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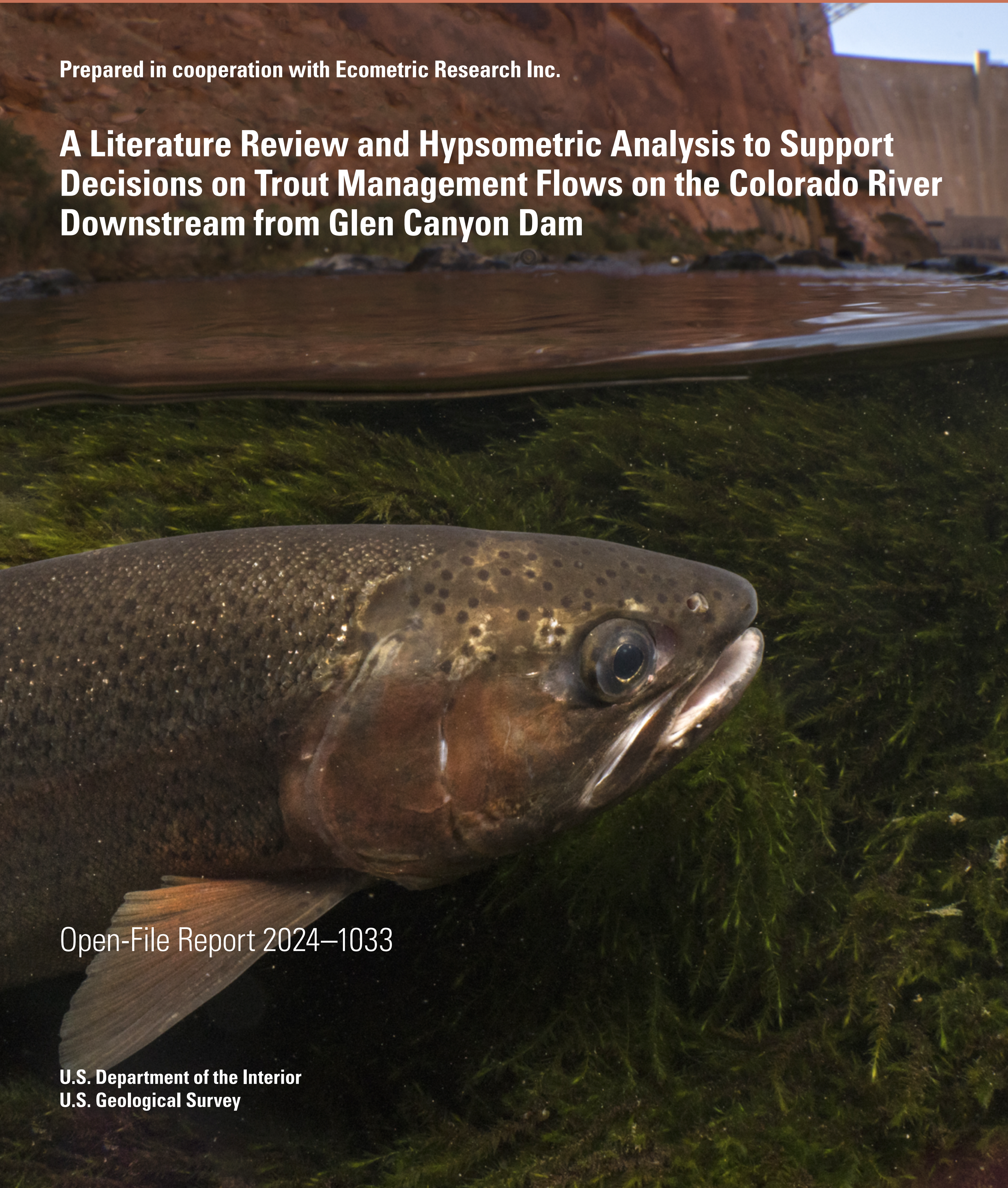

Prepared in cooperation with Ecometric Research Inc.

A Literature Review and Hypsometric Analysis to Support Decisions on Trout Management Flows on the Colorado River Downstream from Glen Canyon Dam

Open-File Report 2024–1033

U.S. Department of the Interior
U.S. Geological Survey

Cover: Rainbow Trout (*Oncorhynchus mykiss*) in Colorado River below Glen Canyon Dam, Arizona. Photograph by David Herasimtschuk, U.S. Geological Survey and Freshwaters Illustrated, August 2019.

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Conversion Factors

U.S. customary units to International System of Units

Multiply	By	To obtain
	Length	
inch (in.)	2.54	centimeter (cm)
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
square mile (mi^2)	259.0	hectare (ha)
square mile (mi^2)	2.590	square kilometer (km^2)
	Volume	
cubic foot (ft^3)	28.32	cubic decimeter (dm^3)
cubic foot (ft^3)	0.02832	cubic meter (m^3)
	Flow rate	
foot per second (ft/s)	0.3048	meter per second (m/s)
foot per minute (ft/min)	0.3048	meter per minute (m/min)
foot per hour (ft/h)	0.3048	meter per hour (m/h)
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)

International System of Units to U.S. customary units

Multiply	By	To obtain
	Length	
centimeter (cm)	0.3937	inch (in.)
millimeter (mm)	0.03937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
	Area	
square meter (m^2)	0.0002471	acre
hectare (ha)	2.471	acre
square kilometer (km^2)	0.3861	square mile (mi^2)
	Volume	
cubic meter (m^3)	264.2	gallon (gal)
cubic meter (m^3)	0.0002642	million gallons (Mgal)
cubic meter (m^3)	35.31	cubic foot (ft^3)
cubic meter (m^3)	1.308	cubic yard (yd^3)
	Flow rate	
meter per second (m/s)	3.281	foot per second (ft/s)
meter per minute (m/min)	3.281	foot per minute (ft/min)
meter per hour (m/h)	3.281	foot per hour (ft/h)
meter per day (m/d)	3.281	foot per day (ft/d)
meter per year (m/yr)	3.281	foot per year ft/yr)
cubic meter per second (m^3/s)	35.31	cubic foot per second (ft^3/s)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$°F = (1.8 \times °C) + 32.$

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:

$°C = (°F - 32) / 1.8.$

Executive Summary

Fish stranding has been studied in select rivers worldwide, often with the purpose of determining how to mitigate adverse effects of dam operations on highly valued salmon and trout populations. However, where a reduction in trout population size is desired by resource managers, as is the case downstream of the Glen Canyon Dam on the Colorado River, flow manipulations termed trout management flows (TMFs) may be used to optimize fish stranding and mortality. To inform the design and implementation of potential future TMFs, we reviewed relevant literature to identify key factors that influence fish stranding. We found that key factors were highly interdependent and site-specific, but general trends suggest that down-ramping (decreasing flow) at rapid rates in daytime during the late spring to summer emergence period would lead to stranding of age-0 rainbow trout in shallow shoreline habitat. A hypsometric analysis was then used to predict stranding risk for age-0 rainbow trout in Glen Canyon for a range of TMFs, which incorporated existing bathymetric data and flow and habitat suitability models. Our results indicate that a TMF with a steady high flow ranging from 12,000 to 16,000 cubic feet per second (ft^3/s) combined with a minimum flow ranging from 3,000 to 5,000 ft^3/s may effectively strand age-0 fish while also minimizing risk to water storage in Lake Powell and other resources. This strategy implemented under normal hydropeaking operations was predicted to lead to a substantive stranding risk when paired with low flows of 5,000 ft^3/s, and especially 3,000 ft^3/s. However, there remains uncertainty associated with elements of implementing an effective TMF downstream from Glen Canyon Dam. The main uncertainties include (1) the down-ramp rate that maximizes stranding of age-0 trout, (2) the duration of drawdown to maximize stranding mortality while minimizing impact to downstream resources, (3) duration of high flows required for age-0 fish to colonize newly created shoreline habitat (this is only for certain TMF hydrographs), (4) number of repetitions of TMF cycles to minimize compensatory survival response, and (5) recruitment threshold of both rainbow and brown trout populations to trigger TMF implementation.

Abbreviations

EIS	Environmental Impact Statement
FaSTMECH	Flow and Sediment Transport with Morphologic Evolution of Channels
HFE	high-flow experiments
HSI	habitat suitability index
LTEMP	Glen Canyon Dam Long-Term Experimental Management Plan
MLFF	Modified Low Fluctuating Flow
ROD	Record of Decision
SR	stranding risk
TMF	trout management flow
TRGD	Trout Recruitment and Growth Dynamics
USGS	U.S. Geological Survey

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By Mariah Giardina,[1] Josh Korman,[2] Michael D. Yard,[1] Scott Wright,[1] Matt Kaplinski,[1] and Glenn Bennett [1]

Introduction

In many waterways in the Western United States, particularly downstream from dams, rainbow trout (*Oncorhynchus mykiss*) and brown trout (*Salmo trutta*) have been introduced to promote sport fishing. The introduction of nonnative species in certain instances have resulted in the development of abundant and self-sustaining populations, which can have negative effects on native fish assemblages through competition and predation (Yard and others, 2016; Yackulic and others, 2018; Healy and others, 2020). Resource managers sometimes face the challenge of balancing mandates to promote sport fisheries and maintain or enhance endangered native fish populations (Dibble and others, 2015). With a lack of evidence-based outcomes for invasive species control measures, many management actions fall short in effectively mitigating the negative effect of invasive species on native fish populations (Mueller, 2005).

Downstream from the Glen Canyon Dam in Arizona, the Colorado River is a highly regulated river system. The Glen Canyon Dam Adaptive Management Program, tasked with improving the condition of this ecosystem and other resources (water, hydropower, recreation, cultural, and others), has a mandate to protect federally threatened species in the Grand Canyon region of the Colorado River while simultaneously maintaining a nonnative rainbow trout sport fishery in Glen Canyon. Humpback chub (*Gila cypha*) are endemic to the Colorado River and the quality of their habitat has declined because of the construction and operation of the Glen Canyon Dam. Declining humpback chub populations in the upper Colorado River Basin and in the Grand Canyon led to its initial listing as an Endangered Species under the Endangered Species Act of 1973 (16 U.S. Code 1531–1544). More recently, on November 17, 2021, the status of the humpback chub was downlisted from endangered to threatened by the (U.S. Fish and Wildlife Service, 2021).

The largest aggregations of humpback chub in the Colorado River basin are found downstream from Glen Canyon Dam, near the confluence with the Little Colorado River and in western Grand Canyon (Yackulic and others, 2014; Van Haverbeke and others, 2017). Higher population densities of rainbow trout in the mainstem of the Colorado River near the confluence of Little Colorado River are one of the factors that reduce growth and survival rates of humpback chub (Yackulic and others, 2018). The density of rainbow trout near the Little Colorado River depends on two factors: (1) the number of young-of-year trout that disperse from Glen Canyon into Marble Canyon and eventually to the reach of river near the Little Colorado River confluence (Korman and others, 2016) and (2) conditions in Marble Canyon that determine the persistence of rainbow trout in that reach (namely prey availability, water temperature, and turbidity; Korman and others, 2021). Korman and others (2016) showed that the density of trout at the Little Colorado River and Colorado River confluence between 2012 and 2016 was controlled by the abundance of trout in Marble Canyon and downstream dispersal rates. During this period, low immigration rates from an abundant population in upper and middle Marble Canyon produced sufficient trout numbers to the Little Colorado River confluence to increase the population density. The high density of rainbow trout in upper and middle Marble Canyon in 2012 was caused by the production of a large cohort of young-of-year rainbow trout in Glen Canyon in 2011, which resulted from high and steady equalization flows and higher nutrient concentrations that stimulated the food base (Avery and others, 2015; Korman and others, 2021). The young-of-year then dispersed into upper and middle Marble Canyon.

Rainbow trout population dynamics have been, in part, shaped by releases from Glen Canyon Dam that have changed over time in response to management priorities and associated policies. Once Glen Canyon Dam was completed and the powerplant came online in 1965, normal operations were limited only by interagency agreements and the maximum power generation capacity of 31,500 cubic feet per second (ft^3/s; U.S. Department of Interior, 1995). Flows were less constrained than current operations, with minimum levels of 1,000 ft^3/s between Labor Day and Easter and 3,000 ft^3/s between Easter and Labor Day. In February of 1992, interim flow criteria were implemented for Glen Canyon Dam operations, pending the completion of the 1995 Environmental Impact Statement (EIS; U.S. Department of Interior, 1995), these interim flow criteria were eventually codified with the 1996 Record of Decicision

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(ROD; U.S. Department of Interior, 1996), called Modified Low Fluctuating Flow (MLFF). These flow characteristics were limited to a daily minimum and maximum flow ranging between 5,000 ft^3/s and 25,000 ft^3/s, respectively. The minimum flow level could be no less than 5,000 ft^3/s from 19:00 to 07:00, and no less than 8,000 ft^3/s from 07:00 to 19:00. The range in daily flow discharge could not exceed 8,000 ft^3/s over a 24-hour period. Lastly, up-ramp (increasing discharge) and down-ramp, (decrease discharge) rates could not exceed 4,000 cubic feet per second per hour ($ft^3/s/h$) and 1,500 $ft^3/s/h$, respectively. Some exceptions to flow restraints were allowed for emergencies and experiments (for example, high-flow experiments [HFEs]). In 2016, after conducting an EIS to reevaluate the effects of dam operations on downstream resources, the Glen Canyon Dam Long-Term Experimental and Management Plan (LTEMP), along with the 1995 EIS and its associated ROD became the guiding documents for the subsequent 20 years. The LTEMP made very few changes to the previous MLFF regime but provides a slightly more even flow release throughout the year and allows greater down-ramp rates up to 2,500 $ft^3/s/h$. Additionally, the LTEMP includes a “condition-dependent adaptive design” for experimental trout management flows (TMFs) and mechanical removal of trout near the Little Colorado River confluence (U.S. Department of Interior, 2016b). Because the Glen Canyon Dam has experienced multiple periods of operational criteria, we have distilled major changes in operating regimes into a few periods: the time prior to the 1995 EIS will hereinafter be referred to as pre-EIS (1970–1991), and all operations after that time will be referred to as MLFF-LTEMP (1992–2020).

Prior to the operational changes in 1992, the Lees Ferry trout fishery required supplemental stocking because natural recruitment (production of young-of-year) was relatively low. Using a tetracycline tracer, Maddux and others (1987) determined that less than 30 percent of the trout population was produced naturally during a period of exceptionally high and stable flows (1983–87). Past researchers hypothesized that fluctuating flows reduced the spawning success and survival rates of early life stages thus limiting natural reproduction (McKinney and others, 1999, 2001; Maddux and others, 1987; Persons, 2002).

In 1990, rainbow trout abundance was approximately 100,000 (Angradi and others, 1992), and stocking was discontinued by 1998 (McKinney and others, 2001; Korman and others, 2012). Recent population estimates for rainbow trout are much higher and variable across years. Population estimates have varied from 1.2 million in 2012 to 200,000 in 2016 (Korman and others, 2017). Numerous studies have attributed the increase in rainbow trout abundance to an increase in survival of early life history stages caused by a reduction in flow variation (McKinney and others, 2001; Korman and Campana, 2009; Korman and others, 2012; 2017).

There are at least three potential management actions that could reduce high rainbow trout densities near the Little Colorado River confluence if conditions in Lees Ferry were to produce a large recruitment event followed by dispersement of young-of-year downstream. These management actions are (1) mechanical removal, (2) turbidity enhancement, and (3) TMFs. Of these actions, only mechanical removal has been evaluated using a large-scale field experiment (Coggins and others, 2011).

Mechanical removal of nonnative species was implemented from 2003 to 2006 in lower Marble Canyon (control reach) and the Little Colorado River inflow reach (removal reach). The abundance of rainbow trout in the removal reach near the Little Colorado River declined over the study period, however, the abundance in a nearby control reach located upstream from the removal reach also declined (Coggins and others, 2011). More recently (that is, 2012–16), studies that did not remove trout have observed similar system-wide declines in populations in Glen and Marble Canyons and near the Little Colorado River, and correlate this to reduced prey availability and elevated turbidity and water temperatures (Korman and others, 2016, 2017, 2021). These conditions also occurred during the 2003–2006 decline, thus reducing certainty about the effect of mechanical removal on trout populations near the Little Colorado River. When trout density in Marble Canyon is high, downstream dispersal of trout to the Little Colorado River reach is expected to overwhelm mechanical removal efforts (Bair and others, 2018). In addition, prescriptive measures using mechanical removal are costly and are not supported by local Native American Tribes. Mechanical removal is likely most effective during periods when trout immigration rates into the Little Colorado River reach are low, owing to lower population abundance combined with low abundance of humpback chub (Bair and others, 2018).

The goal of a turbidity enhancement strategy would be to increase turbidity levels in Marble Canyon to reduce growth rates of rainbow trout, either by adding fine-grained sediment into the system, or through management of fine-grained sediment that is naturally supplied by tributaries (Korman and others, 2021). This strategy was not considered as part of the LTEMP nor in the EIS analysis, though it was considered in early discussions of alternatives. Trout are visual sight feeders and low turbidity levels in late fall through early summer in Marble Canyon promote rainbow trout growth and successful reproduction. High turbidity can lead to substantial reductions in rainbow trout growth and conditions that can reduce spawning success and overall survival in Marble Canyon and downstream from the Little Colorado River (Korman and others, 2021). Consequently, flows that reduce turbidity may result in an increase in rainbow trout growth and reproduction in Marble Canyon, which can increase the number of rainbow trout near the Little Colorado River confluence. For example, autumn HFEs that are followed by high monthly flow volumes and hydropeaking during winter months may result in clearer water compared to an alternative without an HFE following sediment input from tributaries (Rubin and others, 2002). HFEs resuspend sediment from the channel bed, deposit it on higher elevation sandbars, and export suspended fine-grained sediment from the system (Rubin and others, 2002). This process in turn reduces background turbidity levels, and therefore could help maintain high trout densities in Marble Canyon and the Little Colorado River. In years when trout density in Marble

Canyon is high and there is sufficient sediment, leaving more fine-grained sediment on the bed could result in a longer period of elevated turbidity levels, which would help limit the abundance of rainbow trout near the Little Colorado River. Although the effects of turbidity on foraging and growth of rainbow trout are well established (Yard and others, 2011; Korman and others, 2021), there is still considerable uncertainty about the extent to which high winter flow volumes, winter hydropeaking, and autumn HFEs influence the turbidity regime in Marble Canyon, and how the effects of these processes vary with sediment supply.

Trout management flows are an experimental element identified in the LTEMP directed at limiting large recruitment events of rainbow trout (U.S. Department of Interior, 2016a, b, c). Owing to density-dependent factors, young rainbow trout produced in Glen Canyon disperse into Marble Canyon by midsummer (Korman and others, 2016). These young recruits can survive in Marble Canyon under suitable conditions, and slowly make their way down to the Little Colorado River (Korman and others, 2016, 2017; Yard and others, 2016).

Trout management flows are designed to reduce the survival of early life stages of rainbow trout in Glen Canyon to limit the extent of downstream dispersal into Marble Canyon. Because of the recent increase in brown trout in Glen Canyon (Runge and others, 2018; Yackulic and others, 2020), interest has been expressed in using TMFs to limit their abundance by substantially reducing or eliminating recruitment. However, since TMFs in the LTEMP were intendedfor rainbow trout, any application of this management strategy to brown trout would require regulatory changes to the LTEMP to account for the difference in time of year (earlier) required to target brown trout during their more vulnerable life history stages. Trout management flows designed for brown trout may also have to be altered if early life stage habitat use is centered more on deep, offshore habitats than shallow-edge habitats.

The purpose of this report is to address some of the critical uncertainties about rainbow trout and brown trout responses to TMFs. We provide: (1) a comprehensive literature review on fish stranding to identify the key factors affecting stranding, knowledge gaps, and possible research studies to reconcile uncertainties; (2) results of a hypsometric analysis that predicts stranding risk for age-0 trout in Glen Canyon under a range of TMFs using existing bathymetric data and flow and habitat suitability models; and lastly, (3) considerations on future research and monitoring for TMFs.

Literature Review

We conducted a literature review relevant to juvenile fish stranding in regulated rivers worldwide. Fish become stranded on the shoreline or in isolated pools when water levels drop quickly, often resulting in mortality. Fish stranding in regulated rivers can be caused by fluctuating flows due to hydropower operations but can also occur in unregulated rivers that have natural variation in flows (Thompson, 1970; Bauersfeld, 1978; Pflug and others, 1989; Hunter, 1992; Irvine and others, 2009; Nagrodski and others, 2012). Most of the fish stranding literature is focused on mitigating stranding-related mortality resulting from hydropeaking flows on high-valued salmon and trout species. Only one study identified stranding as a strategy to control invasive fish populations, and it was conducted in a reservoir to promote the growth of smallmouth bass (*Micropterus dolomieu*) for recreational fishing by reducing the population of invasive bluegill (*Lepomis macrochirus*) (LeRoy Heman and others, 1969).

Consequently, the TMFs proposed in the LTEMP are a new management concept that has not been implemented or tested elsewhere. Reviewing the existing literature on fish stranding can help inform key elements of potential TMF hydrographs.

Methods

Data used in this study are provided in Korman and others (2024).

Literature Search

The literature review was based on results returned from two major search engines: Web of Science and Google Scholar. Multiple combinations of keywords were used in the search: strand*, entrap*, entrain*, fish kill, fish salvage, fish, juvenile fish, age 0 fish, salmonid, trout, hydropeak*, fish response to altered flow, flow regime, mortality, population effect, regulated river, hydropower, fluctuat*, dam effects, and dewater*. For searches in Google Scholar that generated more than 30,000 results, sources were sorted by "most relevant," and the first 1,000 sources were evaluated for selection, or until there were multiple pages with no relevant results. In Web of Science, all search results were evaluated for selection. Using various combinations of keywords in both search engines resulted in over 40,000 sources. Following this, sources were selected based on two criteria: (1) included salmonid species (note: the few stranding papers focused on cyprinids were included to understand potential TMF impacts on humpback chub, but cyprinids were not specifically searched for in keywords as this was not primary focus of this paper), (2) the intent of the paper was to study stranding (whether observed, estimated, or modelled). These criteria eliminated papers on stranding of marine species and tropical fishes, as well as vessel-induced stranding. Similarly, other sources where fish stranding was due to anthropogenic structures that led to entrainment or entrapment were also excluded. A small percentage of relevant papers were reviews of the primary literature on stranding and associated variables; these secondary sources were excluded from the review to eliminate redundancy.

After applying the above selection criteria, 69 relevant papers were entered into a database (appendix 1). Each paper was evaluated for whether it addressed major factors influencing stranding, which include: (1) flow characteristics (flow range, duration of high and low flows, down-ramp rate, magnitude of stage change, frequency of flow fluctuations), (2) fish life stage and size class, (3) channel morphology (substrate, lateral slope, and structural features), (4) diel cycle (night versus day), and (5) water temperature. The selected studies used varying metrics for the above factors or, in some cases, did not include specific factors. Therefore, we classified each study on whether it included each factor based on the following criteria: a factor was not included in study, a factor was identified in the study but not directly evaluated to quantify its effect, or a factor was evaluated (for example, a comparative analysis using a range of values) for effect on stranding.

Additionally, we searched the selected studies for the effect of stranding mortality events on long-term population trends to supplement our understanding of using stranding as a population-control mechanism.

Analysis of Flow, Light, and Stage for Glen Canyon

Since we evaluated flow characteristics and water temperature as factors influencing stranding in the literature review, we analyzed data on the physical characteristics of the Colorado River in Glen Canyon to provide site-specific context to interpret results from the literature review. Discharge and temperature data for the Colorado River were acquired from U.S. Geological Survey monitoring station at Lees Ferry (U.S. Geological Survey, 2022). Our analysis of the Glen Canyon Dam flow data is limited to only the winter, spring, and early summer (that is, January–July), is when stranding of young-of-year rainbow and brown trout would likely occur. To characterize normal hydropeaking operations at Glen Canyon Dam, we separated daily discharges into two operational periods: pre-EIS and MLFF-LTEMP. We defined an early period between 1970 and 1991 as pre-EIS but excluded flow records that were considered anomalous because of operational changes from filling regimes (March 15–June 30, 1977), tests (June 1–30, 1980), or flood risk (1983–87). We defined a later Modified Low Fluctuating Flow (MLFF) period between 1992 and 2016, but we excluded experimental flows (March 23–April 7, 1996; March 26–June 30, 2000; March 5, 2008), high flows (1997–98), and reservoir equalization operations (2011). Since the current operations under the LTEMP have not significantly changed from the MLFF period (except for monthly water distribution and the down-ramp rate, which we address where necessary), we grouped these operations between 1992 and 2022 as MLFF-LTEMP for the purposes of this paper. We calculated several flow characteristics for the two periods, including (1) the average monthly minimum flow, (2) the average monthly down-ramp rate based on highest hourly rate encountered per day, (3) the mean monthly standard deviation of daily flow minimums and mean monthly standard deviation of daily down-ramp rates, and (4) the daily flow range (maximum minus minimum).

To compare the effects of flow variation reported in the literature with those in the Colorado River, we calculated the stage change in Glen Canyon using stage-discharge relationships (Randle and Pemberton, 1987). Exposure of the river bottom to direct sunlight following decreases in stage height will expose stranded fish to warmer water temperatures, which will have a negative impact on their survival. We therefore calculated the average time of day when direct sunlight strikes the mid-channel of Glen Canyon using a solar model that accounts for topographic obstruction and solar declination (Yard and others, 2005). Estimates of solar exposure are based on mid-river points at 100-meter intervals from Lees Ferry upstream to Glen Canyon Dam. For comparison, we calculated solar exposure on two different dates that would reflect conditions during TMFs targeting brown trout (March 21) and rainbow trout (June 1). These dates reflect a period of peak emergence based on average water temperatures.

Findings

General Overview of Stranding Literature

Of the 69 stranding papers analyzed, 59 percent were peer-reviewed journal articles, 33 percent were technical reports (9 of the 23 technical reports were conducted on the Hanford Reach of the Columbia River), 4 percent were conference papers, and 3 percent were graduate or doctoral theses. More than 73 percent of the studies were published in the last 20 years. Rainbow trout and (or) brown trout were included in 52 percent (36) of the 69 stranding studies. Table 1 provides an overview of the extent that the stranding literature quantifies factors influencing stranding. Structural complexity (37 studies) and down-ramp rate (26 studies) were the factors most evaluated for effect on stranding. Although life stage was the most common factor identified in the 69 stranding papers, it was only quantified for its effect on stranding outcome in 17 studies.

In reviewing the existing stranding literature, we found that the major factors contributing to stranding are highly interdependent and site-specific. For example, the effects of ramping rates will depend on channel morphology. Effects of flow regime on stranding can vary substantially between winter and summer, and between night and day.

