerlings were introduced into the Gunnison River near Delta by Colorado Division of Wildlife (Burdick 2008); none were reported in 1979-1981 surveys (Valdez et al. 1982).

- 1982: the UDWR stocked 500 SMB in Lake Powell as a sportfish that could use rocky shoreline habitats available at high reservoir elevations.

- ~1990: SMB were stocked by CPW in McPhee Reservoir on the Dolores River (CPW 2022).

- SMB had an unknown stocking source for the upper Colorado River between Rifle and Westwater Canyon where SMB were "incidental, rare captures" prior to 2003 but increased rapidly starting in 2003 (Burdick 2008).

- SMB were not reported present or stocked in Aspinall Units of Gunnison River (Blue Mesa Reservoir [CPW 2023], Morrow Point, Curecanti), Taylor Park Reservoir; or Shadow Mountain, Green Mountain.

Expansion of SMB

Green River Subbasin

- 1967-1972: SMB were not reported from any of 15 locations sampled in the Upper Basin (Holden 1973).

- 1979 to 1981: SMB were found in small numbers in the Green River subbasin, mostly in and near the mouth of the Duchesne River in the Uinta Basin where the channel meanders in a wide alluvial valley (Tyus et al. 1982). The location near the mouth of the Duchesne River indicates that the species had escaped from Starvation Reservoir and moved downstream into the Duchesne and Green Rivers.

- Prior to the initial stocking in Elkhead Reservoir in 1978, SMB were not detected in the Yampa River in 1951, 1967–1971 and 1976–1977 (Bailey and Alberti 1952; Holden and Stalnaker 1975; Carlson et al. 1979). Sampling in 1981–1982 produced one SMB, the first one detected in the Yampa River (Hawkins et al. 2009). Similar efforts from 1986 through 1988 did not detect any SMB (Wick et al. 1985; McAda et al. 1994).

- Relatively large numbers of SMB escaped into the Yampa River in 1992, when Elkhead Reservoir was drained for dam repairs and resident fish escaped downstream through the unscreened outlet structure. After 1992, SMB numbers increased in the Yampa River. In 1992 (only 49 were captured), but by 2003 were 18%, and by 2007, 51% of the adult fish captured within Little Yampa Canyon (McAda et al. 1994; Hawkins et al. 2009), located just downstream from Craig.

- SMB were translocated to Elkhead Reservoir as mandated by the state of Colorado as a condition of their removal from the Yampa River to provide fishing opportunities for recreational anglers (Roehm 2004). Elkhead Lake Management Plan established a maximum allowable escapement rate for translocated SMB of 10% (CDOW 2007).

Expansion in the upper Colorado River Subbasin

- 1979-1981: SMB were not reported from the upper Colorado River (Valdez et al. 1982).
  - Before 2003: SMB were "incidental, rare captures" but increased rapidly starting in 2003 (Burdick 2008).
  - Source of SMB in the upper Colorado River is considered uncertain but may have been escapement from Rifle Gap Reservoir.
- Numbers of SMB in the upper Colorado River subbasin have not been as high as in Green River subbasin.

Appendix 3 SMB background information

Pflieger (1975) reviewed SMB recruitment in native streams and rivers concurrent with a detailed study of native SMB in Courtois Creek, Missouri, during 18 years. Courtois Creek is a very small system, but these observations of native SMB ecology are informative to understanding how SMB may respond in larger river systems where they have been introduced. Pflieger (1975) described a complicated relationship between river discharge and stage and reproductive success. Flood events characterized by rapid changes in stage and discharge during the nesting season were detrimental to recruitment, mainly if those floods occurred when fry were emerging from their nests. Smaller increases in stage and discharge before nesting or after larvae had emerged from the nest may be beneficial to reproductive success because of increased production of food resources for SMB fry, dispersal of larvae reducing intraspecific competition, and inundation of habitat to minimize predation risk of fry. In this same review, Pflieger (1975) examined temperature effects on recruitment from field and lab studies, including the abandonment of SMB nets when water temperatures were ≤ 10C and swimming impairment of fry when temperature changed by 10C. From the detailed field study in Courtois Creek, short-term night-time low-temperature events associated with cold fronts ≤ 8C did not disrupt spawning, possibly because daytime high temperatures were about 12.8C. Pflieger (1975) identified the difficulty in separating temperature from river discharge effects on spawning, because these changes often coincided but he suggested that temperature may be less of a factor than water level fluctuations in disrupting SMB nesting in Courtois Creek, because low water temperature events were often of short duration during the April and May SMB spawning period. During the 18-year study, Pflieger (1975) studied SMB in Courtois Creek, no year-class failure was documented from scale-based year-class reconstructions despite the occurrence of different river discharge and temperature events during the spawning season that may have reduced egg or larvae survival. Pflieger (1975) attributed this potentially to the ability of SMB to re-nest following nest destruction by floods or other events. No relationship between adult abundance and recruitment was observed by Pflieger (1975).