Flow Factors Influencing Fish Stranding

Most of the reviewed literature evaluated fluctuating flow characteristics and the effects on fish stranding. In total, 86 percent of the studies were conducted on regulated rivers. In these studies, dams were primarily operated for hydroelectric power generation, which resulted in considerable variation of flows within a day. In studies of smaller volume rivers, some dams served as water storage facilities and were subject to more constant releases within a day, and instead flows varied only by season to meet downstream water needs. As such, the metrics used to quantify each flow-related factor varied across studies, making comparisons difficult. Some of the metrics included the flow range (minimum and maximum), duration of time under high and low flows (that is, shoreline habitat inundation or “wetted history”), down-ramp rate, stage change, and frequency of flow fluctuations, as defined below. We used data from Glen Canyon Dam to put literature review results into context (U.S. Geological Survey, 2022).

Table 1. Summary of key factors evaluated in chosen studies from literature review.

[The first column lists the factors assessed. The second column shows the number of studies that did not address a given factor. The third column shows the number of studies that identified a factor but did not evaluate it for effect on stranding outcome. The last column shows the number of studies that evaluated a given factor for its effect on stranding outcome.]

Key Factors	Not addressed	Identified, not evaluated	Evaluated for effect
Flow characteristics			
Magnitude of stage change	27	30	12
Duration of flows	32	26	11
Down-ramp rate	23	20	26
Frequency of fluctuation	23	37	9
Channel morphology			
Slope	47	9	13
Substrate	17	37	15
Structural complexity	16	16	37
Other factors			
Life stage and size class	4	48	17
Diel cycle	33	18	18
Temperature	34	27	8

Flow Range

Flow range is the difference between maximum and minimum flows over a specific time interval. Higher flow ranges can have a large effect on shoreline habitat, which then affects aquatic biota, however, the effect of flow range is highly dependent on channel morphology. For example, Bauersfeld (1978) found that the effect on stranding was less for a flow decrease in the higher range of discharge levels from 9,000 to 5,000 ft^3/s compared to the lower range of discharge levels from 5,000 to 2,300 ft^3/s. Similarly, Dauwalter and others (2013) found that age-0 rainbow trout were stranded less at a discharge range from 1,800 to 900 ft^3/s compared to a discharge range from 600 to 300 ft^3/s. Stranding was greatest at the lower flows regardless of ramping rates because of greater stage changes at lower flows and increased presence of low-angle gravel bars (Phinney, 1974).

Originally, discharge levels for Glen Canyon Dam were highly variable (1970–91), with mean daily minimum and maximum discharges of 5,022 ft^3/s and 17,261 ft^3/s, respectively. Whereas, for MLFF-LTEMP operations (1992–2020), the respective mean daily minimum and maximum discharges were 8,330 ft^3/s and 14,648 ft^3/s. Minimum daily discharges were highly variable within and across months during the pre-EIS period, primarily due to differences in mean flows between weekends and weekdays (fig. 1). There was less predictability in the daily minimum flow within and across months during the

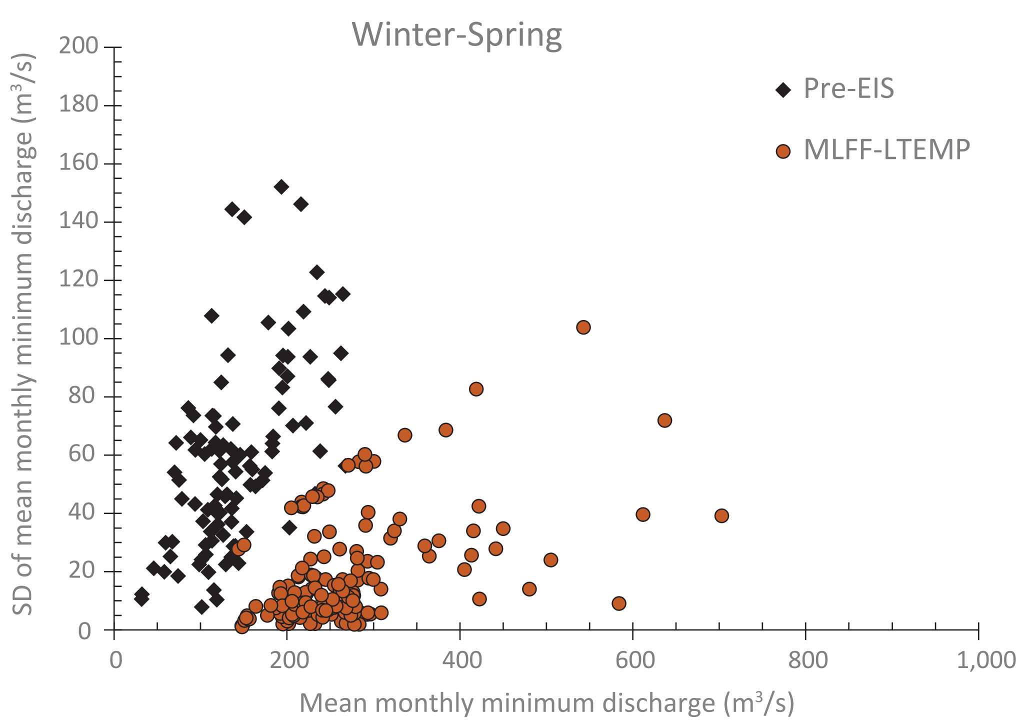

Figure 1. Plot showing relationship between the standard deviation (SD) in the daily minimum discharges for each month and the mean of daily minimum discharges by month from Glen Canyon Dam, Arizona. Black and red points identify data from the pre-Environmental Impact Statement (Pre-EIS;1970–91) and Modified Low Flucatuating Flow Long-Term Environmental Management Plan (MLFF-LTEMP; 1992–2020) operating periods. Data are from streamgage 09380000 at Lees Ferry (U.S. Geological Survey, 2022) for the period of January 1 to June 31 each year. m^3/s, cubic meter per second.

dam's pre-EIS period, which may have been an important factor that limited the extent of natural recruitment (see "Frequency of Fluctuating Flows" section).

Similarly, the difference in the daily flow range between the maximum and minimum discharge levels from Glen Canyon Dam was much greater in the pre-EIS period (12,240 ft^3/s) than during the MLFF-LTEMP period (6,320 ft^3/s) (fig. 2). The daily flow range under pre-EIS operations required higher down-ramp and up-ramp rates. Greater differences in daily discharge ranges would have dewatered larger areas compared to current operations.

Stage Change

River stage measures the vertical elevation of the water surface. As such, the extent of stranding can increase with the magnitude and rate of stage change. For a given flow range, the extent of stage change will be greater in narrow channels compared to wider ones. Only a few of the stranding studies (15 percent) have documented change in stage. Bauersfeld (1978) identified stage change as a more important factor than down-ramp rate on stranding. Long-term monitoring studies on the Columbia River identified stage change as having an important effect on stranding rates (Anglin and others, 2006). Irvine and others (2015) modelled the probability of large stranding events based on magnitude of stage changes on the Columbia and Kootenay Rivers and found that stranding risk increases with greater changes in stage heights .

In Glen Canyon, the relative change in stage is greatest at lowest discharge levels owing to the shape of the cross section of the channel. On average, stage decreases from 6.5 inches per 1,000 ft^3/s decrease in flow at discharge rates ranging between 5,000 and 10,000 ft^3/s, to 4.4 inches per 1,000 ft^3/s decrease in flow at discharge rates between 10,000 and 15,000 ft^3/s, and to only 3.8 inches per 1,000 ft^3/s decrease in flow at discharge rates ranging between 15,000 and 20,000 ft^3/s (Randle and Pemberton, 1987). Halleraker and others (2003) found that fish stranding increased substantially at stage changes greater than 23.6 inches per hour (in./h). For Lees Ferry, a 23.6 in./h stage change would require a down-ramp rate equivalent to 3,630 cubic feet per hour (ft^3/h) for discharges ranging between 5,000 and 10,000 ft^3/s. These down-ramp rates would have to be increased for higher discharges.

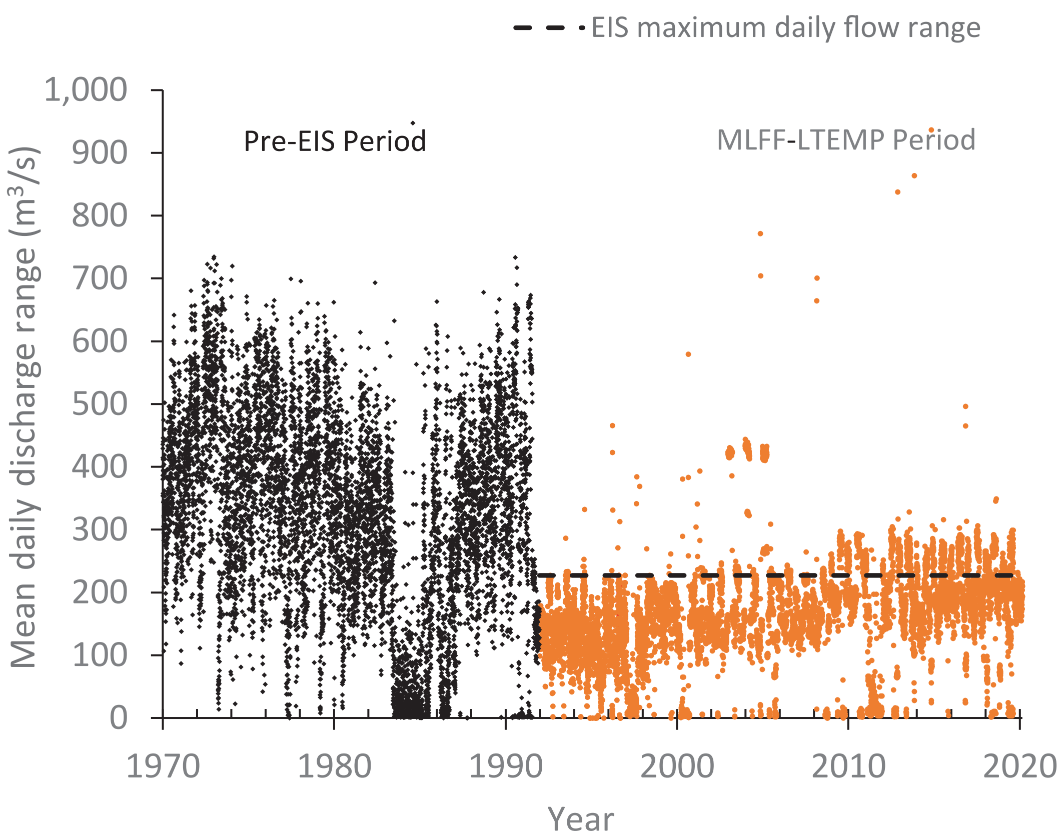

Figure 2. Plot showing mean daily discharge range between 1970 and 2020. The daily range is calculated as the difference between the daily maximum and minimum discharge value, in cubic meters per second (m^3/s). Data are from streamgage 09380000 at Lees Ferry (U.S. Geological Survey, 2022). Black and orange points identify data from pre-Environmental Impact Statement (Pre-EIS;1970–91) and Modified Low Flucatuating Flow Long-Term Environmenbtal Management Plan (MLFF-LTEMP; 1992–2020) operating periods. EIS flow range shows the maximum allowable range of flows with a day (0 to height of dashed line) for the MLFF-LTEMP period.

Duration of Drawdown

The duration of the low-flow periods during hydropeaking is an important factor for determining survival of eggs, alevins (early fry stage), and juvenile fish in dewatered habitats. Experiments conducted by Becker and others (1986) found that survival rates of eggs that were dewatered for 4 to 12 days was considerably higher than for alevins, where 96 percent died when exposed for less than 1 hour. In Glen Canyon, Korman and others (2011a) showed that redds located at higher elevations, which were exposed to air for longer durations, were more likely to contain dead eggs compared to redds located at lower elevations where exposure periods were shorter. Other factors—such as inter-gravel flow, air temperature, water temperature, habitat type, and time of year—can also affect survival rates following exposure to air. If the duration of dewatering is short and air temperature is cool (as would be the case during nighttime drawdown and up-ramp), eggs, alevins, and even juvenile fish that become stranded may survive until water levels return (Casas-Mulet and others, 2015, 2016). Lastly, owing to compensatory effects, decreased egg survival in dewatered redds had no population-level effect on recruitment levels (see fig. 3 of Korman and others, 2011a), as fish that survived experienced less competition from other fish.

Mortality rates of fish that are dewatered will increase with air temperature (Gingerich and others, 2007). Unfortunately, limited information exists on specific dewatering mortality rates for cold-water species like trout (Becker and others, 1986). In salmonids, smaller rainbow trout are more resistant to hypoxia (suffocation) and less resistant to heat gain and desiccation (Shi and others, 2018). Because of intolerance to higher air temperature with decreasing body mass, it is likely that smaller fish will have lower survival compared to larger fish when exposed to warmer air temperatures that occur under direct sunlight (which occurs by midmorning in most locations in Glen Canyon). In contrast, we would expect smaller fish to survive at a higher rate compared to larger fish when stranded at night due to cooler air temperatures and moist conditions in the interstitial spaces of rocks and in small pools on exposed shorelines.

Owing to the high canyon walls and channel orientation, most of Glen Canyon does not receive direct sunlight until middle to late morning (Yard and others, 2005). Though air temperatures become elevated by midday at Lees Ferry, the early to midmorning air temperatures at ground level are likely cooler along the shoreline due to evaporative effects from the recent

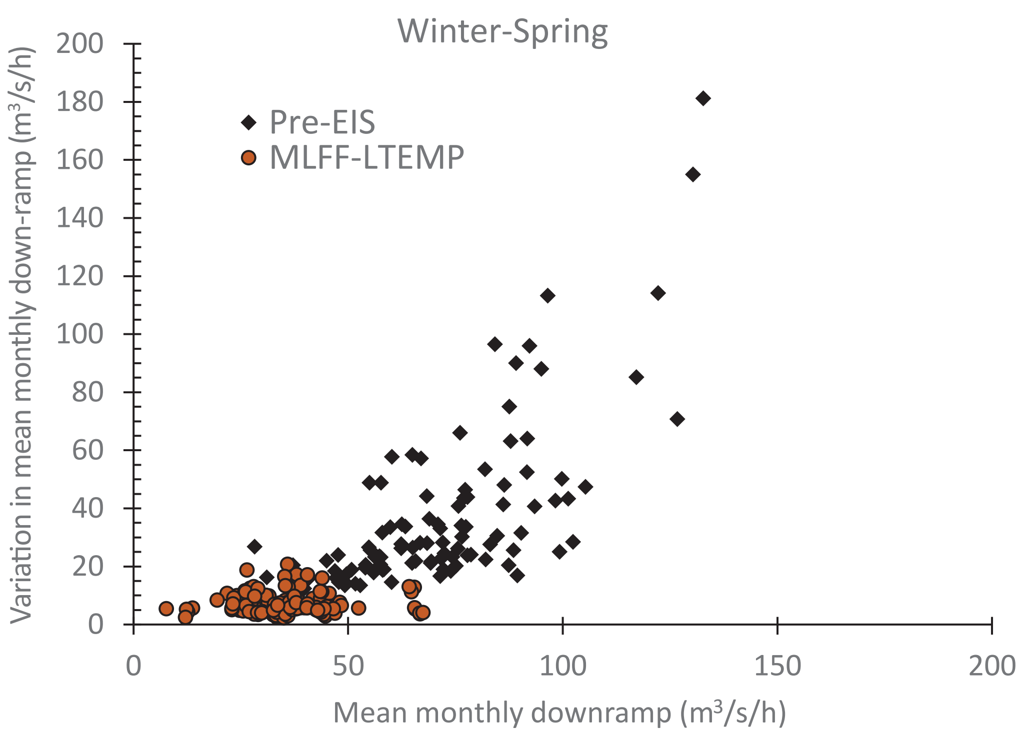

Figure 3. Plot showing relationship between the standard deviation (SD) in the daily down-ramp rate for each month and the mean of daily down-ramp rates for each month during winter and spring for Glen Canyon Dam, Arizona, in cubic meters per second per hour ($m^3/s/h$). Black and red points identify data from the pre-Environmental Impact Statement (Pre-EIS; 1970–91) and Modified Low Flucatuating Flow Long-Term Environmental Management Plan (MLFF-LTEMP; 1992–2020) operating periods. Variability is the standard deviation of the mean monthly down-ramp rate.

drawdown. Particularly in dewatered shoreline with vegetation (aquatic and terrestrial), cooler temperatures may be retained longer due to moisture retention from vegetation. These thermal and humidity differences, however, are currently unknown. Therefore, stranding mortality will be more effective if the flow drawdown occurs during warmer periods. As of 2021, the average up-ramp time begins at 06:45 at the Lees Ferry gage station, therefore, the upper 25 km of Glen Canyon is almost never exposed to the lowest stage discharge elevation during daylight hours [see “Diel Cycle (Day versus Night)” section].

Duration of High and Stable Flow Conditions

The “wetted history,” or how long shoreline habitat has been inundated before down-ramping, has been shown to affect the stranding rate of juvenile fish (Armstrong and others, 1998; Freeman and others, 2001; Irvine and others, 2009; Cocherell and others, 2012; Rolls and others, 2012; Irvine and others, 2015). Higher rates of fish stranding were found in the Columbia River following long periods of high and steady flows that allowed for fish to colonize nearshore habitat and become stranded when flows dropped (Irvine and others, 2015). However, Hvidsten (1985) found no significant relationship between duration of time a habitat was inundated and the extent of stranding once the water levels were dropped. Avery and others (2015) found that young-of-year rainbow trout colonized high elevation habitat that was wetted during high and steady equalization flows in the spring and summer of 2011, indicating that fish will colonize newly wetted habitat if given enough time. However, the 2011 equalization flows spanned a period much longer (April–September) than any TMF option and the specific duration of higher flows required to encourage colonization remains unknown.

In Lees Ferry, Korman and Campana (2009) found that, under a predictable 24-hour fluctuating flow regime, most small rainbow trout fry do not follow the wetted edge as it rises and, instead, remain near the daily minimum flow elevation. This suggests that a longer period of higher flows is required for fish to move laterally to higher channel elevations. In a stranding experiment using an artificial channel, Halleraker and others (2003), found that the proportion of fish stranded over a sequence of dewatering tests declined over successive flow drops suggesting fish remained in deeper water levels after longer exposure to repeated flow fluctuations.

TMF hydrographs described in LTEMP include a high-flow period to entice rainbow trout fry to move into shallow low-angle habitats that would then be dewatered when flows are rapidly reduced (although other TMF hydrographs are possible, see hypsometric analysis). The colonization time is likely in the order of weeks and remains a major uncertainty with respect to designing TMFs. However, under equalization flows, which caused the large recruitment event for rainbow trout in Glen Canyon in 2011 (Avery and others, 2015), flows were high and steady for a period of months (April–September) when vulnerable fry were present.

Down-ramp Rate

The rate at which discharge decreases over time (ft^3/s/h) has been identified as an important factor influencing stranding in regulated rivers. Stranding rates of juvenile brown trout were reduced by 50 percent when the down-ramp rate was reduced from 635.7 to 21.2 ft^3/s/h (Halleraker and others, 2003). Similarly, on the Nidelva River in Norway, a down-ramp rate reduction from 1,907 to 459.1 ft^3/s/h nearly eliminated stranding of juvenile Atlantic salmon (*Salmo salar*) (Saltveit and others, 2001). On the Sultan River, Olson and Metzgar (1987) concluded that down-ramp rates from 88.3 to 176.6 ft^3/s/h were low enough to reduce stranding.

However, the influence of down-ramp rates on stranding outcome is dependent on other secondary factors like season, time of day, and habitat types. For example, juvenile fish using shallow shoreline with complex structural features are less likely to detect or escape with receding water levels (Hunter, 1992). Along shorelines with steeper talus slopes, however, the down-ramp rate has less of an effect because fish can more readily follow decreasing water levels (Halleraker and others, 1999, Schmutz and others, 2015). Bauersfeld (1978) and Phinney (1974) concluded that the same down-ramp rate had less of an impact at higher elevational stage changes and more impact on stranding at lower elevational stage changes. Comparing stranding on gravel bars and side-channel habitats, Bradford (1997) found that ramping rate was less important for the gravel bar trials and very important in the side-channel stranding trials. Further demonstrating the effect of the down-ramp rate can differ depending on habitat type, Pflug and others (1989) found that down-ramp rates of 1,000, 2,000, and 5,000 ft^3/s/h did not influence stranding mortality on the Skagit River because fish tended to occupy pothole habitat types not dewatered by the fluctuating flows. These studies on the Skagit River ultimately determined that a down-ramp rate of 3,000 ft^3/s/h or less was sufficient to minimize stranding (Connor and Pflug, 2004). Berland and others (2004) concluded that time of day directly determined the impact of down-ramping rates on stranding and when day and night data were pooled together, there was no significant difference between fast and slow down-ramp rates.

On the Colorado River, down-ramp rates during the pre-EIS operational period were twofold higher than the MLFF period. The overall mean maximum daily down-ramp rate was 2,628 ft^3/s/h and 1,309 ft^3/s/h under the pre-EIS and MLFF periods, respectively. Current LTEMP operations increased the down-ramp rate from the MLFF rate of 1,309 ft^3/s/h to 2,500 ft^3/s/h, close to the pre-EIS down-ramp rate. However, these higher down-ramp rates are still considerably less than those in other rivers where fish stranding has been observed (Connor and Pflug, 2004). For comparisons, refer to the relationship between the mean monthly standard deviation in down-ramp rate and the mean monthly down-ramp rate for the pre-EIS operations and the MLFF period (fig. 3). Owing to the greater variability in pre-EIS down-ramp rates, the unpredictable nature of the hydropeaking regime across months and seasons may have had a greater effect on fish stranding than current rates.

Frequency of Fluctuating Flows

Fish behavior and habitat use are influenced by the frequency of flow fluctuations. While some literature addresses the effect of hydropeaking regimes on fish behavior, frequency of fluctuations in relation to stranding events is not well studied as only 13 percent of the stranding literature assessed this factor. Fish have been found to adapt to frequent and predictable hydropeaking patterns. In artificial channel experiments, Halleraker and others (2003) found that fish exposed to frequent and similar fluctuating flow patterns had low stranding rates. With a predictable and consistent flow fluctuation regime on the Colorado River, most age-0 trout remained along the wetted edge at minimum flow levels and did not follow changes in the shoreline edge during higher flows at night (Korman and others, 2009). Stranding studies on the River Nidelva in Norway that evaluated sudden and large changes in flow found that fish were unable to avoid stranding after repeated experiments under unpredictable flow patterns (Hvidsten, 1985; Saltveit and others, 2001). On the Columbia River, Anglin and others (2006) found that reducing the frequency of fluctuating flows from 10 to 5 times per week would reduce stranding events.

Life History Stage and Fish Size

Life history stage and fish size can influence susceptibility to stranding. Following emergence from spawning nests (redds), the fry and parr stages are more susceptible to stranding than larger sized fish (Hvidsten 1985; Hubert and others, 1994; Saltveit and others, 2001, Halleraker and others, 2003; Almeida and others, 2014; Boavida and others, 2015). In their larval stage, fish typically occupy the interstitial spaces between cobble/gravel where eggs are deposited, often in deeper water that renders them less susceptible to changing water levels (Hayes and others 2019, Moreira and others 2019). If located in shallow water, inter-gravel flow avoids possible desiccation. Fry and parr use nearshore habitat and are therefore more susceptible to stranding than larger adult fish.

Dauwalter and others (2013) found that age-0 rainbow trout stranded more often than larger sized rainbow trout on the South Fork of the Boise River. Salmonids up to 50 millimeters (mm) in length were highly vulnerable to stranding but their susceptibility rapidly declined with increasing size (Pflug and others, 1989; Halleraker and others, 2003). In artificial stream experiments, brown trout with a fork length less than 50 mm stranded at twice the rate compared to 75-mm fish (Halleraker and others, 2003). In other studies, Steelhead greater than 40 mm in fork-length were found to be less vulnerable to stranding than smaller sized fish (Pflug and others, 1989; Hunter, 1992). As fish continue to grow in length, they rely less on nearshore habitat which decreases their risk of being stranding. Adult fish are rarely stranded except under very large and rapid decreases in water levels (Hunter, 1992; Armstrong and others, 1998).

The effect of fish size on vulnerability to stranding has implications for the seasonal timing of TMFs downstream from Glen Canyon Dam. In Glen Canyon, rainbow trout primarily spawn in early spring (Korman and others, 2011b; Avery and others, 2015) and following emergence (fig. 4) small fry are common from midsummer to fall (fig. 5; Korman and others, 2011b). Less definitive information is available for brown trout in

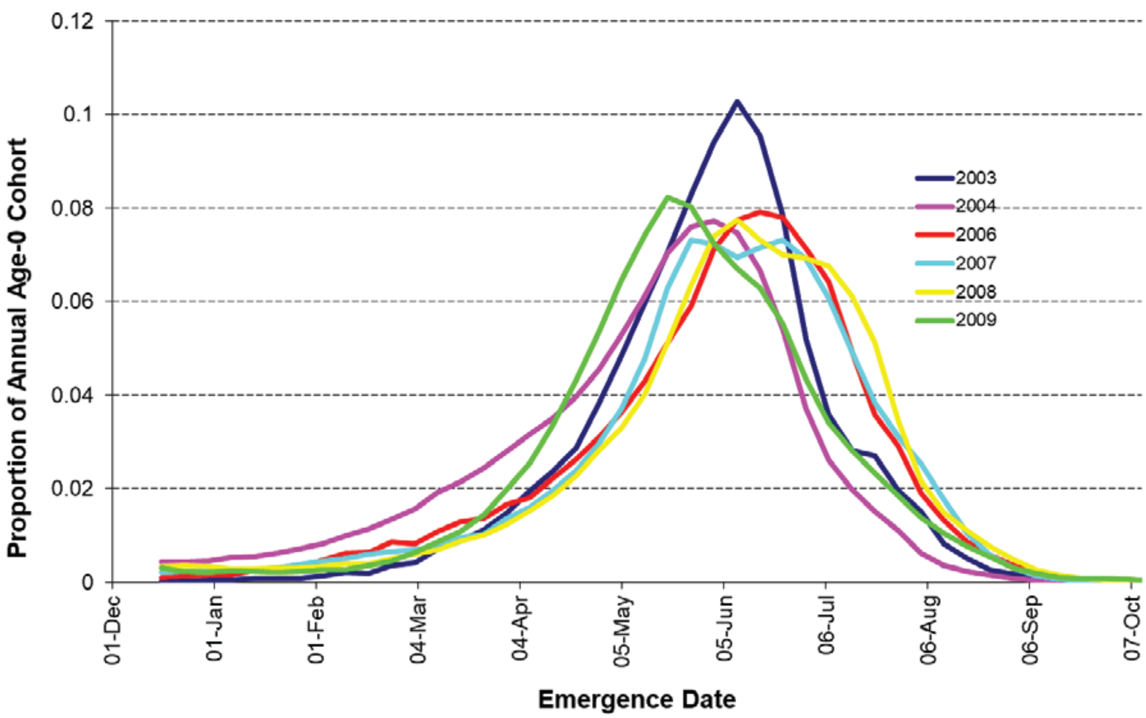

Figure 4. Plot showing predicted emergence date for rainbow trout in Glen Canyon, Arizona, based on the capture of fry in the spring through fall of each year, their individual fork lengths, and daily age-length relationships for each year. Generated from data in Korman and others, 2011a.

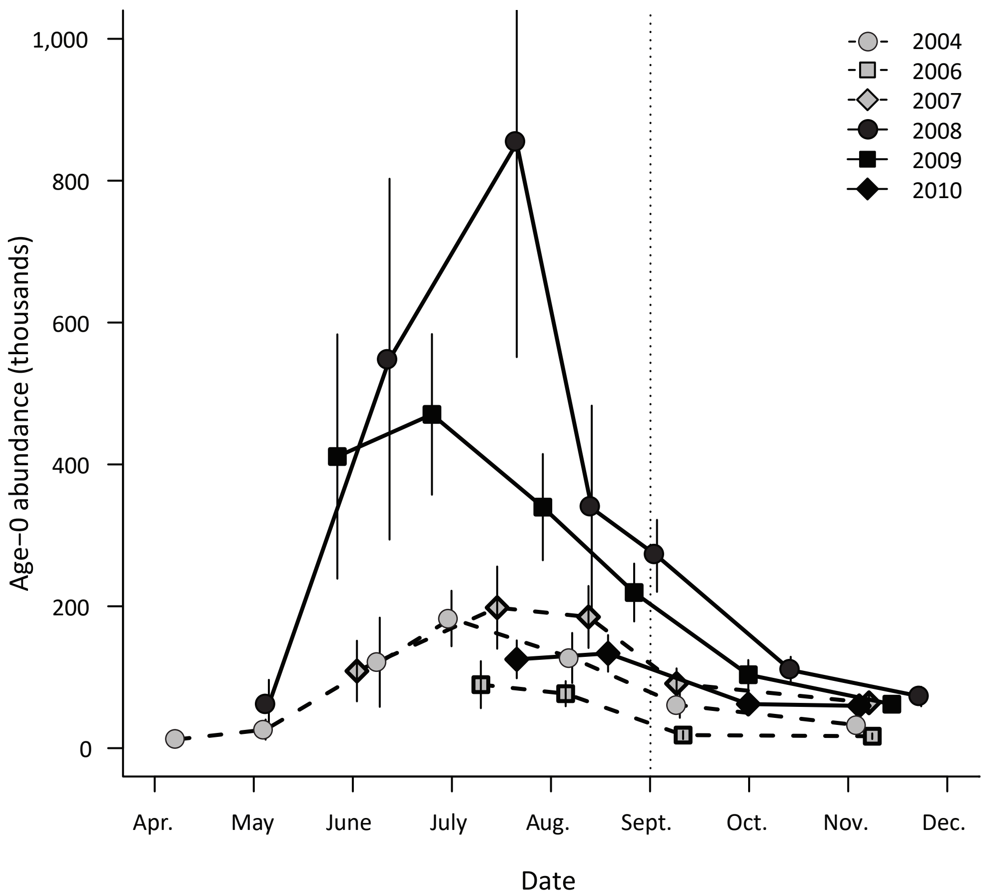

Figure 5. Plot showing seasonal trends in the total abundance of age-0 rainbow trout in the Lees Ferry reach by year between 2004 and 2010 (no data collected in 2005). The estimated reach-wide abundance during the mid-July sample period in 2003 is also shown. The vertical lines on each data point show the 95-percent confidence limits for the age-0 abundance estimates. Dotted line represents September 1. Reproduced without modification from Korman and others (2011b).

Lees Ferry but spawning likely occurs between late November and January (fig. 6; Runge and others, 2018). However, age-0 brown trout are rarely caught during the early sampling season (January) even though their spawning period is likely 3 months in advance of rainbow trout. The disparity in age-0 rainbow trout and brown trout catch could be due to species misidentification at smaller sizes. The catch difference between the two trout species could also suggest that brown trout fry are not occupying the wetted edge. During late summer and fall, the catch of slightly large (greater than 100mm) brown trout increases substantially. If age-0 brown trout with a fork length less than 75 mm are using different types of habitats (for example, offshore vegetation) other than proximal to the shoreline edge, then they are not as likely to be stranded by TMFs scheduled in the spring.

Channel Morphology Factors Influencing Fish Stranding

Fish typically occupy shoreline habitat during their early life stages. The morphologic features of shoreline habitat, such as substrate, lateral slope, and structural complexity of habitat, can affect stranding rates. For example, as water levels recede, fish can become stranded in interstitial pore spaces within the substrate, on shorelines with shallow lateral slopes, or entangled in vegetated habitats.

Substrate

Coarse gravel and cobble substrate can lead to higher rates of stranding because fish become trapped within the interstitial spaces as flows drop (Pflug and others, 1989; Hunter, 1992; Bradford and others, 1995; ICF Jones and Stokes, 2009; Young and others, 2011; Dauwalter and others, 2013; Hauer and others, 2014). Heggenes (1988) found that brown trout fry sought coarse gravel substrate (50–70 mm) over finer sized materials (8–16 mm) because the former has larger interstitial spaces that provide greater refuge from high water velocities and predation. This cover-seeking behavior can be more pronounced in cobble substrate, particularly at colder water temperatures (Bradford and others, 1995). In other studies, where stranding was compared between homogenous and heterogenous substrate, a higher degree of stranding occurred in heterogenous substrate (Auer and others 2017; Boavida and others, 2015).

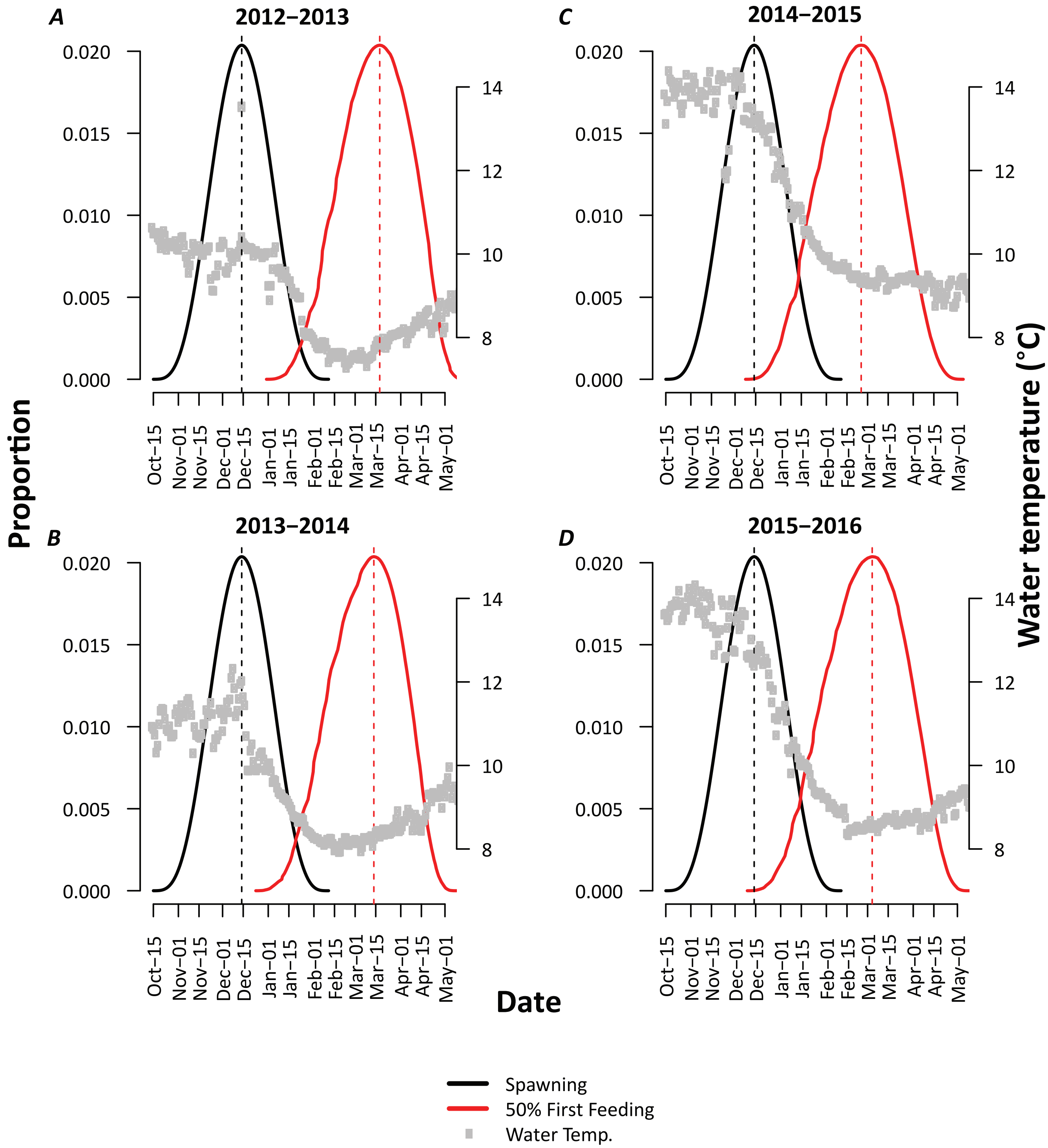

Figure 6. Plots showing estimated spawn and predicted emergence timing for brown trout in Glen Canyon, Arizona for 2012–2013 (*A*), 2013–2014 (*B*), 2014–2015 (*C*), and 2015–2016 (*D*). Emergence timing indexed by the date at which 50 percent of fry are estimated to have begun feeding. Predictions are based on assumed spawn timing and water temperature: temperature determines number of accumulated thermal units, which predicts amount of time required between fertilization and fry feeding. Water temperature (Water Temp.) data from Korman and others (2011b). %, percent.

In Glen Canyon, some fine-grained sediment delivered by ephemeral tributaries is deposited on riverbanks following HFEs, and the encroachment of riparian vegetation decrease the extent of unvegetated cobble and sand bars available (Sankey and others, 2015; Kasprak and others, 2021). A decrease in cobble bars could reduce the amount of habitat conducive to stranding small fish; however, an increase of vegetated shoreline could increase stranding risk as small fish can become entangled in riparian vegetation.

Lateral Slope

Higher rates of stranding are expected over shorelines with low-angle lateral slopes (in other words, low-angle banks directly perpendicular to upstream-downstream). In low-angle shoreline habitats, receding water levels may be harder to detect and causes a fish to travel farther to remain in water. Stranding occurs more frequently on slopes less than 5 percent (Young and others, 2011). Olson and Metzgar (1987) found that the highest rates of stranding occurred over slopes less than 4 percent. Bradford and others (1995) and Bradford (1997) conducted experiments using rainbow trout and Coho salmon (*Oncorhynchus kisutch*) in artificial streams and found that stranding rates were higher in channels with a 2 percent slope than a 6 percent slope. Bell and others (2008) found that most fish were stranded on slopes less than 6 percent compared to steeper ones. Tiffan and others (2002) found that, when it came to habitat use by sub-yearling Chinook salmon (*Oncorhynchus tshawytscha*), more fish occupied lower angle lateral slope habitat than steeper slopes. In developing a model for estimating stranding risk areas, lateral slope was the primary predictor of habitat use and subsequently stranding risk (Tiffan and others, 2002). Similarly, Tuhtan and others (2012) identified slope as a factor that can determine the extent of a stranding risk area in their modeling approach (refer to the “Hypsometric Analysis” section).

Once stranded, fish must return to water in time to avoid desiccation and hypoxia (Boumis and others, 2014). The ability of a fish to rapidly propel itself and return to water through rolling and springing (Gibb and others, 2011) is accomplished by sensing the downslope direction (Boumis and others, 2014). Once stranded, recovery is more likely for fish located on steeper than lower angled slopes because of less travel distance and ability in detecting downslope.

In Glen Canyon, age-0 rainbow trout make more use of low-angle shoreline habitats shortly after emergence (fig. 7). As a result, the population in stranding-sensitive low-angle habitat is greatest in May and June, and then declines as the fish grow and make the ontogenetic shift to shorelines with steeper talus slopes. The timing in the use of low-angle shorelines is an important consideration in implementing TMFs.

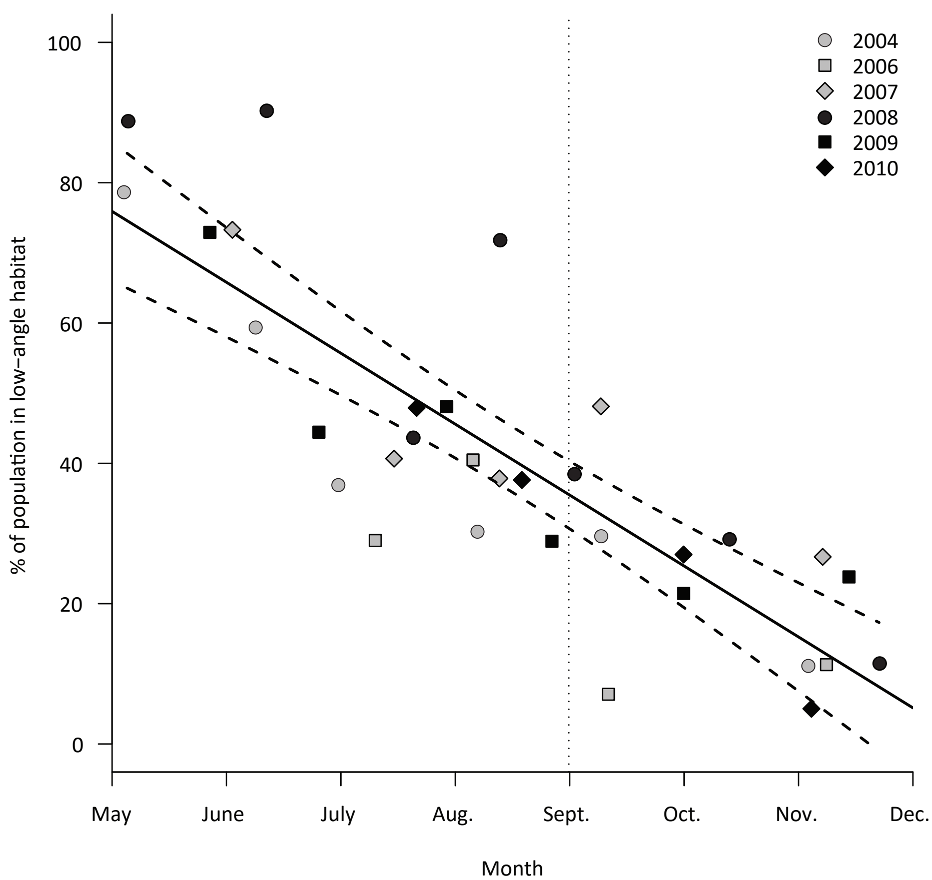

Figure 7. Plot showing the percentage of the age-0 rainbow trout population in Glen Canyon, Arizona, in low-angle shoreline habitat between May and December, 2004–2010. The solid and dashed lines represent the mean relationship estimated using data from all years and the 95-percent confidence limits, respectively. Reproduced without modification from Korman and others (2011b). %, percent.

Structural Features

Potholes and pools, side channels, and vegetation are examples of structural features that provide cover but also can lead to higher rates of stranding because they make it difficult for fish to detect changing water levels (Puffer and others, 2019; Vehanen and others, 2000). Stranding can be higher in systems with more varied channel morphology, like braided side-channels. Bradford (1997) found that under similar down-ramp rates stranding occurred more often in side-channels than on gravel bars. As the water level drops, these features can become disconnected from the main channel, isolating fish in pools that can later be completely dewatered. Increased mortality prior to dewatering may also occur due to higher levels of predation and elevated water temperatures (Hunter, 1992).

Pflug and others (1989) conducted fish stranding surveys on the Skagit River and found that different discharge levels of 989, 2,013, and 5,015 ft^3/s had little effect, since fish occupied pothole features that were not dewatered by flow decreases. They found that at minimum flows (2,790 ft^3/s), potholes were connected to the river. In another study, recontouring gravel bars to reduce entrapment pockets lowered stranding rates during down-ramping (Irvine and others, 2015).

In 1965, high sustained flows of ~65,000 ft^3/s were intentionally used to lower the elevation of the tailwater in Glen Canyon, a 15.6-mile-long stretch of river below Glen Canyon Dam downstream to Lees Ferry, to increase flow conveyance and power production from Glen Canyon Dam (Grams and others, 2010). These high flows exported large quantities of sediment and lowered the elevation of the channel bed (Melis and others, 2012). Overall, these high flows coarsened the channel bed and adjacent cobble bars (Melis and others, 2012). The elimination of large pre-dam floods increased vegetation in the “new high-water zone” above normal flow operations (Carothers and Brown, 1991; Sankey and others, 2015). Since the implementation of the MLFF pattern and its occasional experimental flood, further channel changes have occurred along the lower, less colonized “fluctuating zone” (20,059–5,015 ft^3/s). In the fluctuating zone, the interstitial spaces of cobbles are filled in with fine-grained sediment, which makes it conducive for the establishment of semiaquatic and aquatic vegetation (Sankey and others, 2015; Kasprak and others, 2021). The tailwater section below Glen Canyon Dam has only a few ephemeral tributaries that contribute fine-grained sediment, yet owing to the annual supply of fine-grained sediment over 50 years and the periodic high-flow experiments, channel sediment has been resuspended and deposited at higher elevations along the shoreline (Schmidt and Grams, 2011). These changes have likely increased the extent of aquatic vegetation along the shoreline that could contribute the entrapment of small sized fish.

Diel Cycle (Day versus Night)

The influence of the diel cycle on the magnitude of stranding varies by fish species, season, and cover type. Juvenile salmonids have been observed to seek cover during the day to reduce their vulnerability to predation and to avoid high water velocity (Bradford and others, 1995). Studies have found that decreasing flows at night led to a reduction in stranding rates since fish are more active and more likely to change their position and therefore move with decreasing water levels (Bradford, 1997; Heggenes, 1988; Larranaga and others, 2018; Saltveit and others, 2001). Halleraker and others (2003) found that down-ramping at night reduced the risk of stranding regardless of the season, with higher stranding rates during all daytime experiments. Additionally, in the Skagit River, Connor and Pflug (2004) found that minimizing down-ramping events during the day greatly reduced stranding for salmon species. In contrast, Anglin and others (2006) found that fish were less active at nighttime and suggested higher stranding rates would occur at night. Olson and Metzgar (1987) also found that stranding rates of steelhead fry were three to five times greater at night than during the day.

An important aspect to daytime stranding compared to nighttime stranding is the air temperature and its effect on the mortality of stranded fish. The Colorado River flows through extremely arid land with air temperatures reaching highs of 40 to 45 °C by midday in summer. Owing to high temperatures, fish that are standing during the day may experience higher mortality compared to fish stranded at night (Cook and others, 2015). Conducting down-ramps during the day also has the potential benefit of increasing stranding rates owing to greater concealment behavior among age-0 fish during daylight hours (fig. 8) .

Owing to seasonal differences in trout spawning and emergence, the time-of-day during down-ramp and the duration of the minimum flow level required for stranding vulnerable life stages will be different for brown trout versus rainbow trout (see “Life History and Fish Size” section). This may require adjusting the timing of flow released at Glen Canyon Dam. For brown trout spawning periods, by the vernal equinox (March 21), direct sunlight strikes ~75 percent of the wetted shoreline by 10:00 in the morning and most of this direct sunlight exposure occurs in north-south oriented canyon sections (fig. 9). For rainbow trout spawning periods, by late spring (June 1), direct sunlight strikes ~75 percent of the wetted shoreline two hours earlier (07:57) in the morning (fig. 9).

Water Temperature

Most studies that evaluated the effects of water temperature on stranding found that juvenile salmonids were more likely to be stranded in winter when water temperatures are lower. This likely occurs because salmonid fry and parr are concealed within the interstitial spaces when water temperatures are low (Bradford and Higgins, 2001), which could increase their susceptibility to stranding. Lower water temperatures also tend to reduce the activity levels and swimming abilities of most salmonids (Elliott, 1994). Bradford (1997) found a difference in fish behavior during the diel cycle that was more pronounced during colder seasons, when salmonids conserve energy, than in the warmer spring and summer months. In flume experiments, Halleraker and others (2003) showed that water temperature was the most important factor influencing stranding rates as more stranding occurred at lower temperatures.

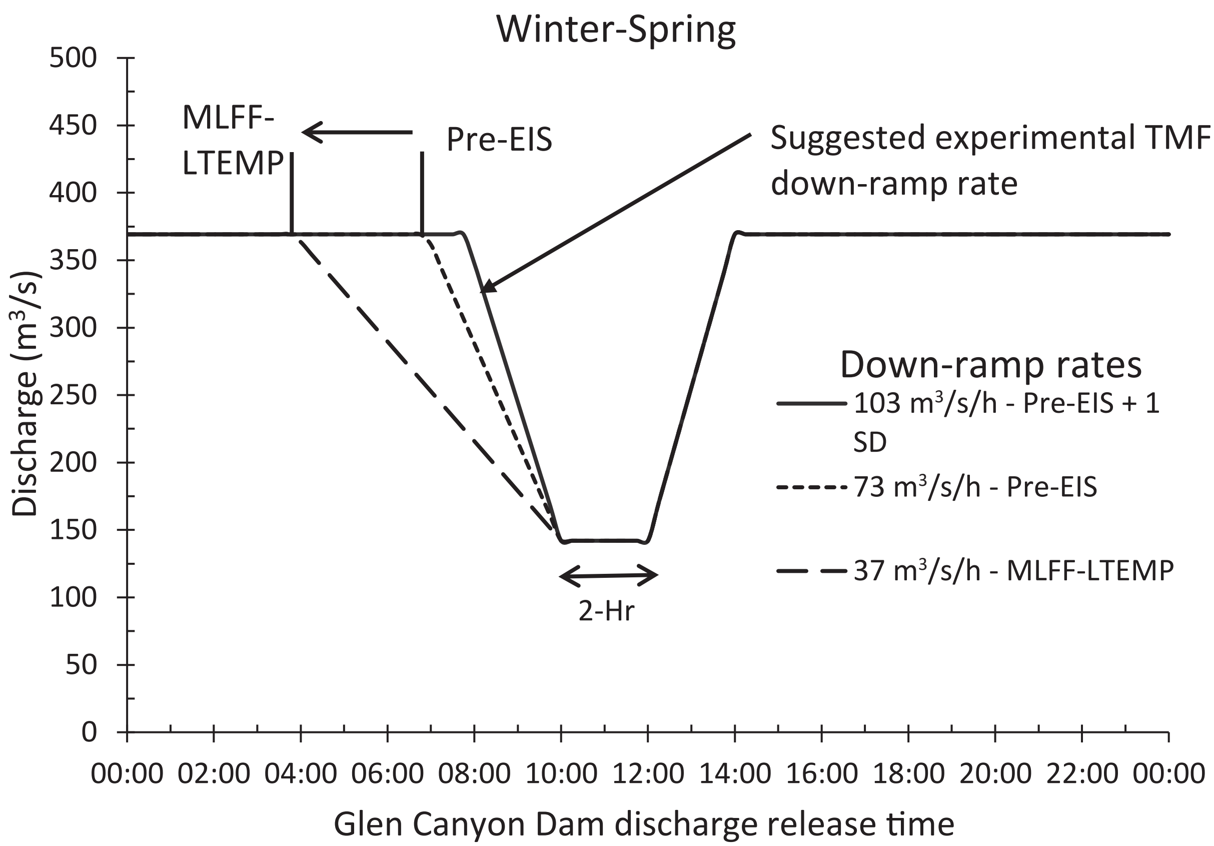

Figure 8. Daily hydrographs comparing three different down-ramp rates for Glen Canyon Dam (GCD). The down-ramp rates shown are based on maximum allowable rates during the Modified Low Fluctuation Flow Long-Term Enivornmental Management Plan (MLFF-LTEMP) period, the pre-Environmental Impact Statement (Pre-EIS) period, and for a suggested trout management flow (TMF) down-ramp. Hr and h, hour; m^3/s, cubic meter per second.

Because of the need to conserve energy during colder winter months, juvenile brown trout have been observed to seek shallow shorelines, which increases their susceptibility to stranding, when compared to other types of habitat used during the summer when water temperatures are higher (Vehanen and others, 2000; Saltveit and others, 2001; Halleraker and others, 2003). For instance, Vehanen and others (2000) found that brown trout used structural features that increase visual cover more often in winter months and sought habitat with higher velocity flows during summer months. Additionally, Vehanen and others (2000) observed that aggressive behavior in brown trout decreased during winter months and observed an overall reduction in swimming activity. This can lead to a higher rate of stranding in winter months owing to habitat fidelity and slower relocation response because swimming speeds slow as water levels decrease in hydropeaking rivers.

Heggenes (1988) identified that brown trout fry readily sought cobble gravel substrate as cover at temperatures ranging between 5 and 10 °C. Halleraker and others (2003) found that more juveniles were stranded at 7 °C relative to 11 °C during the nighttime. Similarly, Saltveit and others (2001) found that brown trout were more likely to be stranded during winter in water temperatures less than 4.5 °C than during summer and autumn in water temperatures greater than 9 °C. Bradford and others (1995) found that low temperatures (3–4 °C) could initiate fry concealment behavior in shallow shoreline habitat, which increased vulnerability to stranding. In later studies, Bradford (1997) found that stranding rates of juvenile Chinook salmon were sixfold higher at 6 °C than at 12 °C.

For comparison, winter and spring water temperatures for the Colorado River are frequently at 8 °C or higher (fig. 10). This is slightly above the water temperature levels (7 °C or less) that induces concealment behavior of early life history stages of rainbow trout or brown trout (Bradford, 1997; Halleraker and others, 2003), which in turn increases the susceptibility to stranding. Therefore, seasonal variation in water temperature is unlikely to influence susceptibility to stranding because temperatures are typically greater than 7 °C, and seasonal timing of TMFs should be determined by the period when small fry are present.