Fish in the genus *Micropterus* (including smallmouth bass [SMB]) are considered one of the most invasive globally due to their high productivity, nest guarding, aggressive predation, and flexible habitat and trophic niche, among other reasons (Brown 2009; Costantini et al. 2023). Most of the work on the ecological effects of invasive SMB on native fish populations has been conducted in cold-water river and lakes. Several studies have documented SMB impacts to native fish, particularly small-bodied species, in lakes in Canada, New York State, and the upper midwest (reviewed in Brown et al. 2009 and Schiphouwer et al. 2017). In impounded stream reaches of the Pacific northwest, SMB have also been documented to prey on native pikeminnow, salmonids, and small-bodied fish like minnows (reviewed in Schiphouwer et al. 2017). SMB are also thought to have outcompeted native pikeminnow in the rivers in Washington (Carey et al. 2011) and California (Gard 2004).

SMB introductions into river systems typically occurred several decades ago and have only recently become a concern and a subsequent area of study. While declines in native fish are often attributed to invasive SMB, the SMB introductions are often concurrent with introductions other non-native species as well as significant changes in land use, pollution, and hydrologic modification that could also account for declines in native fish. In addition, the attribution of native fish decline to SMB is typically based on evidence of SMB predation and spatial correlations between the abundance of SMB and native fish rather than experimental manipulations that would more clearly demonstrate causation. For example, studies in regions as different as Spain, Zimbabwe, and South Korea have found native fish abundance was dramatically lower where *Micropterus* were present compared to sites where they were absent (Gratwicke 2001; Jang et al. 2006, Blanco-Garrido et al. 2009). While suggestive of *Micropterus* impacts, few studies can conclusively demonstrate that a decline in native fish was due to *Micropterus* (Wyle et al. 2014). **One exception is Kirk et al. (2022), who used a before-after, control-impact (BACI) design to study the effects of SMB expansion on native fish in the Laramie River, Wyoming. They found a decline in the relative abundance of small species of native minnows, but no overall decline in native species richness. However, the SMB expansion was relatively recent, so the full extent of SMB related community effects may not yet be realized.**

The effects of non-native fish introductions in the Gila River, NM was studied by Stefferud et al. (2011). They sampled fish in multiple habitats over 21 years under a variety of flow conditions. They modeled the responses of native cyprinids and catostomids to flow and the presence of predacious non-native fishes (primarily *Ameiurus natalis, Salmo trutta* or *Micropterus dolomieu*). They found that the abundance, richness, and recruitment of native fish was related principally to discharge. Still, small native species were uncommon in areas with higher densities of SMB, especially in low-flow years. Interestingly, the densities of large native species were positively associated with densities of non-native predators. These results suggest a complex size-specific interaction between native fish and SMB and that SMB can further reduce the abundance of native species under natural stressors, such as drought conditions.

A study in Fossil Creek, AZ, evaluated the relative distribution of SMB and native fish using surveys of stream habitats upstream and downstream of a fish barrier that excluded SMB (Jenny et al. 2023). They found that SMB appeared to reduce the local density of native species with niches similar to SMB. In contrast, species with the least niche overlap with SMB were the least affected. The authors also found that size classes of native fish most vulnerable to SMB predation shifted to more protective riffle habitats and avoided pools in reaches where SMB were present (Jenny et al. 2023). The authors attributed the habitat shifts to SMB competition with and predation on the native species. The results also suggest riffles and runs can serve as refugia for native fish (Jenny et al. 2023).

Since the mid-1980s, there has been a decline in native fish species in the Yampa River, Colorado, concurrently with a significant and rapid increase in SMB. In addition, surveys indicated SMB density is negatively associated with the presence and abundance of native fish in pool habitats in the Yampa River (Bestgen et al. 2007). Bestgen et al. (2007) and Hawkins et al. (2008), reported on a multi-year (2003-2007) experiment in which SMB were removed from the Yampa River and the relative abundance of native fish was compared between the reach with removal and a comparable reach in which no removal occurred. They found no significant difference in the relative abundance of native fish in removal areas compared to control reaches (Bestgen et al. 2007). Several potential explanations for the lack of an apparent effect from SMB removal included an insufficient number of SMB being removed, an increase in other non-native fish due to SMB removal that in turn suppressed native fish numbers, changes in flow and water temperature that were unfavorable to native fishes, and an insufficient number of adult native fish available to reproduce.