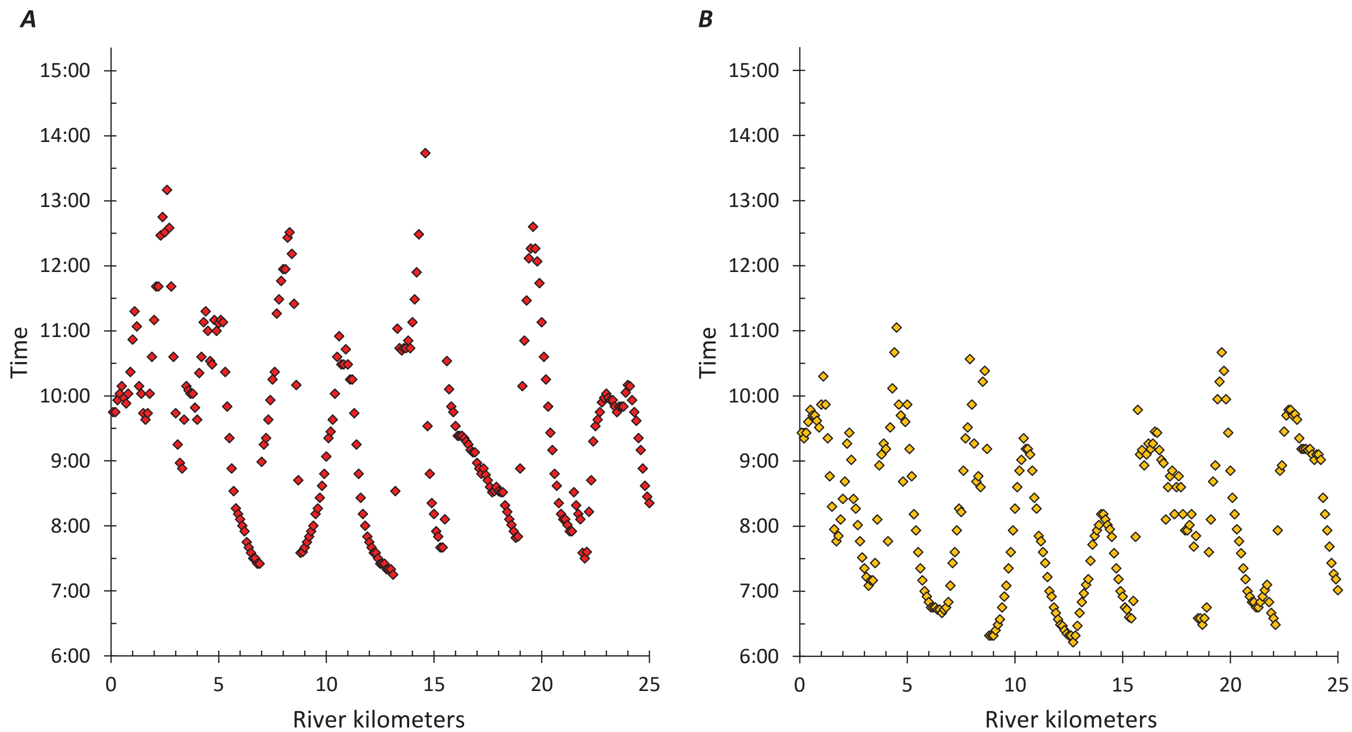

Figure 9. Plots showing estimated time of direct sunlight striking the water surface of the Colorado River throughout Glen Canyon in relation to the river kilometers from Lees Ferry upstream to Glen Canyon Dam for March 21 (*A*) and June 1 (*B*). Estimates are based on mid-river points at 100-meter intervals over a 25- kilometer length (Yard and others, 2005).

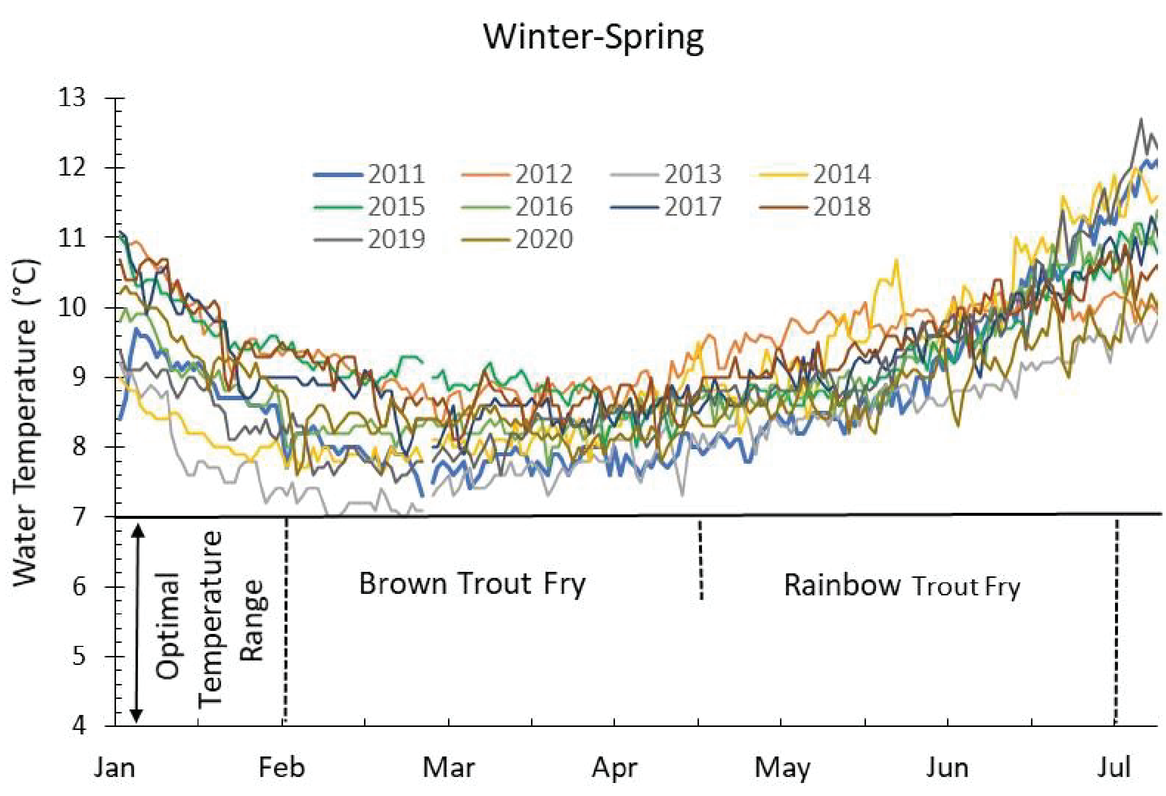

Figure 10. Plot showing daily minimum water temperatures (°C) measured at the Lees Ferry gage between 2011 and 2020.

Population Effect of Stranding Mortality Rate

Population-level effects of stranding mortality have been assessed in a limited number of studies. Hedger and others (2018) found that stranding mortality occurring in the winter and spring had greater negative effects on long-term population trends than stranding in other seasons. Additionally, owing to compensatory survival, one year of stranding had little to no effect on overall population abundance, whereas repeated stranding over six years reduced the abundance of small fish by two-thirds (Hedger and others, 2018). In the Dale River in Norway, large juvenile stranding events reduced the number of returning spawners, thus driving down population abundance over time (Sauterleute and others, 2016). Similarly, on the River Alta in Norway, hydropeaking flows led to stranding of juvenile size classes resulting in smaller population abundances over a 20-year period (Ugedal and others, 2008).

In a long-term monitoring program on the Hanford Reach of the Columbia River, Harnish and others (2014) found that spring flows designed to protect Chinook salmon redds from dewatering increased population abundance by 217 percent and subsequent flows designed to protect juveniles from stranding resulted in a 130-percent increase in overall productivity.

In Glen Canyon, the dewatering of redds resulting from a daily flow range of 5,000 to 20,000 ft^3/s, did not result in a reduction in fry abundance, even in a year where as many as 50 percent of the redds were dewatered (Korman and others, 2011a). The relationship between egg deposition and fry abundance suggests there is a strong compensatory response in survival rates. That is, dewatering of redds reduces the number of fry that emerge, but the survival rates of fry from redds that are not dewatered is higher, mitigating the population-level effects of dewatering. TMFs aim to reduce the survival rate of post-emergent fry. It is uncertain whether density-dependent survival at this life stage will compensate for losses resulting from TMF-related stranding, however, repeated stranding events in a year could potentially overcome compensatory survival responses.

Hypsometric Analysis

High densities of rainbow trout in Glen Canyon resulting from occasional large recruitment events, in conjunction with low prey availability, can lead to poor growth and population collapses (Korman and others, 2021). Young rainbow trout from very abundant year classes disperse into Marble Canyon, which can result in higher densities of rainbow trout near the Little Colorado River confluence (Korman and others, 2016) that can negatively affect threatened humpback chub (Yackulic and others, 2018). Brown trout abundance in Glen Canyon has been increasing rapidly since 2014 (Runge and others, 2018; Yackulic and others, 2020), and could result in negative effects on the rainbow trout population and fishery in Glen Canyon, and on humpback chub near the Little Colorado River. The LTEMP EIS describes the use of TMFs to limit the production of age-0 rainbow trout in some years due to the potential negative consequences of high recruitment. Although not discussed in the LTEMP, TMFs also have the potential to help control brown trout abundance in Glen Canyon, an issue of increasing concern in the Glen Canyon Dam Adaptive Management Program. TMFs are a modified Glen Canyon Dam operation that purposefully attempts to strand age-0 rainbow trout by holding flows high and steady (i in fig. 11*A*), and then rapidly dropping them (ii in fig. 11*A*) to a lower level for a short period (iii in fig. 11*A*). Such flows are avoided in many regulated rivers owing to concerns about their negative effects on native fish populations, thus these same flow characteristics have the potential to limit trout recruitment in Glen Canyon.

An understanding of the habitat requirements and spatial distribution of age-0 trout in Glen Canyon is critical to designing an effective TMF. Under normal hydropeaking operations of Glen Canyon Dam, flows are typically held at steady low flows during nighttime when power demand is low and raised to higher and relatively steady flows during the day when power demand is high (fig. 12*A*). Very young and small salmonids, such as age-0 rainbow and brown trout, require shallow depths and low velocities. In rivers without hydropeaking, age-0 fish use immediate shoreline areas close to the wetted edge, where depth and velocities are low and therefore most suitable. In Glen Canyon, Korman and Campana (2009) found that most age-0 rainbow trout do not follow the immediate shoreline area near the wetted edge as it rises and falls with hydropeaking over a 24-hour period, but instead largely remain at or just below the daily minimum flow elevation (fig. 12*A*). Under a TMF with a prolonged high-flow period (for example, fig. 11*A*), age-0 trout are expected to eventually colonize the higher elevation immediate nearshore area over a period of days or weeks, and some of these individuals are expected to be stranded when flows are rapidly dropped to a low level. Stranding rates may be particularly high over lower-angle shorelines (in a direction perpendicular to river flow) as fish are more likely to get trapped as flows are dropped. Stranding rates are also expected to be higher at locations that are farther from the low-flow wetted edge, as fish are more likely to get trapped when forced to move over a longer route. Finally, stranding rates could be particularly high when the TMF low flow is less than 5,000 ft^3/s, because many areas below elevations inundated by this flow have high densities of algae or aquatic vegetation that would act as a sieve and block the passage of age-0 rainbow trout to the low-flow wetted edge during the flow drop.

TMFs can be categorized into three forms. The first is described in the LTEMP EIS (Bureau of Reclamation and National Park Service, 2015) and involves raising flows above normal operating levels for a prolonged period so that age-0 rainbow trout colonize the newly wetted habitat (i in fig. 11*A*). Flows are then dropped rapidly (ii in fig. 11*A*) to a lower flow (iii in fig. 11*A*). We term this type of TMF a “go high then low” operation. During operations influenced by Lake Powell and Lake Mead equalization rules, such as those that occurred in 2011, flows during spring and summer are higher than under normal operations, and relatively steady. Under these conditions, age-0 rainbow trout have been shown to colonize high elevation habitat (Avery and others, 2015), so there would be no need to raise the flows beyond equalization

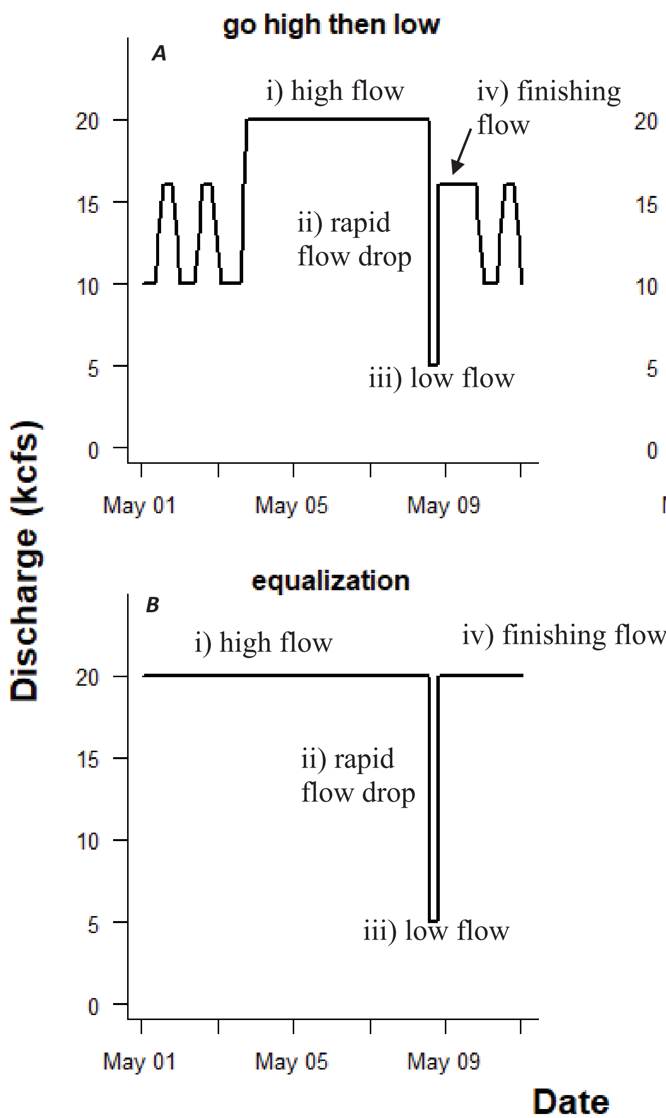

Figure 11. Examples of Trout Management Flows (TMF). *A*, Discharge is increased to a high and steady level (i) to draw age-0 trout into nearshore habitats close to the elevation inundated by the maximum flow (for example, fig. 12*A*). Flows are then rapidly reduced (ii) to a low level (iii) to strand age-0 trout. Following the low flow, flow is increased rapidly (iv) prior to resuming normal operations, to increase the minimum flow and reduce down-ramp rate in Grand Canyon. This TMF cycle can be repeated over the window when age-0 trout emerge from spawning redds. *B*, Flows are already high and steady owing to reservoir equalization or balancing operations, so there is no need to increase flows or implement a finishing flow. *C*, Normal hydropeaking operations occur prior to the rapid flow drop. kcfs, thousand cubic foot per second.

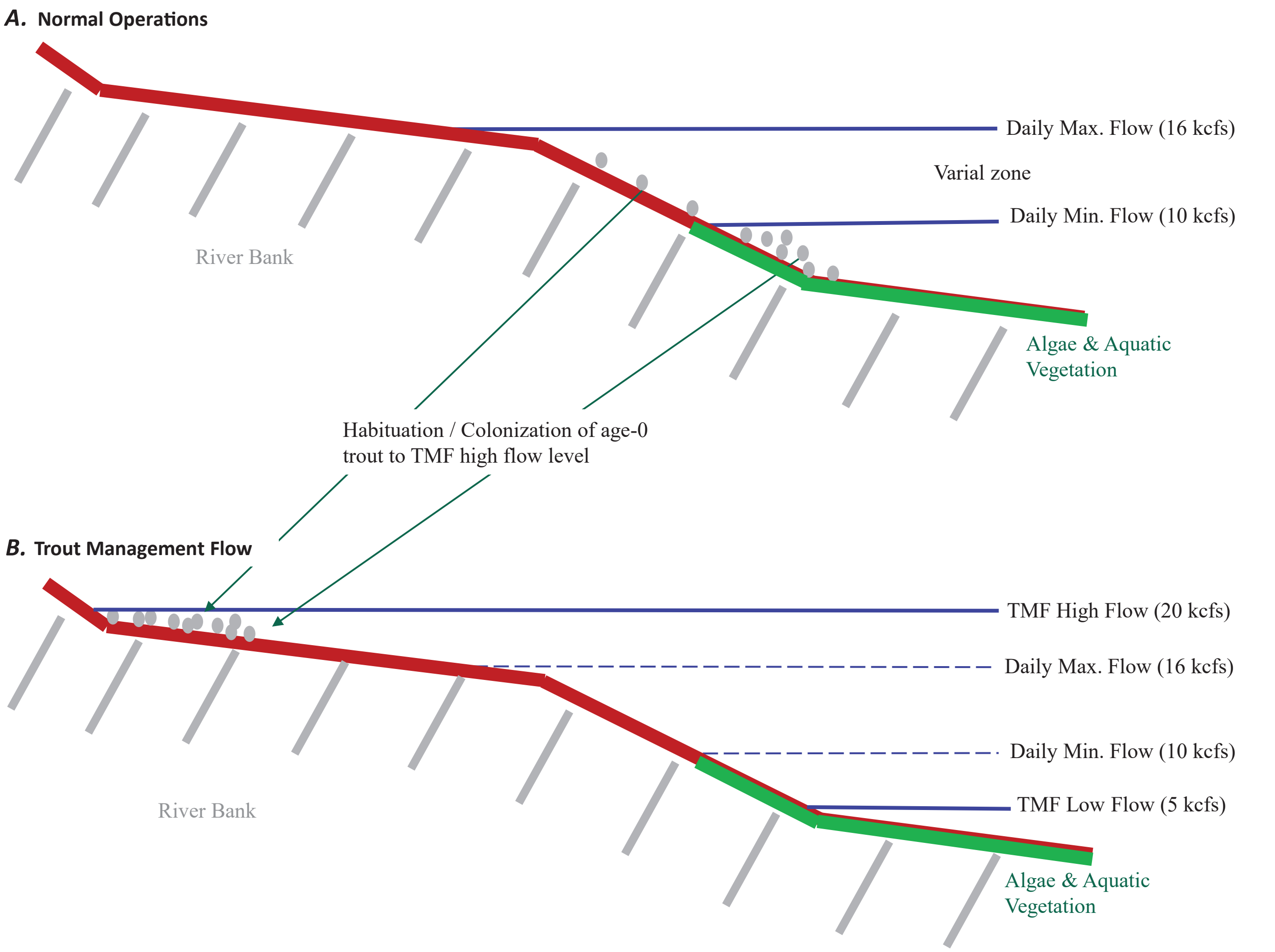

Figure 12. Conceptual model showing the location of age-0 rainbow trout under normal hydropeaking operations (*A*) and at the end of the colonization phase of a Trout Management Flow (TMF; *B*). *B*, age-0 rainbow trout will be stranded when the TMF high flow is rapidly reduced to the TMF low flow (see fig. 11*A*, *B*). It is also possible to strand age-0 rainbow trout under normal operations by rapidly reducing flows from the daily maximum (max.) level to the TMF low flow (for example, 16,000 to 5,000 cubic feet per second ; see fig. 11*C*). kcfs, thousand cubic foot per second; min., minimum.

levels to implement a TMF (fig. 11*B*). Such "equalization" TMFs will likely be more effective than the "go high then low" TMFs because a greater proportion of the age-0 population would colonize high elevation habitats owing to the longer duration of the high and steady flow. Stranding rates during a flow drop would also be expected to be higher owing to higher flow levels prior to the flow drop. The third form of TMF we consider in this report is termed a "go low not high" strategy. Flows are rapidly decreased from the normal daily peak flow to a level lower than the normal daily minimum flow (fig. 11*C*). This results in a smaller flow drop compared to the "go high then low" and "equalization" TMFs, but the operation does not incur negative hydropower and sediment storage effects associated with the high-flow element of the TMF cycle. In the example provided, the effective flow drop is about 5,000 ft^3/s (for example, from 10,000 to 5,000 ft^3/s, rather than 16,000 to 5,000 ft^3/s) because few age-0 rainbow trout colonize the varial zone, which occurs at elevations between the daily minimum (10,000 ft^3/s) and maximum (16,000 ft^3/s) flows. However, owing to high algae population densities and submerged aquatic vegetation at elevations inundated by 5,000 ft^3/s, a substantial proportion of age-0 rainbow trout could potentially get stranded due to a vegetation-sieving effect (fig. 12*B*).

The objective of this section of this report is to predict stranding risk for age-0 rainbow trout in Glen Canyon under a range of TMFs, using existing bathymetric data, and flow and habitat suitability models. We use a two-dimensional (2D) steady flow model to predict wetted area, water depth, and water velocity over a series of 5-meter grid cells for all of Glen Canyon as a function of discharge. These predictions are combined with models that describe the suitability of different depths and velocities for age-0 rainbow trout to predict locations in Glen Canyon that would be used by age-0 rainbow trout during the high-flow component of the TMF (fig. 11). The flow model also predicts which of these cells will be dewatered as flows are dropped from the high- to low-flow components of the TMF. A digital elevation model is used to calculate the horizontal slope and distance between each cell at the high flow and the water's edge at the low flow. These predictions are used to quantify stranding risk for each cell, which is summed over all cells to determine the expected effect for Glen Canyon. The model is run using a range of low and high flows to identify how different combinations affect stranding risk. Finally, we use a one-dimensional (1D) unsteady model to predict the effects of various TMFs on the shape of the hydrograph at locations in Grand Canyon that are used by humpback chub. The unsteady flow modelling can be used to define a TMF that would be effective in Glen Canyon (in other words, a large and rapid flow drop) but with limited effects on the down-ramp rate and minimum flow in Grand Canyon, by taking advantage of wave attenuation dynamics. Mitigating potential impacts, such as drastic flow fluctuations or rapid down-ramp rates, on downstream habitats utilized by threatened humpback chub, remains a goal outlined in the LTEMP. We will show that downstream effects can be virtually eliminated by using a short-duration low-flow period (iii in fig. 11*A*), followed by a rapid increase to a higher flow level (iv in fig. 11*A*). We refer to this last part of the TMF cycle as a "finishing flow." Low flows in Grand Canyon could have negative effects on the food base or native fish, and element iv of the proposed TMFs are designed to minimize the magnitude and duration of low flows downstream of Glen Canyon.

Methods

Hydrodynamic Modeling

Glen Canyon is a 15.6-mile-long tailwater located between Glen Canyon Dam and Lees Ferry, Arizona. There are no perennial streams that enter the tailwater, thus water quality and flow is almost completely determined by outflows from Glen Canyon Dam. Grams and others (2007) investigated geomorphic changes to the reach following dam construction and concluded that the riverbed was transformed from a sand-bedded alluvial reach to a stable and rarely mobilized gravel bedded river. Because the input boundary conditions are well constrained, this reach is well suited for flow modeling.

Wright and others (2024) constructed a two-dimensional hydrodynamic model of the reach. The model used the Flow and Sediment Transport with Morphologic Evolution of Channels (FaSTMECH) solver in the International River Interface Cooperative streamflow modeling package (Nelson and others, 2016). The model grid was developed from a full channel digital elevation model (DEM) derived by combining bathymetric and topographic data collected from March 2013 to February 2016 (Kaplinski and others, 2022). Channel bathymetry was mapped using multibeam and single-beam echosounders, and subaerial topography was mapped to elevations higher than historical flow levels (approximately 300,000 ft^3/s) with a combination of photogrammetry from aerial photography and ground-based total-stations. For computational efficiency the 1-meter (m) resolution DEM was used to map elevations to the computational grid elements, which were approximately 5-m by 5-m in size. The model was run for upstream discharges ranging from 1,000 to 40,000 ft^3/s in 1,000 ft^3/s increments and from 40,000 to 70,000 ft^3/s in 5,000 ft^3/s increments. Boundary conditions for all model runs include a steady discharge at the upstream boundary (Glen Canyon Dam), and a constant water-surface elevation at the downstream boundary (Lees Ferry). The results for each model run are stored in tables that list, for each grid cell, the predicted water-surface elevation, depth, bed elevation, and velocity and shear stress solutions (Wright and others, 2024). We used the information from these output tables as input to the stranding risk model (described in the "Stranding Risk Modeling" section).

Comparison of model output water-surface elevations agree well with stage-discharge rating curves from 20 historical cross sections located throughout the reach (Wright and others, 2024; Grams and others, 2007, 2010). The mean residuals from the comparison ranged from −0.03 to 0.12 m and the mean absolute residuals ranged from 0.08 to 0.18 m and indicate good agreement between the measured and modeled water-surface elevations. The mean error on discharge, a measure of how well the model solution converged, averaged 0.5 percent and was below 2 percent for all flows modeled. These results show that the model outputs are well calibrated and sufficient for the purposes of stranding risk modeling.

Stranding Risk Modeling

The stranding risk (SR) for any 25-m^2 grid cell between TMF high- and low-flow elements (fig. 11) is the product of the probability of age-0 rainbow trout utilizing the cell (p_{use}) at the high flow, and the probability that age-0 rainbow trout in that cell will be stranded when flows are dropped to the low flow (p_{strand}),

$$SR = p_{use} \times p_{strand} \tag{1A}$$

These terms are defined more explicitly in,

$$SR_{ihi,ilo,ix,iy} = HSI_{ihi,ix,iy} \cdot \left[\beta_s \cdot Slope^{-1} + \beta_d \cdot Dist_{ihi,ilo,ix,iy} + \beta_b \cdot Bed_{ihi,ilo,ix,iy}\right] \tag{1B}$$

The *HSI* and the terms within square parentheses in equation 1*B* represent p_{use} and p_{strand} in equation 1a, respectively. *SR* is the stranding risk for a cell in row *ix* and column *iy* when flows drop from the TMF high flow *ihi* to TMF low flow *ilo*. *HSI* is the habitat suitability of the cell at the high flow, *Slope* (*s*) and *Dist* (*d*) are the cross-sectional slope and distance over the exposed terrain from the cell at the high flow to the first cell along the cross section that is wet at the low flow, and *Bed* (*b*) is an index of the complexity of the substrate over the exposed terrain. The *β*'s are coefficients that quantify the effect of these stranding covariates on stranding risk.

Habitat suitability for each cell at the high flow of the TMF was calculated based on predicted depths and velocities and habitat suitability curves (Bovee, 1986; Porter and Rosenfeld, 1999; Rempel and others, 2012). Small age-0 rainbow trout prefer locations with shallow depths and low water velocity (fig. 13). Predicted depths ($D_{ihi,ix,iy}$) and velocity ($V_{ihi,ix,iy}$) at each cell for a particular high-flowvalue, estimated by the 2D flow model, were used to calculate depth ($HSI\_D_{ihi,ix,iy}$) and velocity ($HSI\_V_{ihi,ix,iy}$) suitability by linear interpolation of HSI curves (fig. 13), and overall suitability (*HSI*) was then calculated using,

$$HSI_{ihi,ix,iy} = HSI\_D_{ihi,ix,iy} \cdot HSI\_V_{ihi,ix,iy} \tag{2}$$

The existing literature on stranding did not contain sufficient information to define effect sizes for stranding covariates (*β*'s in eq. 1*B*, see "Literature Review" section). Thus, we defined three simple models of stranding risk based on different assumptions. Our first model was based only on the ratio of suitable habitat that is exposed between the high- and low-flow elements of a TMF relative to the total suitable habitat at the high flow,

$$SR\_H_{ihi,ilo} = 100 \cdot \frac{\sum HSI_{ihi,ix,iy} \middle| D_{ilo,ix,iy} = 0}{\sum HSI_{ihi,ix,iy}} \tag{3A}$$

where the numerator is the sum of *HSI* values across cells at the high flow of the TMF that are exposed at the low flow of the TMF ($D_{ilo,ix,iy}$=0), and the denominator is the sum of *HSI* values across all cells that are wet at the high flow, including those that are not exposed when flows drop to the low flow of the TMF. Thus, *SR_H* (*H* for habitat suitability) is the percentage of the total suitable habitat available at the high flow that is exposed when flow is decreased to the TMF low flow. Model *SR_H* does not consider the slope, distance, and bed complexity that an age-0 trout would encounter as flows drop from high to low values. It uses *HSI* as an imperfect surrogate for p_{use} in equation 1a and assumes that $p_{strand} = 1$ for all cells that are dewatered. Note this and later models assume there has been sufficient time to colonize habitat inundated by the high-flow element of the TMF.

Our second index of stranding risk accounts only for slope and distance factors on stranding risk, and is calculated using,

$$SR\_C_{ihi,ilo} = \sum Slope^{-1}_{ihi,ilo,ix,iy} \cdot Dist_{ihi,ilo,ix,iy} \tag{3B}$$

where the '*C*' in *SR_C* refers to stranding covariate effects. This model predicts p_{strand} in equation 1a and assumes $p_{use} = 1$ for all cells. Thus, two cells with the same slope and distance from their locations at the high flow to the first cells along their cross sections that are wetted at the low flow, will have the same stranding risk, even if one cell has more suitable conditions for age-0 trout at the high flow. This model recognizes that factors other than depth and velocity, such as substrate complexity and proximity to spawning habitats, also influence differences in densities of age-0 trout among cells at the TMF high-flow. The units of model *SR_C* are meters·per degree (degree^{-1}). Conceptually, if all slopes in Glen Canyon were 1 degree, the *SR_C* values represent the sum of the distances between each cell at the high flow to the wetted edge at the low flow. This model assumes slope and distance have equal effects and that the effects are multiplicative. Model *SR_C* was not formulated as an additive model (for example, $Slope^{-1} + Dist$) because slope and distance effect sizes (the *β*'s in eq. 1*B*) have not been determined. The model does not include a bed complexity term because the existing substrate map for Glen Canyon was not available to integrate in this analysis.

Our most complex stranding model (*SR_HC*) accounts for both the probability that age-0 trout will use a cell at the TMF high-flow(p_{use}), and the probability of being stranded when flows are dropped (p_{strand}), and is calculated using,

$$SR\_HC_{ihi,ilo} = \frac{\sum HSI_{ihi,ix,iy} \cdot Slope^{-1}_{ihi,ilo,ix,iy} \cdot Dist_{ihi,ilo,ix,iy} \middle| D_{ilo,ix,iy} = 0}{\sum HSI_{ihi,ix,iy}} \tag{3C}$$

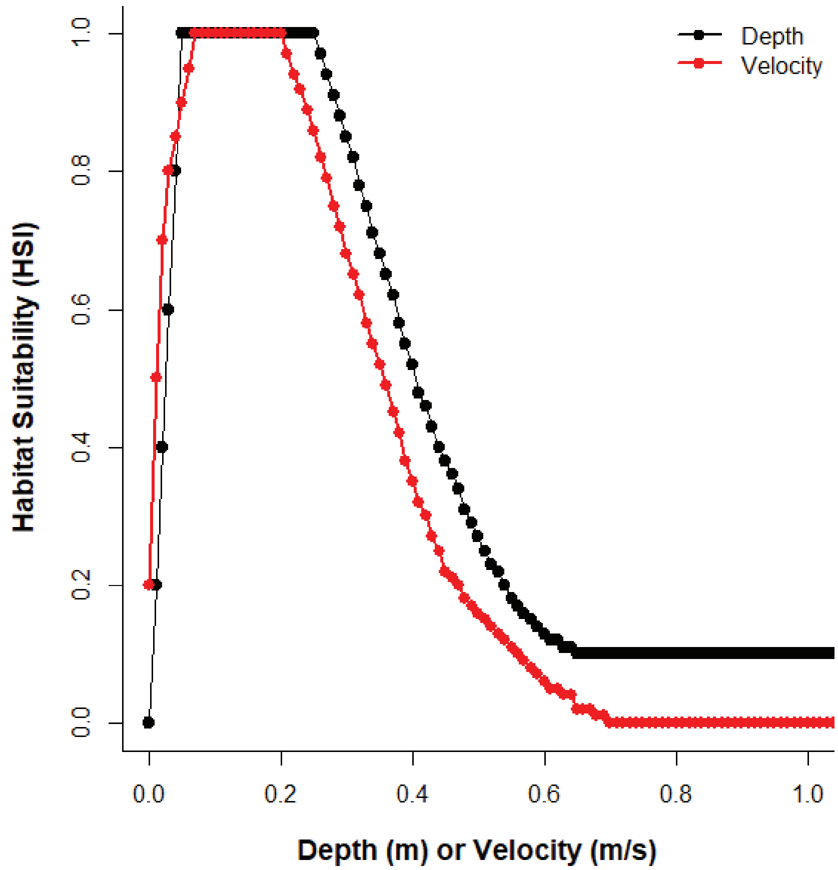

Figure 13. Plot showing the effect of predicted depth and velocity on habitat suitability (HSI) for age-0 rainbow trout. Predicted depth and velocity in each 25-square-meter (m^2) cell from the two-dimensional steady flow model are used as inputs to these relationships to predict habitat suitability for a range of high-flow levels of Trout Management Flows. m/s, meters per second.

where the numerator represents the overall stranding risk for cells that are exposed between TMF high and low flows, and the denominator is the total habitat suitability across exposed and permanently wetted cells. The total HSI will vary with the magnitude of the high-flow, and the denominator ensures this effect will not influence differences in stranding risk predictions across different high-flow values.

Flow Modeling

We used a 1D unsteady flow model (Wiele and Smith, 1996) to predict the shape of TMF hydrographs at various distances downstream from Glen Canyon Dam. Hydrographs were predicted for a location just downstream of the Little Colorado River (river mile [RM] 62 near Crash Canyon) and at Pumpkin Springs (RM 211), which are two mainstem areas used extensively by humpback chub. The objective of the analysis was to evaluate the effects of the duration of the TMF low-flow period and the magnitude of flows that immediately follow the low-flow period (the iv finishing flows in fig. 11), have on the down-ramp rate and minimum flow at downstream locations.

Results

Stranding risk will depend in part on changes in the wetted area between the high and low flows of a TMF. We therefore generated a map of the area that is wetted by a range of flows (fig. 14) by first plotting the wetted surface at the highest flow (30,000 ft^3/s), and then sequentially adding predicted wetted surfaces from lower flows (for example, 28,000 ft^3/s, 26,000 ft^3/s, 24,000 ft^3/s) as additional layers. As a result of this process, the different colors show the incremental decrease in the wetted surface area over decreasing TMF high flows. Colors with wider bands represent flatter areas where there would be

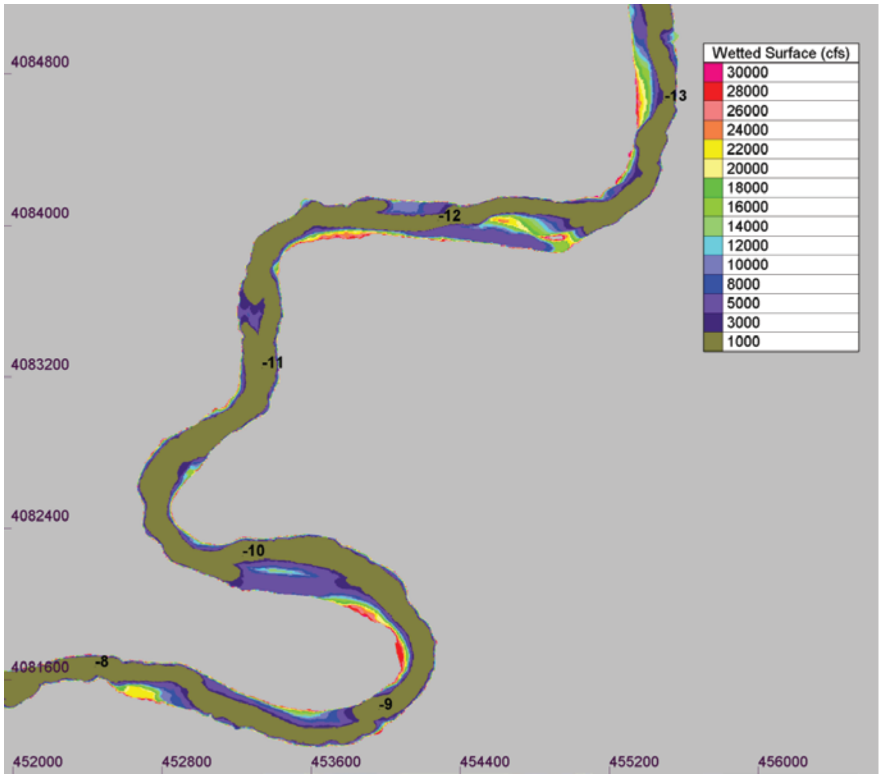

Figure 14. Map showing the predicted areas inundated (wetted) under a range of flows in Glen Canyon between approximately 8 and 13 miles upstream of Lees Ferry, Arizona (shown as negative river miles on map). cfs, cubic foot per second.

a greater decrease in the wetted surface between a flow and the next lower flow. These flat areas are typically located on inside bends (approximately 9.5 miles upstream from Lees Ferry, Arizona [RM −9.5]) but are also associated with other features such as the large cobble bars that form Prop Bar and the Slough at river mile −12. All else being equal, stranding risk will be higher at these wide and flat locations for models that include effects of stranding covariates like slope and distance (eqs. 3*B* and 3*C*).

The area of cells that is wet at each modeled flow (fig. 14) was summed to define the hypsometric relationship between flow and wetted area (fig. 15) for areas between −13 RM and −8 RM. The slope of these relationships decreases as flow increases, because the riverbank steepens at higher elevations, resulting in a smaller increase of wetted area with increasing flow (fig. 15*A*). There is variation in the flow-wetted area curves among reaches within Glen Canyon, such as within the Trout Recruitment and Growth Dynamics project (TRGD) reaches 1A, 1B, and 1C (1A is RM −14.3 to −12.6, 1B is RM −10.2 to −8.4, and 1C is RM −4.3 to −2.7 within our study range) (fig. 15*B*). For example, wetted area is considerably greater in the TRGD reach 1B at flows of approximately 15,000 ft^3/s and higher, relative to reaches 1A and 1C, because 1B is considerably shallower and wider. This reach also shows a steep increase in wetted area as flows increase from approximately 5 to 10,000 ft^3/s. This indicates that there would likely be a higher increase in stranding risk at a 5,000 ft^3/s low flow in the TMF cycle compared to an 8,000 ft^3/s low flow.

Stranding rates of juvenile salmonids have been shown to be higher when slope angles are about 4° (equivalent to 7-percent grade) or less (see “Literature Review” section). We therefore stratified the flow-wetted area relationships by slope angle (fig. 16). In Glen Canyon, about 50 percent of the wetted surface inundated at 15,000 ft^3/s and higher covers bathymetry with cross-sectional slopes of less than or equal to (≤) 4°, and about one-third covers bathymetry with slopes ≤2°. The proportion of wetted area by slope category varied by TRGD reach, with the highest proportions in the ≤2° and ≤4° categories in reach 1B.

Our calculations of habitat suitability at high flows depends on depth and water velocity predictions from the 2D model. There is more area with shallower water depths,

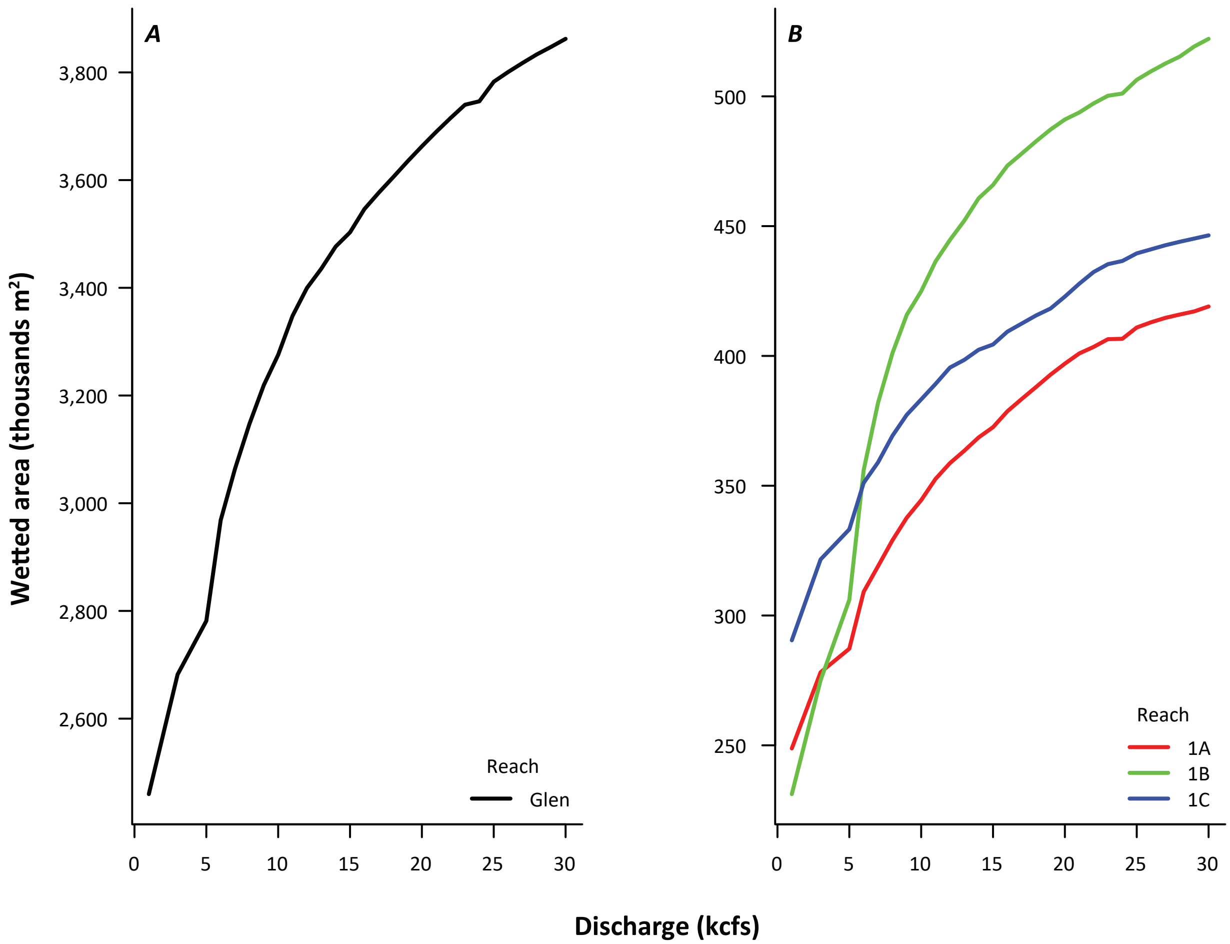

Figure 15. Plot showing relationship between discharge and wetted area for all of Glen Canyon, Arizona (*A*) and for three Trout Recruitment and Growth Dynamics project (TRGD) reaches (*B*). TRGD reaches: 1A (–14.3 to –12.6 miles), 1B (–10.2 to –8.4 miles), and 1C (–4.3 to –2.7 miles). kcfs, thousand cubic foot per second; m^2, square meter.

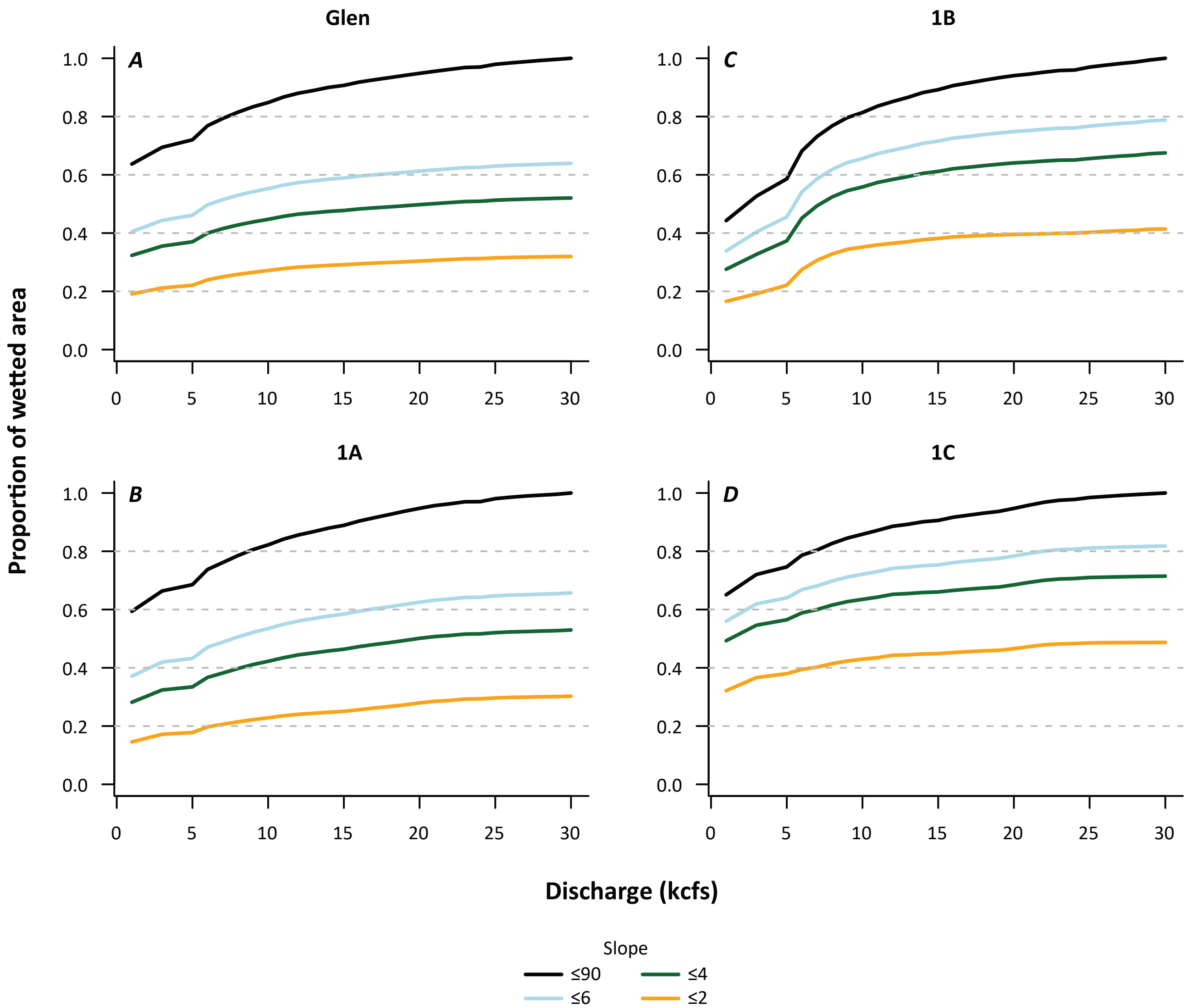

Figure 16. Plots showing the proportion of total wetted area across a range of discharges relative to the total wetted area at a flow of 30,000 cubic feet per second, categorized by slope for all of Glen Canyon (*A*) and for each Trout Recruitment and Growth Dynamics reach (*B*–*D*). Slope (in degrees) is based on topography and calculated perpindicular to the centerline, and therefore reflects the angle of the bank in a direction facing the river. See figure 15 for additional details. kcfs, thousand cubic foot per second.

which is more suitable for age-0 rainbow trout (fig. 13), on inside bends and on bars near channel restrictions (fig. 17). Water velocity also tends to be slower in these shallow areas, and highest in the thalweg on outside river bends and at constrictions caused by gravel bars and debris fans (fig. 18). When flows are high, cells with high habitat suitability for age-0 rainbow trout are limited to a very narrow band on the edge of the river (fig. 19*A*). At lower flows, the band of high suitability near the shoreline widens due to reduced depth and velocity (fig. 19*B*).

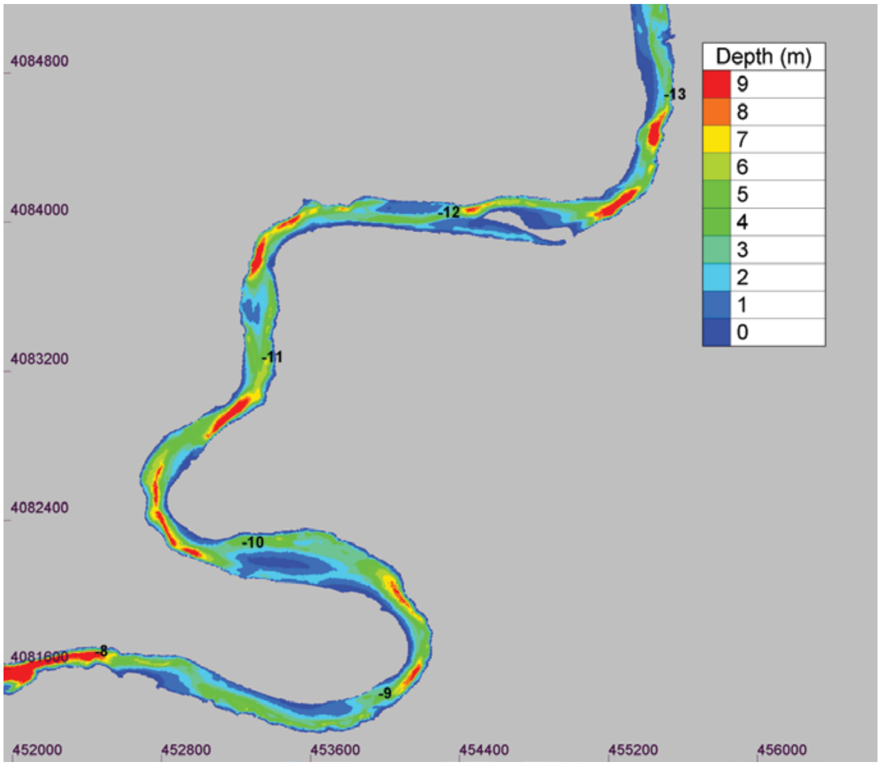

Figure 17. Map showing predicted average water depth at a flow of 20,000 cubic feet per second in Glen Canyon between approximately 8 and 13 miles upstream of Lees Ferry, Arizona (shown as negative river miles on map). Category labels denote the lower limit of the category (that is, 0 denotes depths greater than 0 and less than or equal to 1; 1 denotes depths greater than 1 and less than or equal to 2, and so on). Maps use Universal Transverse Mercator zone 12 south. m, meter.

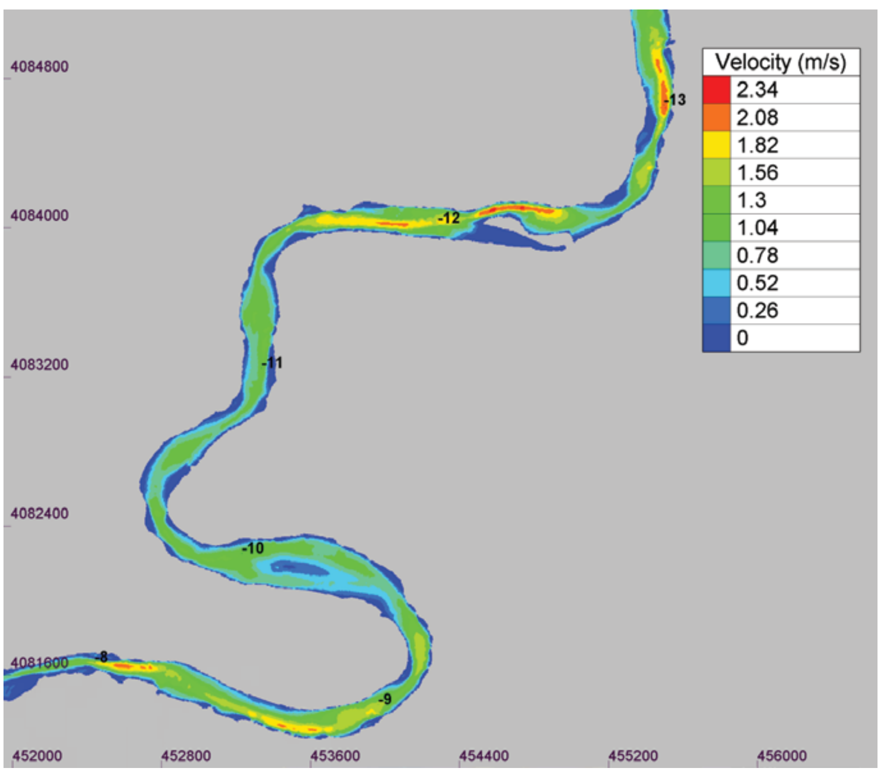

Figure 18. Map showing predicted average water velocity (in meters per second [m/s]) at a discharge rate of 20,000 cubic feet per second in Glen Canyon, Arizona between approximately 8 and 13 miles upstream of Lees Ferry, Arizona (shown as negative miles on map). Maps use Universal Transverse Mercator zone 12 south.