Outside of the US, invasive SMB has been intensively studied in the Cape Floristic Region (CFR) of South Africa (Weyl et al. 2014). In the Rondegat River, native fish were virtually absent in river reaches invaded by SMB but were present in upstream portions where a waterfall prevented SMB invasion (Weyl et al. 2014). Similarly, a study in the Witte River found that, when controlled for environmental conditions, native species were most abundant at sites where SMB was uncommon or absent (Shelton et al. 2014). Like the studies in the Gila River, primarily small (<150 mm adults) native fish were negatively associated with the presence of SMB (Shelton et al. 2015). This finding is likely the result of predation by SMB on small-bodied fish species. Similarly, Pilger et al. (2008) found the San Juan River was dominated by the non-native fish, native catostomids, and cyprinids were disproportionately represented in the diet of largemouth bass.

Unlike most of the observational studies reviewed so far, an experimental study was conducted in the 28-km Rondegat River, a shallow (<1 m deep) and relatively narrow (2–4 m wide) river with summer discharge between 0.07–0.08 m^3/s. The system has relatively few (24) native fish species. Non-native fish (primarily SMB) were removed from an invaded stretch of the river up to a waterfall that prevented upstream invasion by non-native fish (Wyl et al. 2014). Post-removal surveys indicated that the density of native fish rapidly increased following SMB removal, suggesting SMB contributed to the suppression of native fish populations. Woodford (2005) also attributed the reduction in native species in the Rondegat River to predation by SMB, and proposed that the lack of native predatory fish in the system resulted in native fish being behaviorally naïve and highly susceptible to non-native predators. Similar explanations have been used to explain native fish declines after the introduction of largemouth bass in arid rivers of Spain where there are also few native piscivores (Almeida and Grossman 2013).

Several conclusions can be made based on the existing literature regarding the effects of SMB on native fish species. These include:

- It may take decades for SMB impacts on native fish populations to be realized (Breton et al. 2014).
- Both SMB and native fish populations are strongly influenced by annual variations in seasonal flow, which can cause dramatic fluctuations in both populations such that relative abundance of native and non-native species can fluctuate over the long-term (Stefferud et al. 2011). This is particularly relevant in the Colorado River which may experience extended drought and where annual flows can be highly variable.
- Habitat is an essential influence on SMB interactions with native species. SMB may numerically dominate shallow pools, while riffles and deep pools may serve as refugia for native fish (Gibson et al.; Jenny et al. 2023).
- While large-bodied native fish may be vulnerable to SMB predation during early life stages, small-bodied native fish species (e.g., speckled dace) would be susceptible to predation over their entire lifespan and may therefore be most vulnerable to population declines following SMB invasion.
- The evolutionary history of the native population is essential. Systems without large numbers of native predators may be more vulnerable to impacts from SMB (Woodford 2005).

Appendix 4: Observed water temperature and turbidity (FNU) or suspended sediment (silt+clay) from different locations in the CRe 2008-2022.