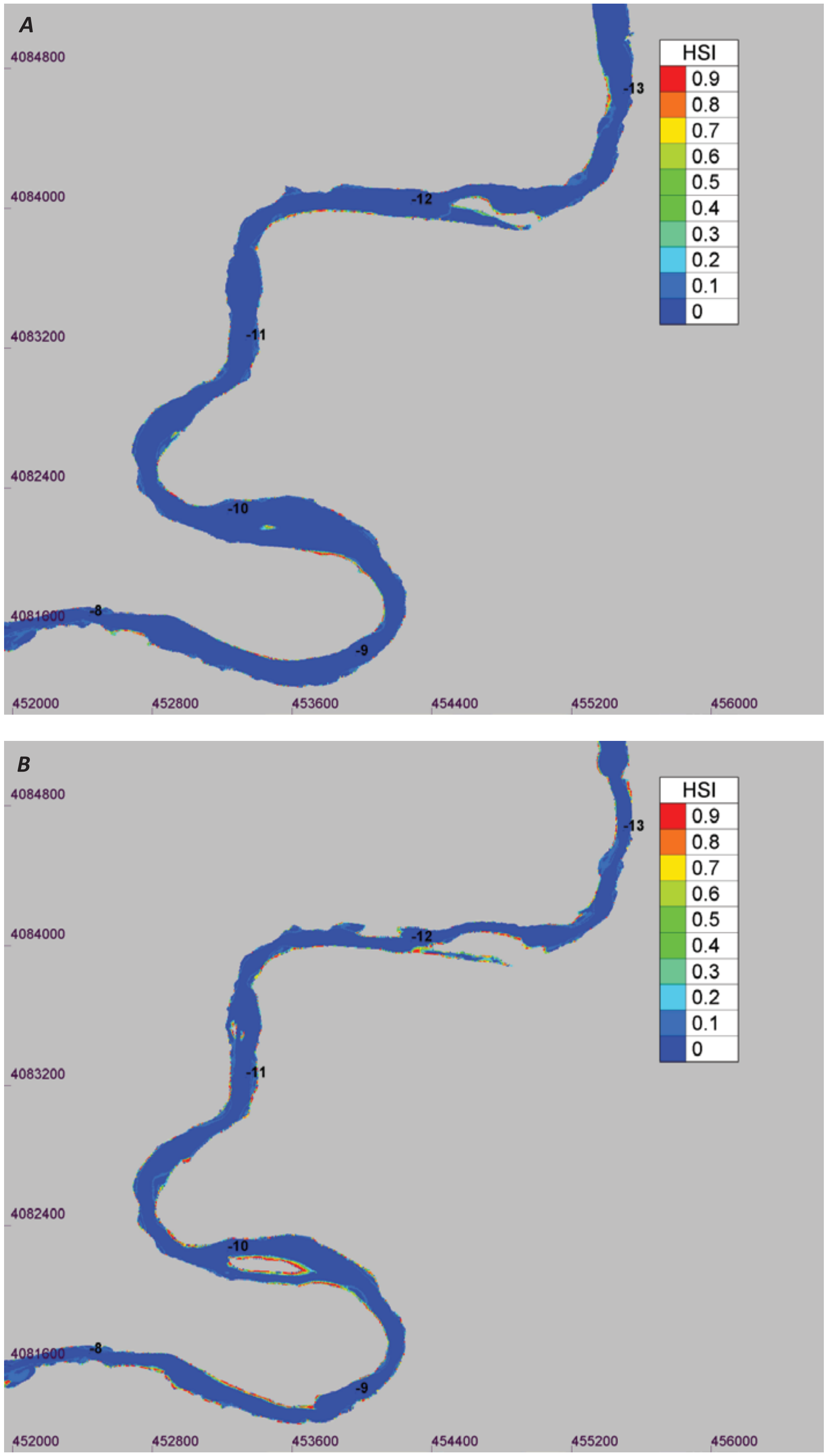

Figure 19. Maps showing predicted habitat suitability (HSI) for age-0 rainbow trout at flow rates of 20,000 cubic feet per second (*A*) and 8,000 cubic feet per second (*B*) in Glen Canyon, Arizona between about 8 and 13 miles upstream of Lees Ferry, Arizona (shown as negative river miles on map). Maps use Universal Transverse Mercator zone 12 south.

To visualize predictions from our stranding model that is based only on habitat suitability (eq. 3*A*), we mapped HSI values at 20,000 ft^3/s and then overlaid the wetted surface at 5,000 ft^3/s (fig. 20). Most of the area that is dewatered between 20,000 and 5,000 ft^3/s has low suitability (blue colors) at the high flow because depths or velocities are too high. Most of the suitable habitat that is dewatered due to the flow reduction comes from cells near the channel margins. Cross-sectional slope of the banks (fig. 21*A*) is included in some of our stranding risk models (eqs. 3b and 3c) but note that only slopes above the minimum flow level of a TMF cycle (that is, greater than 3,000, 5,000, or 8,000 ft^3/s) are used in stranding risk computations (for example, see fig. 21*B*). These maps clearly show the shallow slopes of bars on inside river bends (just upstream and downstream of RM −9), as well as over cobble bars (Prop Bar and the Slough at RM −12), and over sand bars above constrictions (Catchings Bar at RM −13). Models that include stranding covariates will predict greater stranding risk at these locations.

Predictions of stranding risk from models that included stranding risk covariates, and that did (eq. 3*C*, *SR_HC*) or did not (eq. 3*B*, *SR_C*) account for habitat suitability at the high-flow level, were mapped for a 20,000 to 5,000 ft^3/s TMF flow (fig. 11*A*). The *SR_HC* model predicts that stranding risk is greater in cells that have high HSI, and that are farther from the 5,000 ft^3/s wetted edge or where the slope of river bottom to the 5,000 ft^3/s wetted edge is low (fig. 22*A*). Model *SR_HC* predicts lower risk for cells closer to the main channel than the model that does not include HSI effects (*SR_C*) because cells are discounted by low HSI values in the former (fig. 22*B*). Stranding risk, summed across cells from each cross section, shows high variability along the length of Glen Canyon owing to changes in river width and cross-sectional slope (fig. 23). As expected, there is an elevated stranding risk in the wider and shallower TRGD reach 1B compared to reaches 1A and 1C, especially at 3,000 and 5,000 ft^3/s TMF low-flow values.

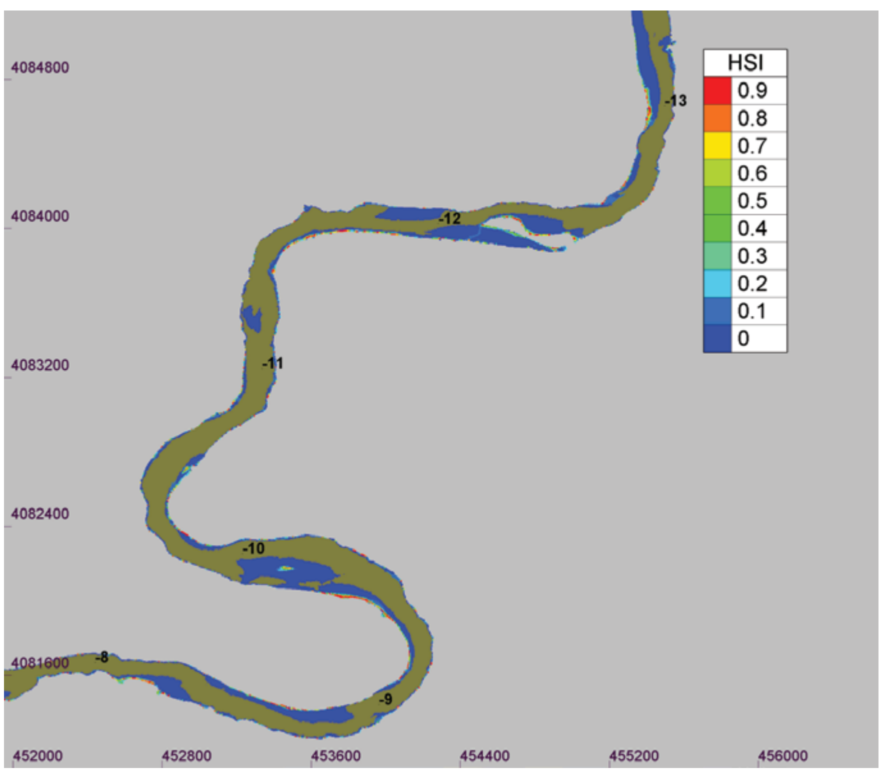

Figure 20. Map showing predicted habitat suitability index values (HSI) for age-0 rainbow trout model at a flow rate of 20,000 cubic feet per second (ft^3/s), and the wetted area at a flow rate of 5,000 ft^3/s (olive-brown color) in Glen Canyon, Arizona between about 8 and 13 miles upstream of Lees Ferry, Arizona (shown as negative miles on map). All colored areas that are not olive-brown will be exposed when flow rates decrease from 20,000 to 5,000 ft^3/s. Map uses Universal Transverse Mercator zone 12S.

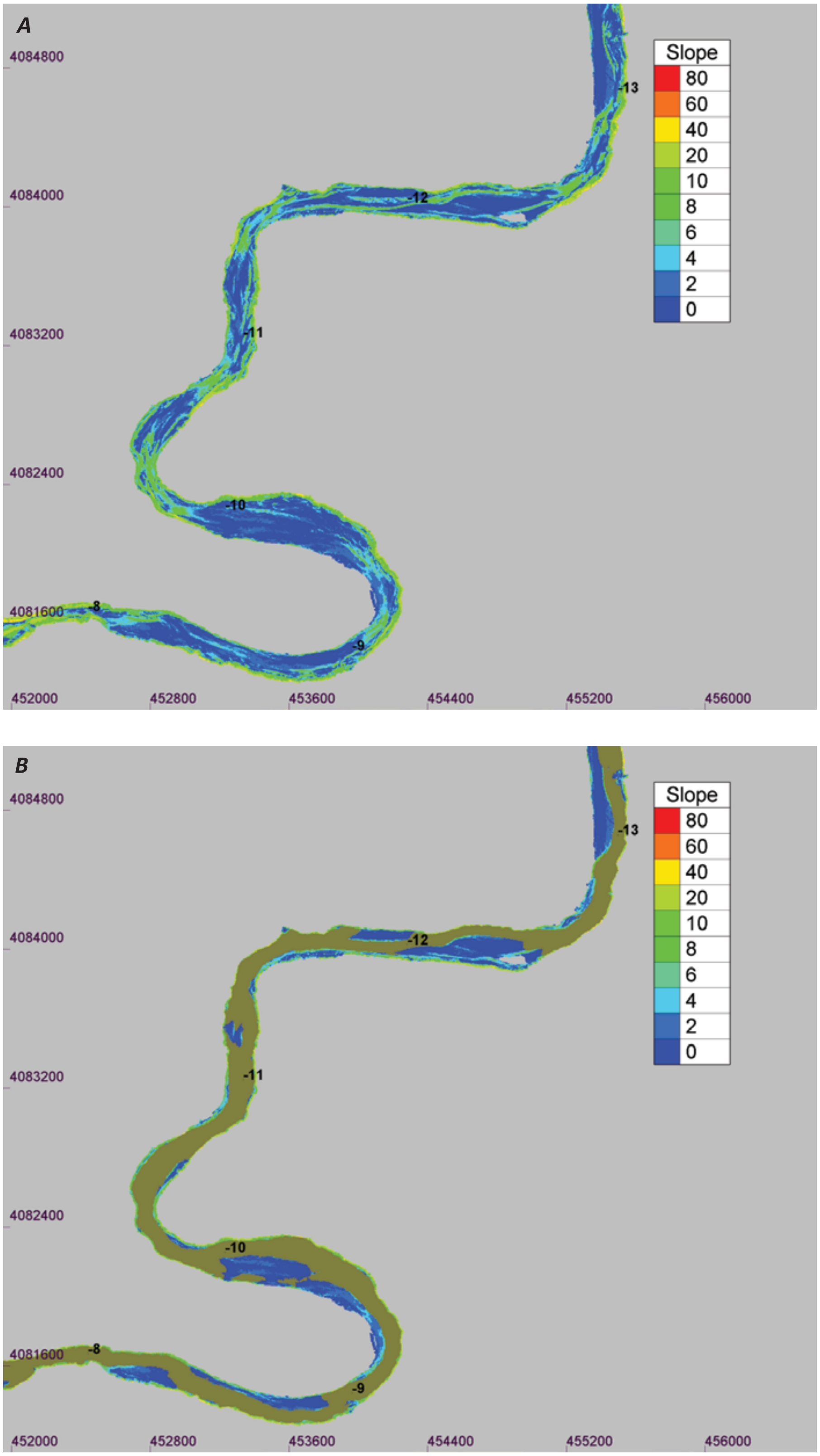

Figure 21. Maps showing shoreline angle (slope, in degrees) in Glen Canyon, Arizona between about 8 and 13 miles upstream of Lees Ferry, Arizona (shown as negative miles on map). The angle of each 25-square-meter cell is calculated based on differences in the elevation of the river bottom among adjacent cells along a cross section (that is, perpendicular to direction of channel). *A*, Map showing the slope of all cells, including those with bed elevations below a flow rate of 5,000 cubic feet per second. *B*, Map showing slope for cells with bed elevation greater than or equal to a flow of 5,000 cubic feet per second. Maps use Universal Transverse Mercator zone 12 south.

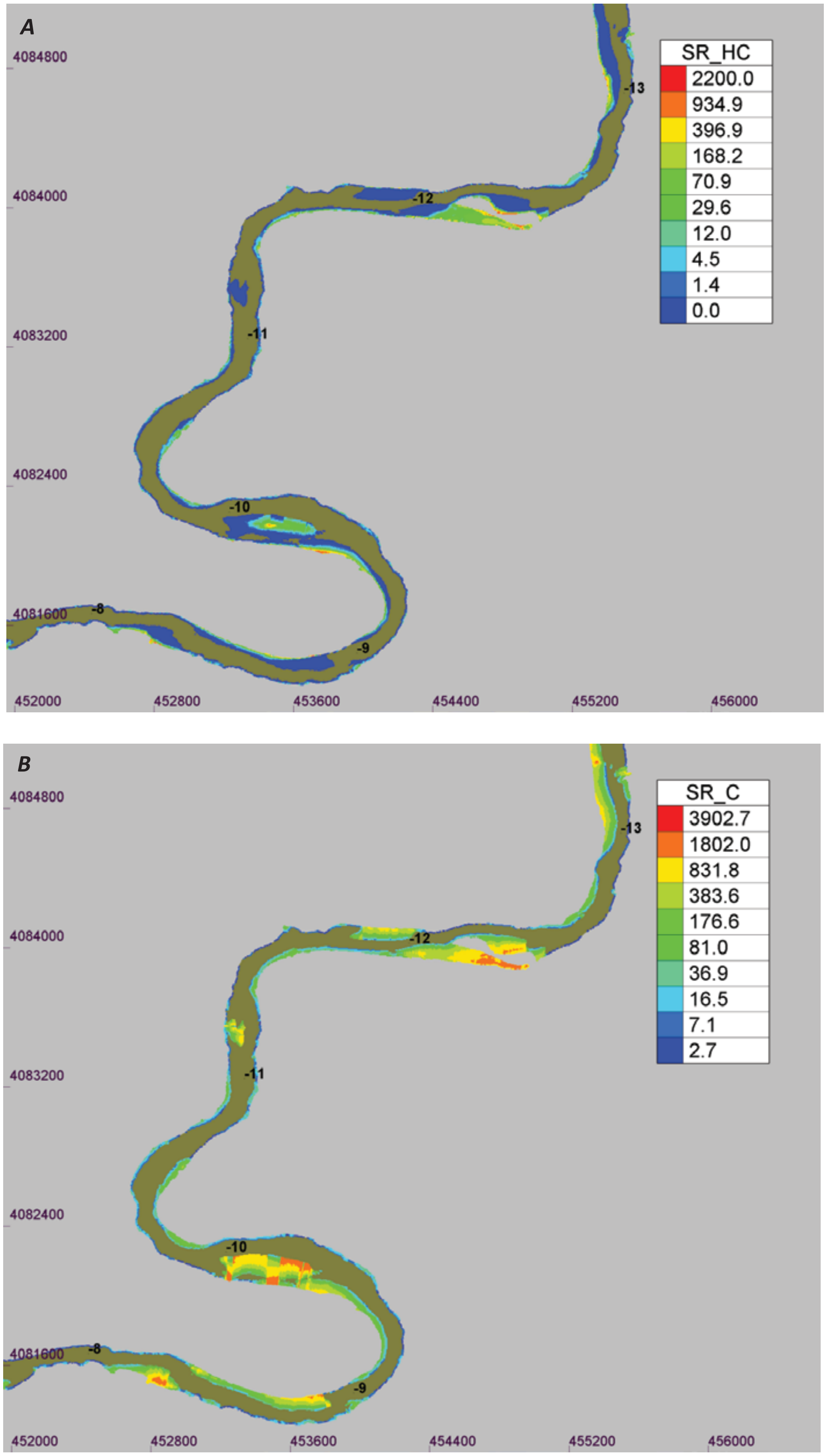

Figure 22. Maps showing predicted stranding risk when flows are dropped from 20,000 cubic feet per second (ft^3/s) to 5,000 ft^3/s based on models that include and do not include habitat suitability at 20,000 ft^3/s in the calculations. *A*, Predicted stranding risk based on model calculations that include habitat suitability for a flow rate of 20,000 ft^3/s (eq. 3*C*). *B*, Predicted stranding risk based on model calculations that do not include habitat suitability for a flow of 20,000 ft^3/s (eq. 3*B*). Olive-brown cells show the wetted area for a flow rate of 5,000 ft^3/s. Maps use Universal Transverse Mercator zone 12 south.

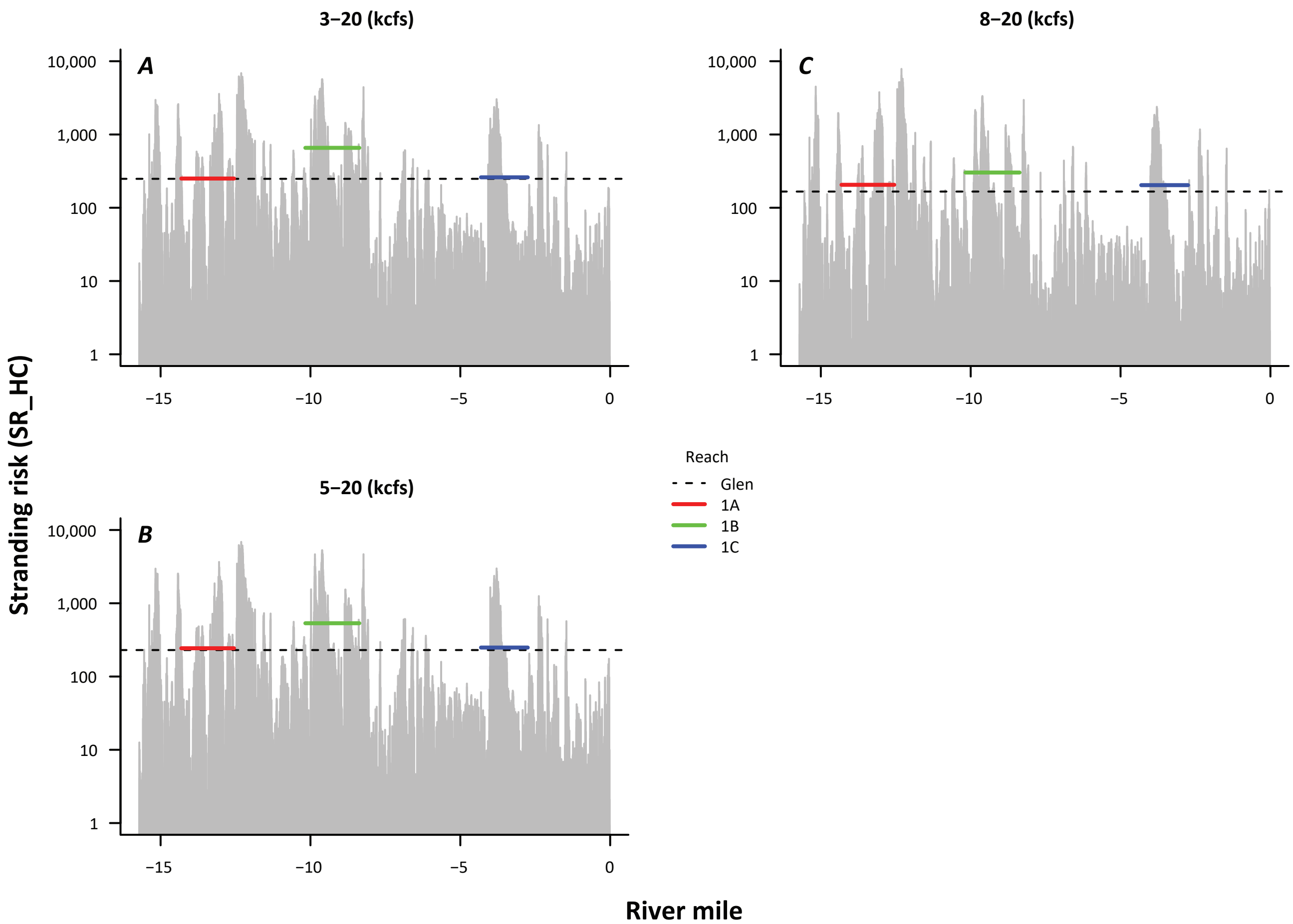

Figure 23. Plots showing predicted stranding risk (eq. 3*C*) between a trout management flow (TMF) high-flow of 20,000 cubic feet per second (ft^3/s) and TMF low-flow of 3,000 (*A*), 5,000 (*B*), and 8,000 (*C*) ft^3/s. Predictions are shown for all 5,001 5-meter-wide cross sections in Glen Canyon, Arizona. Cross sections that occur over wide and flat bars have higher stranding risk owing to lower slope and greater distances between cells with useable habitat at 20,000 cubic feet per second and the wetted edge at the low flow. The dashed line shows the average stranding risk over all cross sections, and the colored solid lines show the averages for each Trout Recruitment and Growth Dynamics project reach. Note y-axis is in logarithmic scale. kcfs, thousand cubic foot per second; SR_HC, the stranding model (eq. 3*C*) that accounts for both the probability that age-0 trout will use a cell at the TMF high-flow and the probability of being stranded when flows are decreased.

Predictions of effects of flow levels on stranding risk varied across stranding risk models (eqs. 3a, 3b, 3c), though the general trends were often similar (fig. 24). Note the scales of stranding risk are not comparable among models, and predictions are only used to assess relative effects of low and high-flow values in the TMF cycle within models. For a given high-flow value, stranding risk increases as the low-flow element of a TMF cycle decreases from 8,000 to 5,000 to 3,000 ft^3/s (fig. 24). The low-flow effect is modest for *SR_H* because suitability is low for cells farther from the shore because of greater depths and velocities. Hence going to a lower flow rate doesn’t have much impact on model predictions. The effect of the low-flow rate of a TMF cycle is greatest for *SR_C* which only includes slope and distance effects because there is no discounting of cells farther from shore during the high-flow element of the TMF. Therefore, the larger the difference between high- and low-flow elements of a TMF cycle, the greater the predicted stranding risk. Not surprisingly then, the stranding risk response to low flows is intermediate for *SR_HC* because it includes habitat, slope, and distance effects. *SR_C* and *SR_HC* indicate there are substantive increases in stranding risk when the low-flow level of a TMF cycle is 3,000 or 5,000 ft^3/s relative to 8,000 ft^3/s. All models show increased stranding risk when the high-flow level of the TMF is increased. The effect of the high-flow level rate is strongest for the *SR_C* model.

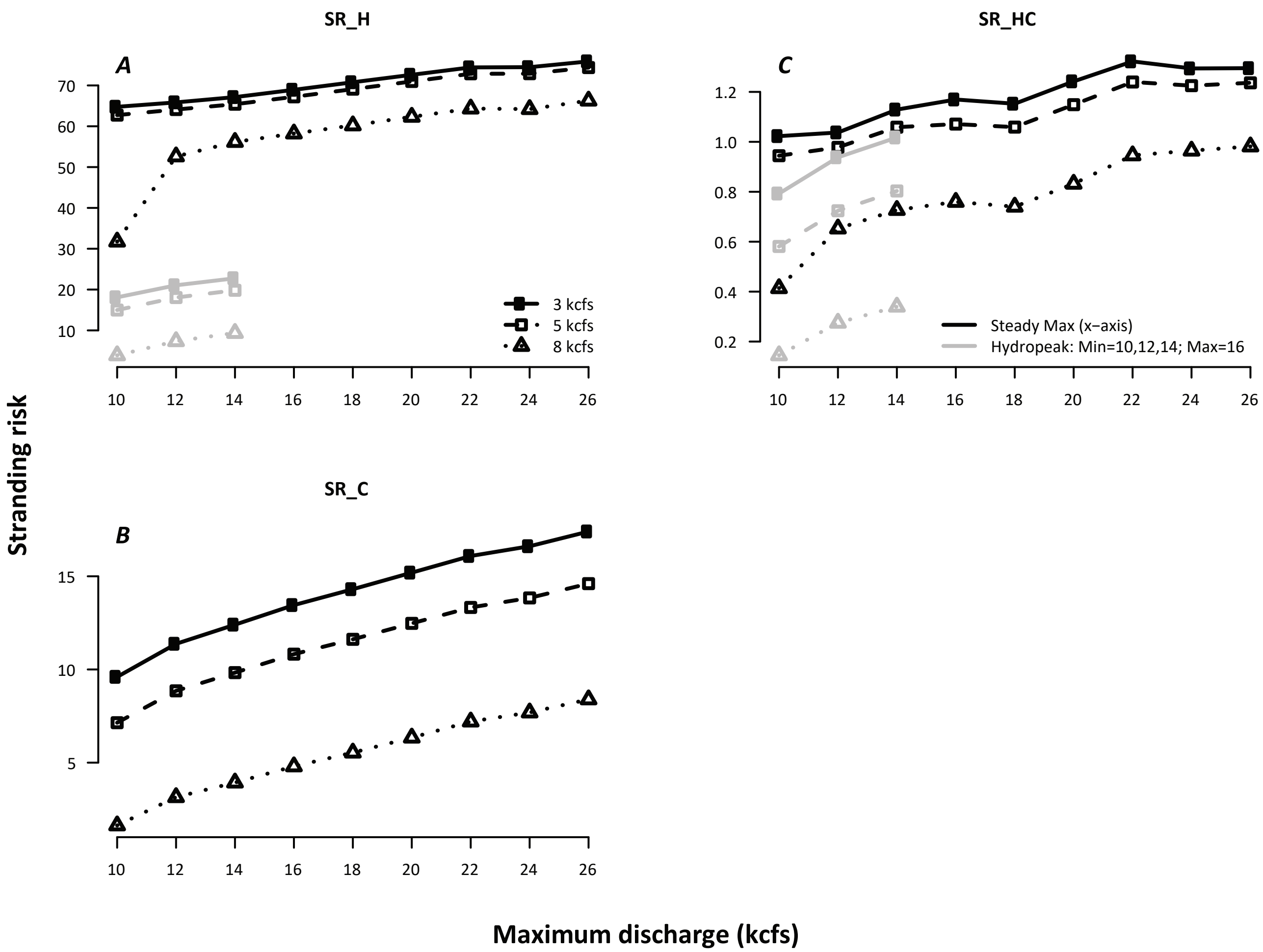

Figure 24. Plots showing predicted stranding risk for age-0 rainbow trout in Glen Canyon, Arizona for Trout Management Flow (TMF) high-flow rates of 10,000 to 26,000 cubic feet per second (ft^3/s) (x-axis value) and TMF low flows of 3,000, 5,000, and 8,000 ft^3/s (lines). *A*, Predicted stranding risk based on models that include only habitat effects (eq. 3*A*; SR_H, values expressed as a percentage of total habitat suitability that is dewatered). *B*, Predicted stranding risk based on models that include only stranding covariate effects (eq. 3*B*; SR_C); y-axis in units of millions of meters per·degree ($m{\cdot}degree^{-1}$). *C*, Predicted stranding risk based on models that include both habitat and stranding covariate effects (eq. 3*C*; SR_HC); y-axis in units of millions of $m{\cdot}degree^{-1}$. Black lines in each plot represent predictions from TMF scenarios with constant high flow prior to the flow decrease (fig. 11*A*, *B*). Grey lines represent predictions for the "go low not high" hydropeaking TMF scenario (fig. 11*C*) where flows within a day fluctuate between minimum values shown on the x-axis (10,000, 12,000, and 14,000 ft^3/s) and a maximum of 16,000 ft^3/s prior to being dropped to a TMF low flow of 3,000, 5,000, or 8,000 ft^3/s. kcfs, thousand cubic foot per second; max, maximum; min, minimum.