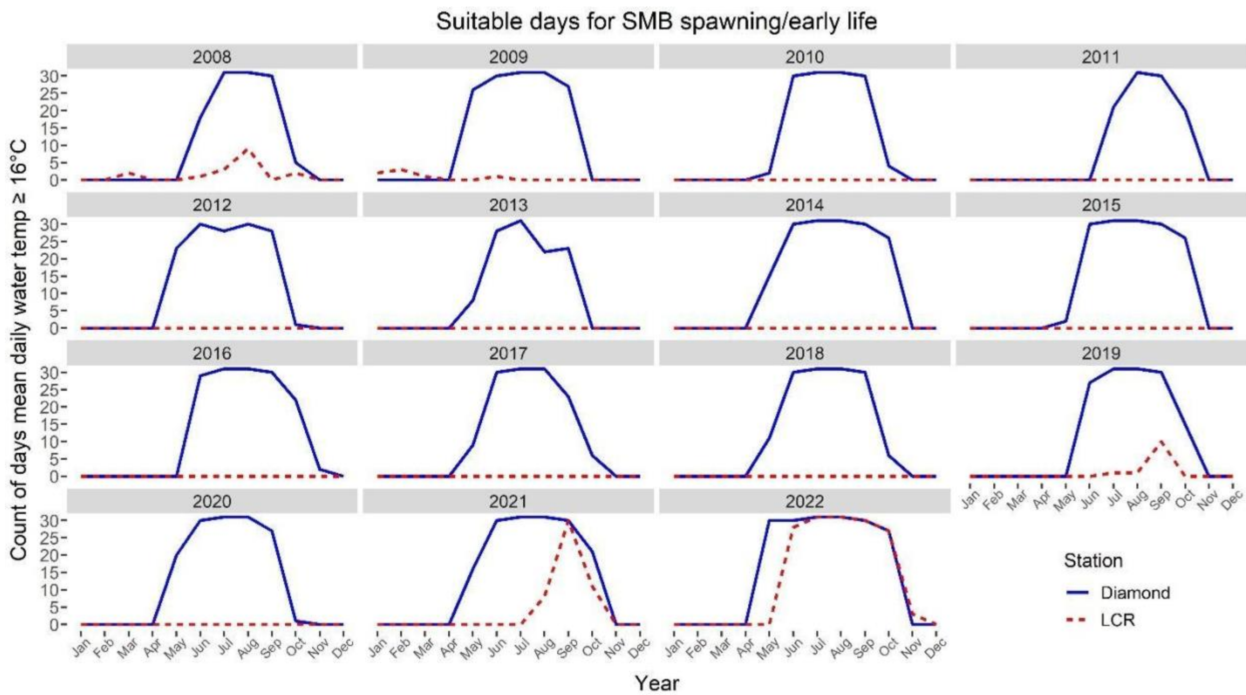

Figure A1. This figure shows the number of days each month (y-axis) the mean daily water temperature at either Diamond Creek (blue line) or the Little Colorado River (red line) is ≥ 16C. Each panel is an individual year, and then the x-axis is the months in that year. Data near the Little Colorado River are for USGS gage 09383100 (Colorado River above Little Colorado River near Desert View) and data in western Grand Canyon are for USGS gage 09404200 (Colorado River above Diamond Creek near Peach Springs). Data downloaded at https://www.gcmrc.gov/discharge_qw_sediment/stations/GCDAMP.