Stranding risk based on a "go low not high" TMF strategy with normal hydropeaking operations (fig. 11*C*) was low under the habitat-only model (*SR_H*) but substantial for the more realistic model (*SR_HC*) (fig. 24*C* gray lines). Under this strategy, the x-axis represents the daily minimum flows of 10,000, 12,000, or 14,000 ft^3/s (fig. 24*C*). We assumed age-0 rainbow trout are only using cells at or below these daily minimum flows (fig. 12*A*), and that the daytime high-flow of 16,000 ft^3/s determines habitat suitability at these locations. That is, even though suitability at the minimum-flow may improve at night when flows are lower, the daytime constraints associated with the daily peak flows determine differences in the distribution of fish among cells. Hydropeaking to 16,000 ft^3/s reduces habitat suitability of some lower angle habitats owing to higher depths and velocities, which would reduce stranding effects. This happens because the most suitable habitats under hydropeaking shift to steeper locations where depth and velocity do not change as much, and these habitats are less sensitive to stranding because they have steeper slopes, or shorter distances between the daily minimum flow and the TMF low flow. For the habitat-only model (*SR_H* model), hydropeaking results in a greater reduction in HSI near the shoreline compared to farther from shore, and reduces the proportion of HSI that is exposed during the TMF low flow. However, under the more realistic model that include habitat and other stranding covariates (*SR_HC* model), differences in stranding risk between hydropeaking and steady flow scenarios are smaller, and stranding risk at low flows of 5,000 ft^3/s and especially 3,000 ft^3/s is considerable.

The response to stranding risk under different TMFs varied among TRGD reaches (fig. 25). Stranding risk was considerably higher in 1B at TMF low flows of 3,000 and 5,000 ft^3/s compared to other reaches, owing to its greater width and shallower depth. Stranding risk increased with higher maximum flows in 1A to a greater extent than other reaches owing to the higher elevations of its bars relative to downstream reaches 1B and 1C. Such reach-specific differences need to be considered in relation to locations of spawning and recruitment when designing a TMF. For example, recent TRGD data shows very high abundance of age-0 brown trout in 1A. If this reach is an important part of the Glen Canyon population, and a minimum TMF flow is set at 8,000 ft^3/s, higher maximum TMF flows will be required.

The 1D unsteady model predicts the change in a TMF hydrograph from Glen Canyon Dam at downstream locations owing to wave attenuation. We first predicted downstream

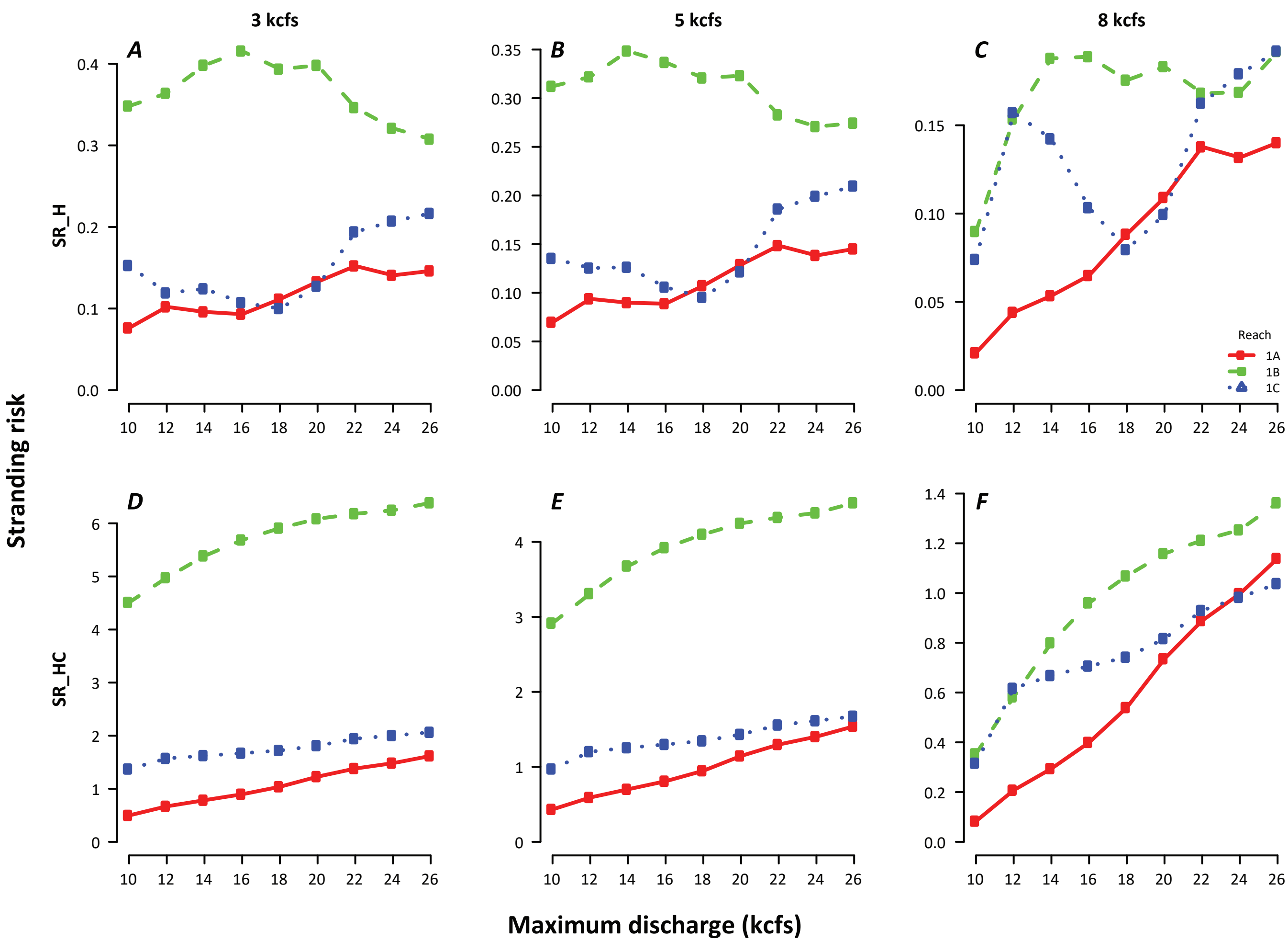

Figure 25. Plots showing predicted stranding risk for the three Trout Recruitment and Growth Dynamics project reaches in Glen Canyon, Arizona at high-flows of 10,000 to 26,000 cubic feet per second (ft^3/s) and low-flow rates of 3,000 (*A*, *D*), 5,000 (*B*, *E*), and 8,000 (*C*, *F*) ft^3/s, for stranding models with only habitat effects (*SR_H*) (*A*–*C*, eq. 3*A*) and models with habitat and stranding covariate effects (*SR_HC*) (*D*–*F*, eq. 3*C*). kcfs, thousand cubic foot per second.

hydrographs using historical flows from a few days in January 2004, which were typical of flows in January and February from 2003 to 2005. These experimental flows were intended to expose rainbow trout spawning redds and ranged from 5,000 to 20,000 ft³/s daily. Both the daily flow range and the ramping rates exceeded values in the 1996 Record of Decision. Predictions show that peak flows of 20,000 ft³/s begin to catch up with the lower flow of 5,000 ft³/s released earlier in the day (fig. 26). This dynamic potentially reduces the down ramp rate and increases the minimum flow at downstream locations. Reduced down-ramp rates were apparent at both the Little Colorado River (RM 61) and Pumpkin Springs (RM 211), but only the latter showed a substantive increase in the minimum flow from 5,000 to about 9,000 ft³/s. We then evaluated simple TMF hydrographs with a 6-hour low-flow period of 5,000 or 8,000 ft³/s, followed by a finishing flow (iv in fig. 11*A*) of 10,000 ft³/s (fig. 27). The travel time for the TMF low flow to reach the Little Colorado River and Pumpkin Springs was about 1 day and about 2.5 days, respectively. These hydrographs still resulted in low flows at downstream locations. However, increasing the TMF finishing flow to 20,000 ft³/s led to substantial increases in the minimum flow at downstream locations, especially when the low-flow period was reduced from 6 to 3 hours (fig. 28).

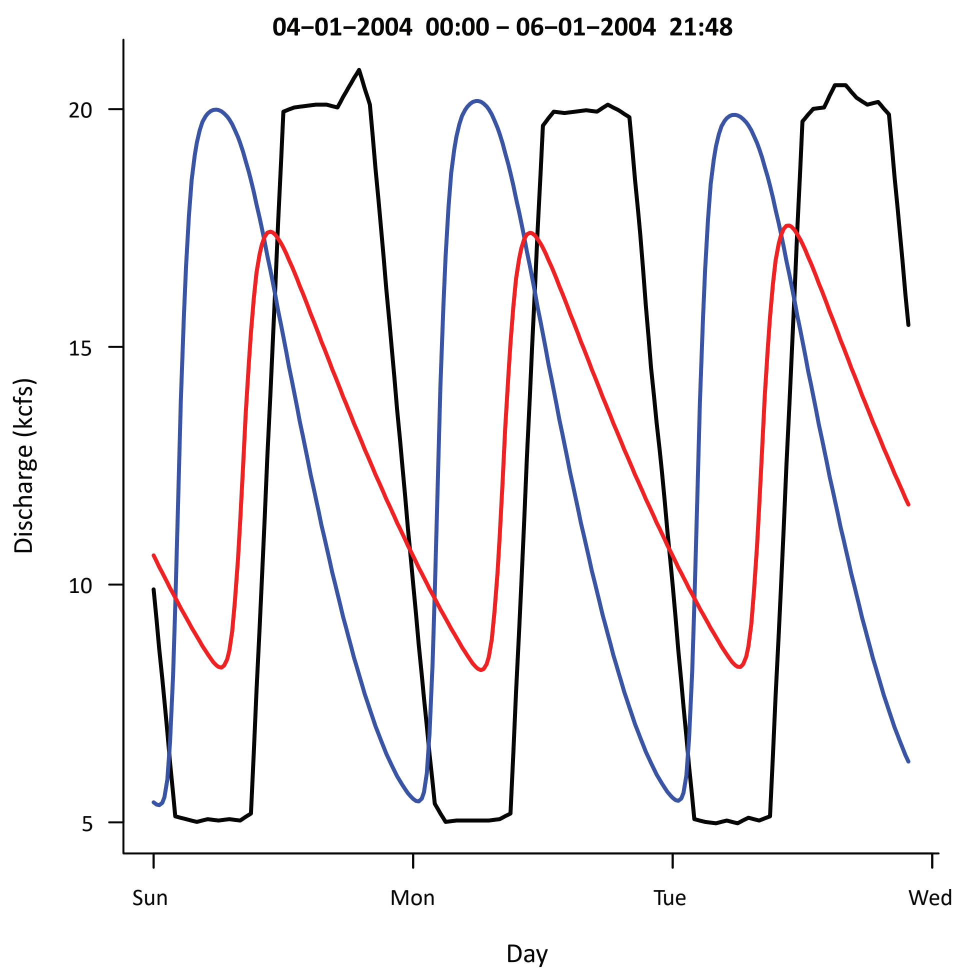

Figure 26. Plot showing discharge from Glen Canyon Dam (black line) in early January 2004 (January 1–6, listed as day-month-year at top of plot) and predicted hydrographs just downstream of the Little Colorado River confluence (Crash Canyon river mile 62, blue line) and near Pumpkin Springs (river mile 211, red line). kcfs, thousand cubic foot per second.

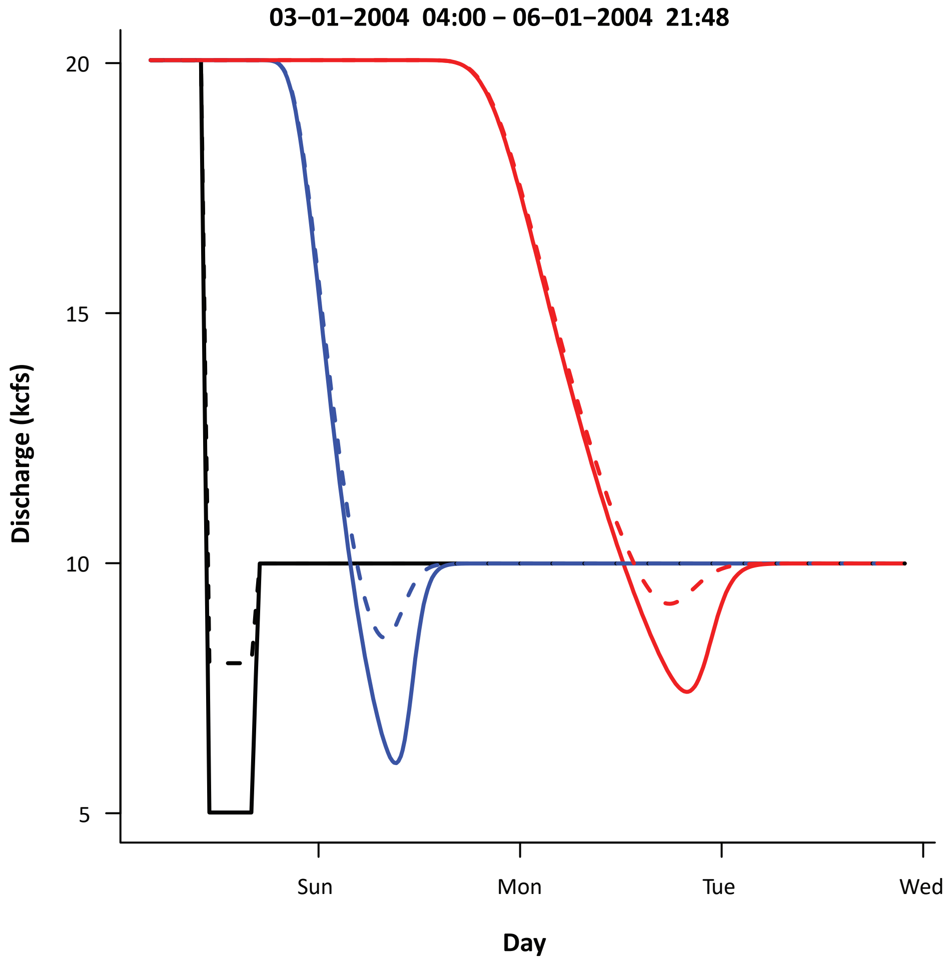

Figure 27. Plot showing two trout management flows (TMFs) from Glen Canyon Dam (black lines) and predicted hydrographs just downstream of the Little Colorado River confluence (Crash Canyon river mile 62, blue lines) and near Pumpkin Springs (river mile 211, red lines). The TMFs consist of holding flows at 20,000 cubic feet per second (ft³/s) and then lowering them within one hour to 5,000 ft³/s (solid lines) or 8,000 ft³/s (dashed lines) for 6 hours, and then bringing them up to 10,000 ft³/s. kcfs, thousand cubic foot per second.

Discussion

There is a considerable area of low-angle shoreline in Glen Canyon where stranding of age-0 rainbow trout and brown trout can occur if the flow changes and timing of TMFs are well designed. Our stranding models predicted that the greater the difference between the high- and low-flow elements of a TMF cycle, the greater the stranding risk. The most realistic model, which included effects of habitat suitability at the TMF high flow, and stranding risk covariates (*SR_HC*), predicted much higher stranding risk at TMF low flows of 3,000 or 5,000 ft³/s, compared to 8,000 ft³/s. The effect of the high-flow element of the TMF cycle was relatively modest, especially when the low-flow element of the TMF cycle was 3,000 or 5,000 ft³/s. Given the potential hydropower and sediment scour costs of the high-flow element of TMFs, our results suggest a TMF high flow of 12,000 to 16,000 ft³/s, combined with a minimum flow of 3,000 or 5,000 ft³/s may represent a good compromise in average or below-average inflow years. This "go low, not high" strategy (fig. 11*C*) under normal hydropeaking operations led to substantive stranding risk when paired with low flows of 5,000 ft³/s, and especially 3,000 ft³/s. This is a potentially attractive strategy given its limited effects on hydropower and sediment resources. Note there are no data relating our predictions of stranding risk to the proportion of the age-0 rainbow or brown trout population that is killed due to a TMF. Thus, there is considerable uncertainty about how effective TMFs will be at regulating trout populations, and effects will have to be empirically determined from field studies, which would include observations of age-0 trout densities immediately before and after a TMF. The stranding risk predictions presented here provide useful information to design a TMF experiment. Conclusions about the effects of TMFs on age-0 abundance will depend on: (1) the proportion of the population of age-0 rainbow trout that are located in habitats that are vulnerable to stranding at the high-flow element of the TMF (p_{use} in eq. 1*A*); (2) the proportion

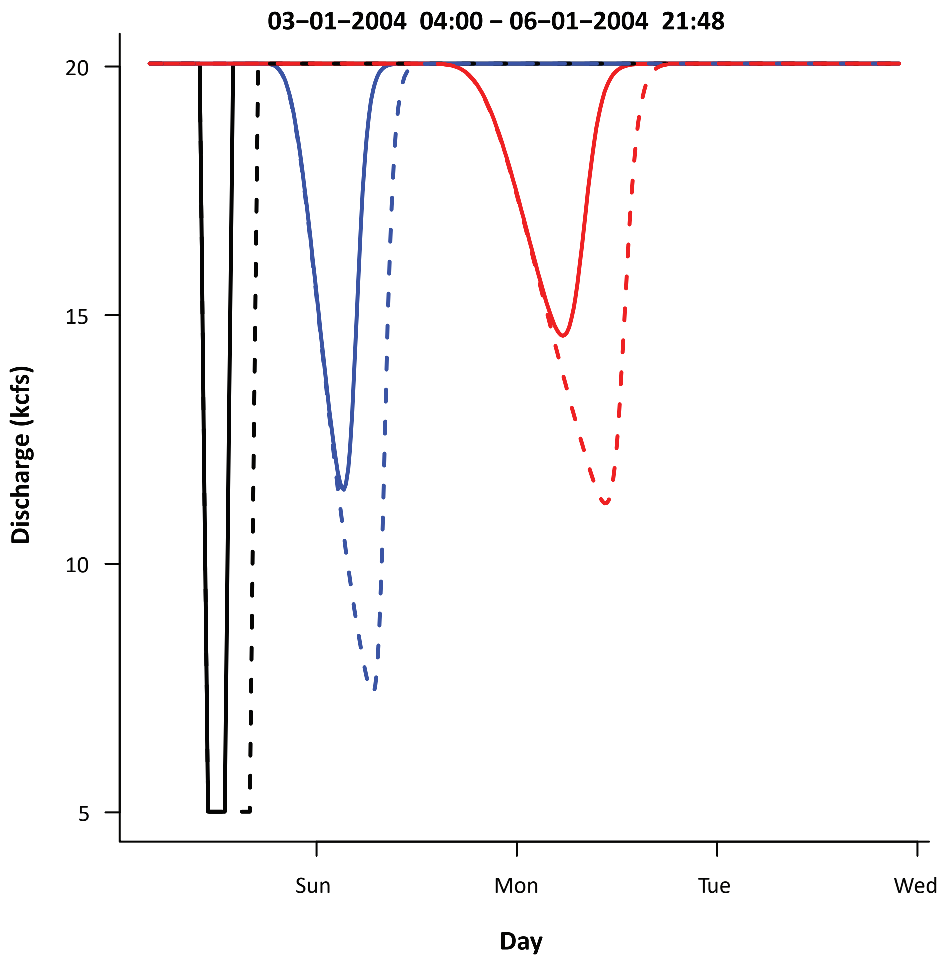

Figure 28. Plot showing two trout management flows (TMFs) from Glen Canyon Dam (black lines) and predicted hydrographs just downstream of the Little Colorado River confluence (Crash Canyon river mile 62, blue lines) and near Pumpkin Springs (river mile 211, red lines). The TMFs consist of holding flows at 20,000 cubic feet per second (ft³/s) and then lowering them (within one hour) to 5,000 ft³/s for 3 (solid lines) or 6 (dashed lines) hours, and then bringing them up to 20,000 ft³/s. kcfs, thousand cubic foot per second.

of the age-0 fish in these vulnerable habitats that will get stranded and die when the flow drop occurs (p_{strand} in eq. 1*A*); and (3) the magnitude of the increase in survival rates of fish that are not stranded, due to lower densities resulting from the TMF (compensatory survival).

The proportion of age-0 rainbow trout using habitats that are vulnerable to stranding can be maximized by selecting dates for TMFs that overlap with the period when age-0 rainbow trout have recently emerged from spawning redds. Age-0 rainbow trout are very small immediately after emergence and therefore require very shallow and low velocity habitat, which is typically located near the waters edge and often over low angle shorelines (Korman and others, 2011b). In Glen Canyon, most age-0 rainbow trout emerge from spawning redds over an extended period between March 1st and July 31st (fig. 4, Korman and others, 2011a). TMFs focused on rainbow trout should therefore occur between the beginning of Mayl to end of June, to affect most of the annual cohort (fig. 5). The proportion of age-0 rainbow trout that utilize low-angle shorelines, that are more vulnerable to stranding, declines from May to December (fig. 7, Korman and others, 2011b), further emphasizing the need to conduct TMFs as early as possible.

The proportion of age-0 rainbow trout using habitat vulnerable to stranding also depends on the length of time that the TMF high-flow element is sustained prior to the flow drop. There is considerable uncertainty about the length of time needed for this colonization process to occur (fig. 12). There are few studies that have attempted to quantify colonization time (see “Duration of Drawdown” section), but the limited information that is available suggests the duration is likely on the scale of weeks rather than the much shorter 3-day period identified in the Glen Canyon LTEMP EIS (Bureau of Reclamation and National Park Service, 2015). Long colonization times are probably the biggest limitation for implementing “go high then low” TMFs in below average and average water years. However, in equalization years, flows will be high and steady for much of the spring and summer (fig. 11*B*). Avery and others (2015) showed extensive colonization of age-0 rainbow trout of low-angle, flooded habitats in Glen Canyon during the 2011 equalization flows. Thus, TMFs have a higher likelihood of reducing age-0 recruitment in equalization years because operating rules already provide a long colonization period. And equalization years are the most likely to produce elevated recruitment rates that may need to be reduced by TMFs. More research on the duration of the high-flow period of TMFs in below-average or average water years could reduce uncertainty in this component of the action. It may be that the two realistic options for TMFs are either the TMF equalization scenario (fig. 11*B*) or the "go low not high" hydropeaking scenario (fig. 11*C*).

The proportion of age-0 rainbow trout in vulnerable habitats that will get stranded when flows are lowered during a TMF will be controlled by factors including fish size and characteristics of the habitat and the hydrograph. For rainbow trout, conducting TMFs in May through June ensures that most age-0 trout will still be very small, making them more vulnerable to stranding when flows are dropped. Our modelling indicates that stranding risk is greater when flows are lowered to 3,000 or 5,000 ft^3/s compared to 8,000 ft^3/s. A more modest increase in stranding risk might be achieved by increasing the high-flow level in the TMF cycle. Both of these hydrograph changes could increase the extent of exposed habitat and the distance between each exposed cell to the wetted edge at the low flow. Our models did not include effects of substrate complexity on stranding probability when flows are lowered. This is a logical extension, but without field data quantifying the increase in stranding with complexity, predictions will be uncertain. That said, there is extensive cover of algae and macrophytes over the bed at elevations inundated by 5,000 ft^3/s. TMFs with a low-flow element that is just below 5,000 ft^3/s (for example, 3,000 or 4,000 ft^3/s) will likely result in greater stranding risk than shown in our analysis.

Following a TMF, age-0 rainbow trout that are not stranded may have higher survival rates owing to lower densities that were caused by the TMF. The extent of this compensatory survival response is uncertain but appears to be high for age-0 rainbow trout in Glen Canyon. Korman and others (2011a) did not see any effect from extensive dewatering of redds (25–50 percent of all redds) on later age-0 abundance relative to years with normal operations. However, the extent of compensatory survival rates associated with TMFs discussed here will likely be less because most of the compensatory survival response for juvenile salmonids is thought to occur during or very shortly after emergence. (Elliott, 1994; Einum and Nislow, 2005; Lobón-Cerviá, 2007). In addition, repeated TMFs in a season can potentially counteract compensatory survival responses. That is, if age-0 rainbow trout that survived an initial TMF in early May and then survive at a higher rate because of reduced population densities, then they could still be potentially affected by additional TMFs conducted in late May and June. Sequential TMFs are also needed given that emergence of rainbow trout in Glen Canyon occurs over a relatively long period (fig. 4). Thus, repeated TMFs within a season have the double benefit of maximizing the probability of negatively effecting the entire year class of age-0 rainbow trout (that is, fish emerging early and late over the protracted emergence period) and reducing the survival rates for fish that were not stranded in earlier TMFs. Ultimately, the age-0 trout compensatory responses to TMFs will have to be evaluated using field data. This could be accomplished over many years by comparing estimated age-0 abundance in years with and without TMFs using data from the TRGD project. Alternatively, age-0 abundance estimates taken immediately before and after TMFs can be compared, but this would require focused age-0 surveys.