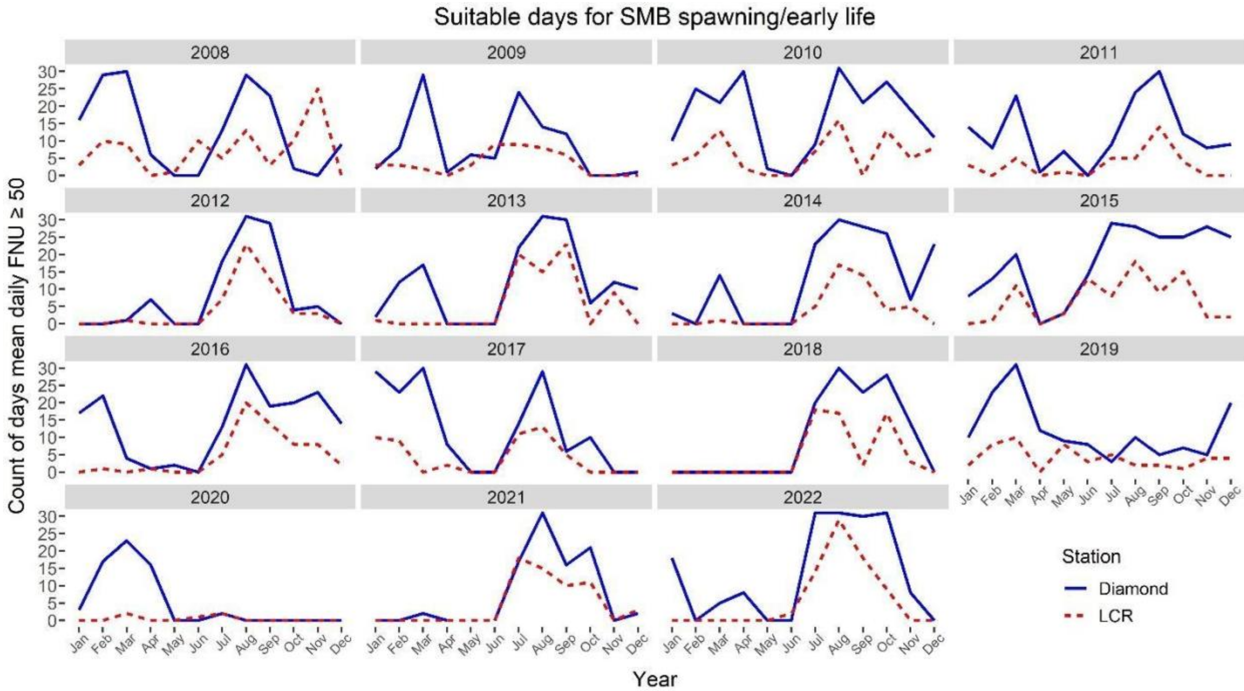

Figure A2. This figure shows the number of days each month (y-axis) the mean daily turbidity at either Diamond Creek (blue line) or the Little Colorado River (red line) is ≥ 50 FNU. Each panel is an individual year, and then the x-axis is the months in that year. Data near the Little Colorado River are for USGS gage 09383100 (Colorado River above Little Colorado River near Desert View) and data in western Grand Canyon are for USGS gage 09404200 (Colorado River above Diamond Creek near Peach Springs). Data downloaded at https://www.gcmrc.gov/discharge_qw_sediment/stations/GCDAMP.

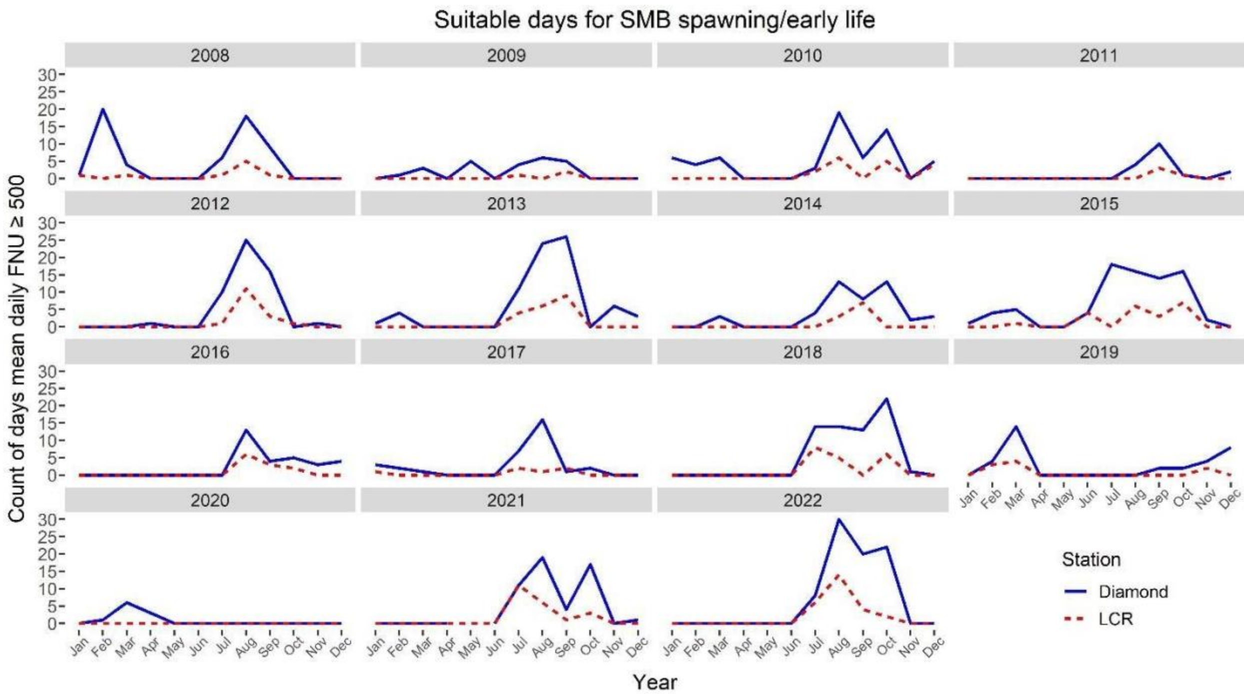

Figure A3. This figure shows the number of days each month (y-axis) the mean daily turbidity at either Diamond Creek (blue line) or the Little Colorado River (red line) is ≥ 500 FNU. Each panel is an individual year, and then the x-axis is the months in that year. Data near the Little Colorado River are for USGS gage 09383100 (Colorado River above Little Colorado River near Desert View) and data in western Grand Canyon are for USGS gage 09404200 (Colorado River above Diamond Creek near Peach Springs). Data downloaded at https://www.gcmrc.gov/discharge_qw_sediment/stations/GCDAMP.

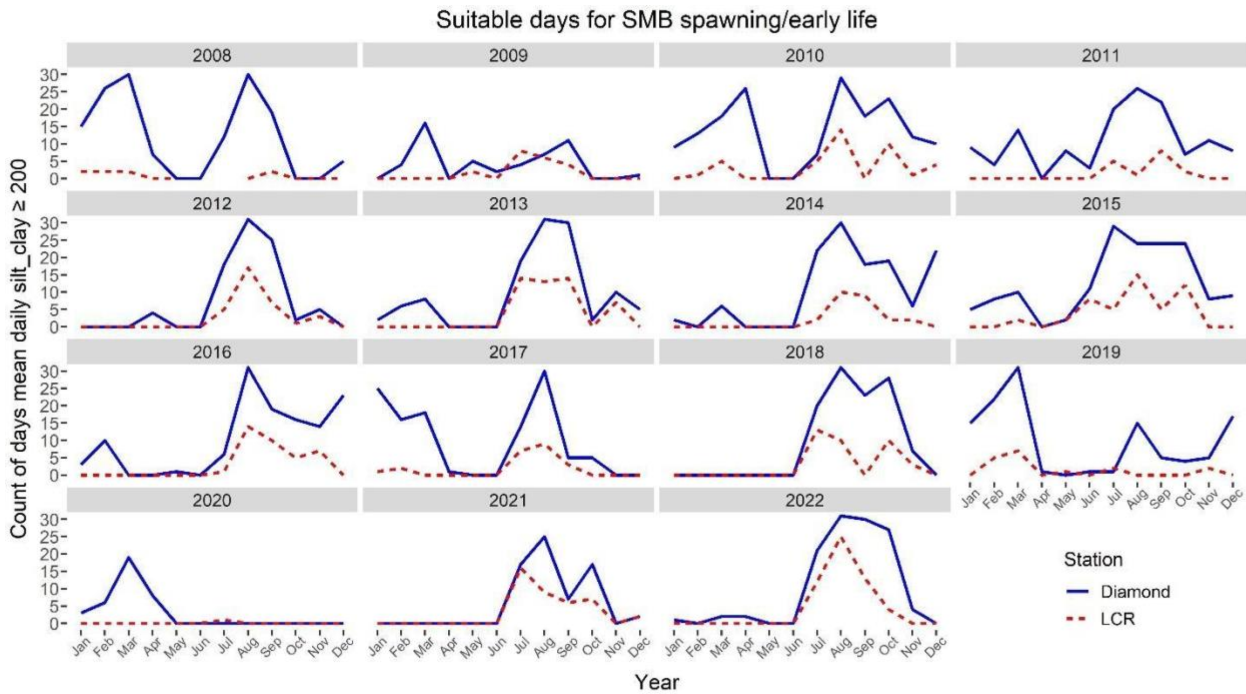

Figure A4. This figure shows the number of days each month (y-axis) the mean daily silt+clay concentration at either Diamond Creek (blue line) or the Little Colorado River (red line) is ≥ 200 mg/L. Each panel is an individual year, and then the x-axis is the months in that year. Data near the Little Colorado River are for USGS gage 09383100 (Colorado River above Little Colorado River near Desert View) and data in western Grand Canyon are for USGS gage 09404200 (Colorado River above Diamond Creek near Peach Springs). Data downloaded at https://www.gcmrc.gov/discharge_qw_sediment/stations/GCDAMP.

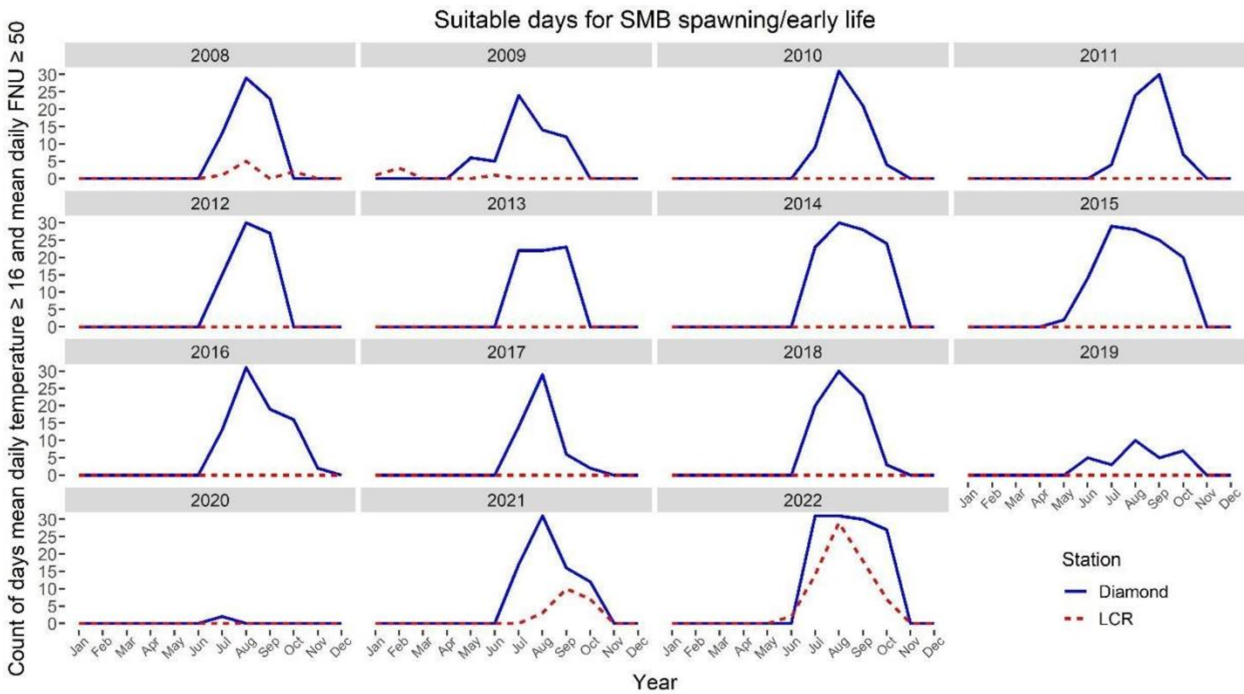

Figure A5. This figure shows the number of days each month (y-axis) the mean daily temperature is ≥ 16C and the mean daily turbidity is ≥ 50 FNU at either Diamond Creek (blue line) or the Little Colorado River (red line). Each panel is an individual year, and then the x-axis is the months in that year. Data near the Little Colorado River are for USGS gage 09383100 (Colorado River above Little Colorado River near Desert View) and data in western Grand Canyon are for USGS gage 09404200 (Colorado River above Diamond Creek near Peach Springs). Data downloaded at https://www.gcmrc.gov/discharge_qw_sediment/stations/GCDAMP. Under these conditions water temperatures would be warm enough for SMB to spawn, but feeding efficiency may be reduced during early life stages due to high turbidity. Slow SMB growth during summer and fall could reduce overwinter survival because of low energy reserves (lipids) headed into winter period.

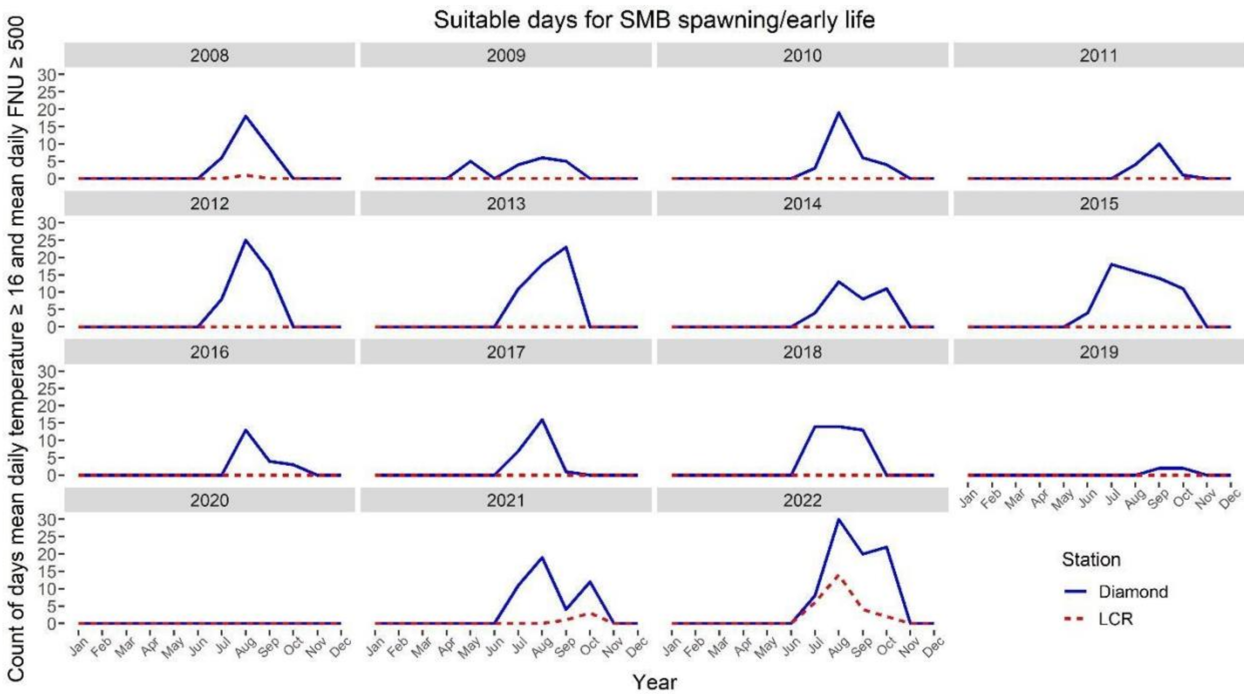

Figure A6. This figure shows the number of days each month (y-axis) the mean daily temperature is ≥ 16C and the mean daily turbidity is ≥ 500 FNU at either Diamond Creek (blue solid line) or the Little Colorado River (red dashed line). Each panel is an individual year, and then the x-axis is the months in that year. Data near the Little Colorado River are for USGS gage 09383100 (Colorado River above Little Colorado River near Desert View) and data in western Grand Canyon are for USGS gage 09404200 (Colorado River above Diamond Creek near Peach Springs). Data downloaded at https://www.gcmrc.gov/discharge_qw_sediment/stations/GCDAMP. Under these conditions water temperatures would be warm enough for SMB to spawn, but feeding for early life stages would likely be difficult due to high turbidity. Slow SMB growth during summer and fall could reduce overwinter survival because of low energy reserves (lipids) headed into winter period.

Appendix 5 Science Panel Charge from Clayton Palmer

draft: 09/15/23

SCIENCE PANEL

To Evaluate Management Options for Smallmouth Bass

In the Colorado River Through Grand Canyon

Membership:

John (Jack) Schmidt (Lead)

Mark Grippo, Argonne National Laboratory (Facilitator)

Richard Valdez

Bill Pine

Josh Korman

Science panel's charge:

The Science Panel shall identify, describe, and evaluate options for long-term[1] operational and management solutions for controlling smallmouth bass in the Colorado River from Glen Canyon Dam to Lake Mead.

The science panel is encouraged to consider:

- short-term and long-term options
- operational, infrastructure and other solutions
- context;
  - § the status and abundance (or lack thereof) of humpback chub and other native and nonnative fish in the Grand Canyon
  - § alternatives being considered in terms of the operations of Mead and Powell

§ projections of future runoff in the Colorado River basin

§ expectations for changes in consumptive use in the basin, and the impact of those changes on future water storage in Lake Powell and Lake Mead

o What is doable? What is feasible? Regarding the management of smallmouth bass in the Grand Canyon.

Science panel's interaction with other scientists and with other interested publics:

· Science panel meetings will NOT be public meetings

· Most of the interaction and information sharing among members of the science panel will be by email, phone calls and zoom meetings

· There will need to be some meetings of all members of the science panel – together (either virtual or face-to-face). Scientists at Grand Canyon Monitoring and Research Center (GCMRC), Bureau of Reclamation, National Park Service, U.S. Fish and Wildlife Service, and Arizona Game and Fish Department will be invited to attend these "all hands" science panel meetings

· One or more of the science panel meetings will allow people who represent other "interested publics".

Schedule:

While the science panel isn't limited in its thinking to the needs and purposes of the LTEMP EIS, the panel should provide information and recommendations in accord to the schedule of the LTEMP EIS. While Reclamation's draft schedule is likely to slip, the comment period of a public draft is set to end on February 12, 2024. This would have to be the latest date possible for a science panel to complete its work.

Guidance and direction from WAPA:

· WAPA will not give the science panel any policy direction or expect certain outcomes.

· WAPA will encourage the attendance of GCMRC scientists and others, but the members of the science panel can decide when they may need exclusive conversations. WAPA is encouraging of free exchange. We are NOT trying to pit scientists against each other.

Final Product:

· The science panel will produce a written report and a final briefing.

[1] i.e. 20 years.