Our findings have relevance for TMFs focused on controlling brown trout in Glen Canyon, and concerns about negative effects of TMFs on native fish and other resources in Grand Canyon. Although detailed studies of age-0 brown trout in Glen Canyon are not available, information on spawn timing from the catch of ripe fish, combined with water temperature and thermal incubation requirements, indicates that emergence likely occurs between January and April, with a peak in early to mid-March (fig. 6). Trout management flows have the potential to strand small native fish in Grand Canyon or dewater shorelines colonized by algae and aquatic insects (Kennedy and others, 2016). However, unsteady flow modelling showed that fast down-ramp rates and low minimum flows can be mitigated by shortening the duration of the low-flow element of the TMF and following it with higher flows for a limited period. This operation would also likely minimize unwanted negative effects on algae and aquatic insects in Glen Canyon if the low-flow element is conducted at night.

The decision to implement a TMF to reduce rainbow or brown trout recruitment in Glen Canyon will be difficult because the strength of recruitment in a particular year is only established after it is too late to implement the TMFs. Age-0 rainbow trout become vulnerable to boat electrofishing in June, which is about two months after the start of the May through July TMF window. Thus, in the absence of early season sampling using backpack electrofishing (Korman and others, 2011b), pre-season recruitment forecasts could be used to determine whether to implement a TMF in a given year. Implementing TMFs in equalization years is a relatively easy decision, as we would be more confident that rainbow trout recruitment will be substantive (Avery and others, 2015). Extended steady and high flows during equalization will lead to more colonization of habitats that are vulnerable to stranding. In addition, the negative effects of high and steady releases on hydropower and sediment retention in Grand Canyon are already being incurred due to the equalization operation, so there should be no additional costs associated with TMFs under these circumstances. The two-month delay between emergence of rainbow trout and the determination of the strength of annual recruitment based on field data also applies to brown trout. However, given the recent population trend and negative consequences of a large population of brown trout, it is perhaps prudent to assume that recruitment in most years will be sufficient to support an expanding population.

Owing to uncertainties about the utility of TMFs for limiting trout recruitment in Glen Canyon, testing and evaluation of TMFs via field studies could provide valuable learning. However, the need to conduct TMFs to control rainbow trout recruitment in the future will likely be infrequent. Age-0 rainbow trout recruitment levels in Glen Canyon from 2012 to 2020 were not large enough to cause increases in abundance of rainbow trout in the Little Colorado River, or a population collapse in Glen Canyon. The collapse in 2015 was in part the result of a large recruitment event in 2011, likely due to equalization flows and elevated phosphorus (Korman and others 2016). Thus, future opportunities to test the effects of TMFs on age-0 rainbow trout in the field will likely be limited. Trout management flows in average or even below-average recruitment years could be implemented for testing purposes, but this would likely have negative effects on the fishery given the current low abundance of rainbow trout in Glen Canyon. In addition, if rainbow and brown trout are competing, reducing rainbow trout recruitment could lead to further increases in the abundance of brown trout. Thus, a logical alternative is to focus TMF testing on brown trout in Glen Canyon. Trout management flows conducted between February and April could be evaluated based on the abundance of brown trout recruits from TRGD sampling in September and November. This effort would need to be repeated over multiple years, with and without TMFs, to estimate the strength of a TMF effect. Inferences could be strengthened by comparing differences in recruitment of brown trout among reaches with different stranding risks. Stranding risk in TRGD reach 1B is considerably higher than in other reaches, so differences in its recruitment levels with the other two TRGD reaches in TMF and non-TMF years would provide useful contrasts to estimate the TMF effect. Unfortunately, sampling in 1B is no longer conducted due to funding limitations. The effects of these decisions on the ability to evaluate actions identified in the LTEMP EIS, such as TMFs, needs to be carefully considered.

Trout Management Flows Implementation and Considerations

The Colorado River downstream of Glen Canyon Dam is managed with the goal to meet many different demands, sometimes with opposing objectives. Implementing trout management flows (TMFs) could impact other resources depending on the particular flow strategy and, therefore, must be carefully planned.

The existing literature provides some relevant information on key factors contributing to stranding that can help inform an effective TMF; however, the literature lacks specificity. The hypsometric analysis provides specific details for Glen Canyon. The combination of both sections of this report provides a useful springboard for designing an effective TMF experiment, while identifying remaining uncertainties. Below is a summarized list of key findings and remaining uncertainties.

Key Findings:

- For rainbow trout, TMFs will be more effective if implemented in May through July to coincide with the period when most small (20–50 millimeters) fry for each annual cohort are present and using low-angle shorelines where vulnerability to stranding is greatest. For brown trout, TMFs will be more effective if implemented in February through April based on their emergence timing in Glen Canyon.

- TMFs will be more effective if flows are decreased during the daytime to maximize stranding and potentially increase mortality rates due to higher air temperatures.

- To mitigate potential compensatory survival response, TMFs will be more effective if multiple cycles are repeated within the TMF period. Repeated TMF cycles will also reduce age-0 abundance by spanning more of the emergence period.

- The hypsometric analysis shows that flows below 8,000 cubic feet per second (ft^3/s) (particularly from 3,000 to 5,000 ft^3/s) can have higher stranding based on the geomorphology of Glen Canyon.

- Implementing TMFs depends on the predicted levels of recruitment for a given year. A predictive model is preferable since field-based estimates of recruitment strength occur after the period when small fry are most vulnerable to rapid changes in flow.

- Owing to the increasing brown trout population trends in Glen Canyon, TMFs focusing on reducing brown trout recruitment could be considered.

Remaining Uncertainties:

- The length of time needed for small fish to colonize newly wetted areas inundated during the high flow element of the TMF cycle. Existing stranding experiments in the literature do not show a clear behavior pattern for small fish response to fluctuating flows. This uncertainty only needs to be addressed if a “go high then low” TMF (fig. 11*A*) is being considered. It is less relevant for the equalization (fig. 11*B*) or “go low not high” (fig. 11*C*) TMF strategies.

- The length of time small-bodied fish can survive out of water in different temperature ranges, substrate/vegetation types, and with exposure to direct sunlight. This will inform the duration of drawdown necessary to achieve mortality goals. The shortest duration of drawdown possible would have the least effect on downstream resources.

- The down-ramp rate most likely to achieve TMF objectives. Faster down-ramp rates can lead to more stranding; however, specific down-ramp rates to optimize extent of stranding in Glen Canyon remain unknown.

- The number of repetitions of TMF cycles to mitigate compensatory response. It is more effective to repeat TMF cycles; however, it is unknown how many cycles must be repeated and how much time between cycles is ideal.

- The specific age-0 recruitment threshold for rainbow trout at which TMFs could be implemented to limit the number of fish emigrating into Marble Canyon. This can be expanded to include specific thresholds for brown trout.

In conclusion, there remains several key uncertainties associated with implementing TMFs from Glen Canyon Dam. While some uncertainties can be resolved through mesocosm experiments (such as fish mortality rate from air exposure at different temperatures and different substrates and vegetation types, and so forth), conducting TMFs under an experimental framework is essential to determine if they are an effective method of controlling abundance of rainbow trout and brown trout in the Colorado River.

References Cited

Almeida, R., Boavida, I., and Pinheiro, A., 2014, Assessment of trout habitat suitability fish shelters downstream a hydropower dam: 10th International Symposium on Ecohydraulics 2014, Trondheim, Norway, accessed April 5, 2020, at https://www.researchgate.net/publication/264010220_Assessment_of_trout_habitat_suitability_fish_shelters_downstream_a_hydropower_dam.

Anglin, D.R., Haeseker, S., Skalicky, J., Schaller, H., Tiffan, K., Hatten, J., Rondorf, D., Hoffarth, P., Nugent, J., Benner, D., and Yoshinaka, M., 2006, Effects of hydropower operations on spawning habitat, rearing habitat, and stranding/entrapment mortality of fall Chinook salmon in the Hanford Reach of the Columbia River: U.S. Fish and Wildlife Service, 260 p., accessed April 4, 2020, at https://pubs.er.usgs.gov/publication/70198141.

Angradi, T.R., Clarkson, R.W., Kinsolving, D.A., Kubly, D.M., and Morgensen, S.A., 1992, Glen Canyon Dam and the Colorado River—Response of aquatic biota to dam operations: Flagstaff, Ariz., U.S. Department of Interior, Bureau of Reclamation, Upper Colorado Region, Glen Canyon Environmental Studies, prepared by Arizona Game and Fish Departrnent, Phoenix, Ariz., under cooperative agreement no. 9-FC40-07940, 155 p.

Armstrong, J.D., Braithwaite, V.A., and Fox, M., 1998, The response of wild Atlantic salmon parr to acute reductions in water flow: Journal of Animal Ecology, v. 67, no. 2, p. 292–297, accessed May 4, 2020, at https://doi.org/10.1046/j.1365-2656.1998.00193.x.

Auer, S., Zeiringer, B., Fuhrer, S., Tonolla, D., and Schmutz, S., 2017, Effects of river bank heterogeneity and time of day on drift and stranding of juvenile European grayling (*Thymallus thymallus* L.) caused by hydropeaking: Science of the Total Environment, v. 575, p. 1515–1521, accessed March 30, 2020, at https://doi.org/10.1016/j.scitotenv.2016.10.029.

Avery, L.A., Korman, J., and Persons, W.R., 2015, Effects of increased discharge on spawning and age-0 recruitment of rainbow trout in the Colorado River at Lees Ferry, Arizona: North American Journal of Fisheries Management, v. 35, no. 4, p. 671–680, accessed January 9, 2022, at https://doi.org/10.1080/02755947.2015.1040560.

Bair, L.S., Yackulic, C.B., Springborn, M.R., Reimer, M.N., Bond, C.A., and Coggins, L.G., 2018, Identifying cost-effective invasive species control to enhance endangered species populations in the Grand Canyon, USA: Biological Conservation, v. 220, p. 12–20, accessed accessed January 9, 2022, at https://doi.org/10.1016/j.biocon.2018.01.032.

Bauersfeld, K., 1978, Stranding of juvenile salmon by flow reductions at Mayfield Dam on the Cowlitz River, 1976: Olympia, Wash., Washington Department of Fisheries, Tech. Rep. 36, 36 p.

Becker, C.D., Neitzel, D.A., and Carlile, D.W., 1986, Survival data for dewatered rainbow trout (*Salmo gairdneri* Rich.) eggs and alevins: Journal of Applied Ichthyology, v. 2, no. 3, p. 102–110, accessed May 5, 2020, at https://doi.org/10.1111/j.1439-0426.1986.tb00436.x.

Bell, E., Kramer, S., Zajanc, D., and Aspittle, J., 2008, Salmonid fry stranding mortality associated with daily water level fluctuations in Trail Bridge reservoir, Oregon: North American Journal of Fisheries Management, v. 28, no. 5, p. 1515–1528, accessed May 20, 2020, at https://doi.org/10.1577/M07-026.1.

Berland, G., Nickelsen, T., Heggenes, J., Okland, F., Thorstad, E.B., and Halleraker, J., 2004, Movements of wild atlantic salmon parr in relation to peaking flows below a hydropower station: River Research and Applications, v. 20, no. 8, p. 957–966, accessed May 15, 2020, at https://doi.org/10.1002/rra.802.

Boavida, I., Santos, J.M., Ferreira, T., and Pinheiro, A., 2015, Barbel habitat alterations due to hydropeaking: Journal of Hydro-environment Research, v. 9, no. 2, p. 237–247, accessed May 15, 2020, at https://doi.org/10.1016/j.jher.2014.07.009.

Boumis, R.J., Ferry, L.A., Pace, C.M., and Gibb, A.C., 2014, Heads or tails—Do stranded fish (mosquitofish, *Gambusia affinis*) know where they are on a slope and how to return to the water?: PLoS One, v. 9, no. 8, article e104569, 11 p., accessed November 20, 2020, at https://doi.org/10.1371/journal.pone.0104569.

Bovee, K.D., 1986, Development and evaluation of habitat suitability criteria for use in the instream flow incremental methodology: Washington, D.C., U.S. Fish and Wildlife Service Instream Flow Information Paper 21, FWS/OBS-86/7, 235 p., accessed January 9, 2022, at https://pubs.er.usgs.gov/publication/70121265.

Bradford, M.J., 1997, An experimental study of stranding of juvenile salmonids on gravel bars and in sidechannels during rapid flow decreases: Regulated Rivers, v. 13, no. 5, p. 395–401, accessed March 30, 2020, at https://doi.org/10.1002/(SICI)1099-1646(199709/10)13:5<395::AID-RRR464>3.0.CO;2-L.

Bradford, M.J., and Higgins, P.S., 2001, Habitat-, season-, and size-specific variation in diel activity patterns of juvenile chinook salmon (*Oncorhynchus tshawytscha*) and steelhead trout (*Oncorhynchus mykiss*): Canadian Journal of Fisheries and Aquatic Sciences, v. 58, no. 2, p. 365–374, accessed May 3, 2020, at https://doi.org/10.1139/f00-253.

Bradford, M.J., Taylor, G.C., Allan, J.A., and Higgins, P.S., 1995, An experimental study of the stranding of juvenile coho salmon and rainbow trout during rapid flow decreases under winter conditions: North American Journal of Fisheries Management, v. 15, no. 2, p. 473–479, accessed May 2, 2020, at https://doi.org/10.1577/1548-8675(1995)015<0473:AESOTS>2.3.CO;2.

Bureau of Reclamation, and National Park Service, 2015, Glen Canyon—Dam long-term experimental and management plan final environmental impact statement: Bureau of Reclamation and National Park Service, Intermountain Region, [variously paged], accessed January 9, 2022, at https://ltempeis.anl.gov/documents/final-eis/.

Carothers, S.W., and Brown, B.T., 1991, The Colorado River through Grand Canyon—Natural history and human change: Tucson, Ariz., University Press, Inc., 235 p.

Casas-Mulet, R., Saltveit, S.J., and Alfredsen, K., 2015, The survival of Atlantic salmon (*Salmo salar*) eggs during dewatering in a river subjected to hydropeaking: River Research and Applications, v. 31, no. 4, p. 433–446, accessed May 5, 2020, at https://doi.org/10.1002/rra.2827.

Casas-Mulet, R., Saltveit, S.J., and Alfredsen, K., 2016, Hydrological and thermal effects of hydropeaking on early life stages of salmonids—A modelling approach for implementing mitigation strategies: Science of the Total Environment, v. 573, p. 1660–1672, accessed May 4, 2020, at https://doi.org/10.1016/j.scitotenv.2016.09.208.

Cocherell, S.A., Chun, S.N., Cocherell, D.E., Thompson, L.C., Klimley, A.P., and Cech, J.J., Jr., 2012, A lateral-displacement flume for fish behavior and stranding studies during simulated pulsed flows: Environmental Biology of Fishes, v. 93, no. 1, p. 143–150, accessed May 6, 2020, at https://doi.org/10.1007/s10641-011-9894-z.

Coggins, L.G., Jr., Yard, M.D., and Pine, W.E., III, 2011, Nonnative fish control in the Colorado River in Grand Canyon, Arizona—An effective program or serendipitous timing?: Transactions of the American Fisheries Society, v. 140, no. 2, p. 456–470, accessed January 9, 2022, at https://doi.org/10.1080/00028487.2011.572009.

Connor, E.J., and Pflug, D.E., 2004, Changes in the distribution and density of pink, chum, and Chinook salmon spawning in the upper Skagit River in response to flow management measures: North American Journal of Fisheries Management, v. 24, no. 3, p. 835–852, accessed May 3, 2020, at https://doi.org/10.1577/M03-066.1.

Cook, K.V., Lennox, R.J., Hinch, S.G., and Cooke, S.J., 2015, Fish out of water—How much air is too much?: Fisheries, v. 40, no. 9, p. 452–461, accessed January 9, 2022, at https://doi.org/10.1080/03632415.2015.1074570.

Dauwalter, D., Vidergar, D., and Kozfkay, J., 2013, A pilot study of fish stranding on the South Fork Boise River, 2012: Boise, Idaho, Trout Unlimited, 14 p., accessed May 3, 2020, at http://www.southforkboise.org/wp-content/uploads/2013/07/2012SFBoise_StrandingSummary.pdf.

Dibble, K.L., Yackulic, C.B., Kennedy, T.A., and Budy, P., 2015, Flow management and fish density regulate salmonid recruitment and adult size in tailwaters across western North America: Ecological Applications, v. 25, no. 8, p. 2168–2179, https://doi.org/10.1890/14-2211.1.

Einum, S., and Nislow, K.H., 2005, Local-scale density-dependent survival of mobile organisms in continuous habitats—An experimental test using Atlantic salmon: Oecologia, v. 143, no. 2, p. 203–210, accessed January 9, 2022, at https://doi.org/10.1007/s00442-004-1793-y.

Elliott, J.M., 1994, Quantitative ecology and the brown trout: New York, Oxford University Press, Inc., 286 p. https://doi.org/10.1093/oso/9780198546788.001.0001.

Freeman, M.C., Bowen, Z.H., Bovee, K.D., and Irwin, E.R., 2001, Flow and habitat effects on juvenile fish abundance in natural and altered flow regimes: Ecological Applications, v. 11, no. 1, p. 179–190, accessed May 4, 2020, at https://doi.org/10.1890/1051-0761(2001)011[0179:FAHEOJ]2.0.CO;2.

Gibb, A.C., Ashley-Ross, M.A., Pace, C.M., and Long, J.H., Jr., 2011, Fish out of water—Terrestrial jumping by fully aquatic fishes: Journal of Experimental Zoology Part A—Ecological Genetics and Physiology, v. 315A, no. 10, p. 649–653, accessed January 9, 2022, at https://doi.org/10.1002/jez.711.

Gingerich, A.J., Cooke, S.J., Hanson, K.C., Donaldson, M.R., Hasler, C.T., Suski, C.D., and Arlinghaus, R., 2007, Evaluation of the interactive effects of air exposure duration and water temperature on the condition and survival of angled and released fish: Fisheries Research, v. 86, nos. 2–3, p. 169–178, accessed January 9, 2022, at https://doi.org/10.1016/j.fishres.2007.06.002.

Grams, P.E., Schmidt, J.C., and Topping, D.J., 2007, The rate and pattern of bed incision and bank adjustment on the Colorado River in Glen Canyon downstream from Glen Canyon Dam, 1956–2000: Geological Society of America Bulletin, v. 119, nos. 5–6, p. 556–575, https://doi.org/10.1130/B25969.1.

Grams, P.E., Schmidt, J.C., and Topping, D.J., 2010, Bed incision and channel adjustment of the Colorado River in Glen Canyon National Recreation Area downstream from Glen Canyon Dam, *in* Melis, T.S., Hamill, J.F., Bennett, G.E., Coggins, L.G., Jr., Grams, P.E., Kennedy, T.A., Kubly, D.M., and Ralston, B.E., eds., Proceedings of the Colorado River Basin Science and Resource Management Symposium, November 18–20, 2008, Scottsdale, Arizona: U.S. Geological Survey Scientific Investigations Report 2010–5135, p. 167–176.

Halleraker, J.H., Alfredsen, K., Arnekleiv, J.V., Fjeldstad, H.P., Harby, A., and Saltveit, S.J., 1999, Environmental impacts of hydro peaking with emphasis on River Nidelva in Trondheim, Norway—Optimum use of run-of-river hydropower schemes in Proceedings for hydro power schemes, Trondheim, Norway, June 21–23, 1999: International Center for Hydropower, 7 p., accessed January 9, 2022, at https://www.researchgate.net/publication/237549128_Environmental_impacts_of_hydropeaking-with_emphasis_on_River_Nidelva_in_Trondheim_Norway/link/0046352cf10a047021000000/download.

Halleraker, J., Saltveit, S.J., Harby, A., Arnekleiv, J.V., Fjeldstad, H.-P., and Kohler, B., 2003, Factors influencing stranding of wild juvenile brown trout (*Salmo trutta*) during rapid and frequent flow decreases in artificial stream: River Research and Applications, v. 19, nos. 5–6, p. 589–603, accessed May 4, 2020, at https://doi.org/10.1002/rra.752.

Harnish, R.A., Sharma, R., McMichael, G.A., Langshaw, R.B., and Pearsons, T.N., 2014, Effect of hydroelectric dam operations on the freshwater productivity of a Columbia River fall Chinook salmon population: Canadian Journal of Fisheries and Aquatic Sciences, v. 71, no. 4, p. 602–615, accessed May 4, 2020, at https://doi.org/10.1139/cjfas-2013-0276.

Hauer, C., Unfer, G., Holzapfel, P., Haimann, M., and Habersack, H., 2014, Impact of channel bar form and grain size variability on estimated stranding risk of juvenile brown trout during hydropeaking: Earth Surface Processes and Landforms, v. 39, p. 1622–1641, accessed May 5, 2020, at https://doi.org/10.1002/esp.3552.

Hayes, D.S., Moreira, M., Boavida, I., Haslauer, M., Unfer, G., Zeiringer, B., Greimel, F., Auer, S., Ferreira, T., and Schmutz, S., 2019, Life stage-specific hydropeaking flow rules: Sustainability (Basel), v. 11, no. 6, article 1547, 17 p., accessed May 4, 2020, at https://doi.org/10.3390/su11061547.

Healy, B.D., Omana Smith, E.C., Schelly, R.C., Trammell, M.A., and Nelson, C.B., 2020, Establishment of a reproducing population of endangered humpback chub through translocations to a Colorado River tributary in Grand Canyon, Arizona: North American Journal of Fisheries Management, v. 40, no. 1, p. 278–292, accessed January 9, 2022, at https://doi.org/10.1002/nafm.10408.

Hedger, R.D., Sauterleute, J., Sundt-Hansen, L.E., Forseth, T., Ugedal, O., Diserud, O.H., and Bakken, T.H., 2018, Modelling the effect of hydropeaking-induced stranding mortality on Atlantic salmon population abundance: Ecohydrology, v. 11, no. 5, article e1960, 14 p., accessed March 30, 2020, at https://doi.org/10.1002/eco.1960.

Heggenes, J., 1988, Substrate preferences of brown trout fry (*Salmo trutta*) in artificial stream channels: Canadian Journal of Fisheries and Aquatic Sciences, v. 45, no. 10, p. 1801–1806, accessed May 5, 2020, at https://doi.org/10.1139/f88-211.

Hubert, W.A., Harris, D.D., and Wesche, T.A., 1994, Diurnal shifts in use of summer habitat by age-0 brown trout in a regulated mountain stream: Hydrobiologia, v. 284, no. 2, p. 147–156, accessed May 5, 2020, at https://doi.org/10.1007/BF00006886.

Hunter, M.A., 1992, Hydropower flow fluctuations and salmonids—A review of the biological effects, mechanical causes, and options for mitigation: Olympia, Wash., Washington Department of Fisheries, Habitat Management Division, Technical Report 119, 58 p.

Hvidsten, N.A., 1985, Mortality of pre-smolt Atlantic salmon, *Salmo salar* L., and brown trout, *Salmo trutta* L., caused by fluctuating water levels in the regulated River Nidelva, central Norway: Journal of Fish Biology, v. 27, no. 6, p. 711–718, accessed May 5, 2020, at https://doi.org/10.1111/j.1095-8649.1985.tb03215.x.

ICF Jones and Stokes, 2009, Lower Yuba River redd dewatering and fry stranding study—2008 annual report: Marysville, Calif., Yuba County Water Agency, prepared by ICF Jones and Stokes 24 p., accessed January 9, 2022, at https://www.yubawater.org/ArchiveCenter/ViewFile/Item/205.

Irvine, R.L., Oussoren, T., Baxter, J.S., and Schmidt, D.C., 2009, The effects of flow reduction rates on fish stranding in British Columbia, Canada: River Research and Applications, v. 25, no. 4, p. 405–415, accessed May 5, 2020, at https://doi.org/10.1002/rra.1172.

Irvine, R.L., Thorley, J.L., Westcott, R., Schmidt, D., and DeRosa, D., 2015, Why do fish strand?—An analysis of ten years of flow reduction monitoring data from the Columbia and Kootenay rivers, Canada: River Research and Applications, v. 31, no. 10, p. 1242–1250, accessed May 5, 2020, at https://doi.org/10.1002/rra.2823.

Kaplinski, M., Hazel, J.E., Jr., Grams, P.E., Gushue, T., Buscombe, D.D., and Kohl, K., 2022, Channel mapping Glen Canyon Dam to Lees Ferry in Glen Canyon National Recreation Area, Arizona: U.S. Geological Survey Open-File Report 2022–1057, 20 p. https://doi.org/10.3133/ofr20221057.

Kasprak, A., Sankey, J.B., and Butterfield, B.J., 2021, Future regulated flows of the Colorado River in Grand Canyon foretell decreased areal extent of sediment and increases in riparian vegetation: Environmental Research Letters, v. 16, article 014029, 14 p., accessed January 9, 2022, at https://doi.org/10.1088/1748-9326/abc9e4.

Kennedy, T.A., Muehlbauer, J.D., Yackulic, C.B., Lytle, D.A., Miller, S.W., Dibble, K.L., Kortenhoeven, E.W., Metcalfe, A.N., and Baxter, C.V., 2016, Flow management for hydropower extirpates aquatic insects, undermining river food webs: Bioscience, v. 66, no. 7, p. 561–575, accessed January 9, 2022, at https://doi.org/10.1093/biosci/biw059.

Korman, J., and Campana, S.E., 2009, Effects of hydropeaking on nearshore habitat use and growth of age-0 rainbow trout in a large regulated river: Transactions of the American Fisheries Society, v. 138, no. 1, p. 76–87, accessed January 9, 2022, at https://doi.org/10.1577/T08-026.1.

Korman, J., Yard, M., Walters, C.J., and Coggins, L.G., 2009, Effects of fish size, habitat, flow, and density on capture probabilities of age-0 rainbow trout estimated from electrofishing at discrete sites in a large river: Transactions of the American Fisheries Society, v. 138, no. 1, p. 58–75, accessed January 9, 2022, at https://doi.org/10.1577/T08-025.1.

Korman, J., Kaplinski, M., and Melis, T.S., 2011a, Effects of fluctuating flows and a controlled flood on incubation success and early survival rates and growth of age-0 rainbow trout in a large regulated river: Transactions of the American Fisheries Society, v. 140, no. 2, p. 487–505, accessed January 9, 2022, at https://doi.org/10.1080/00028487.2011.572015.

Korman, J., Walters, C.J., Martell, S.J.D., Pine, W.E., III, and Dutterer, D., 2011b, Effects of flow fluctuations on habitat use and survival of age-0 rainbow trout (*Oncorhynchus mykiss*) in a large, regulated river: Canadian Journal of Fisheries and Aquatic Sciences, v. 68, p. 1097–1109, accessed January 9, 2022, at https://doi.org/10.1139/f2011-045.

Korman, J., Martell, S.J.D., Walters, C.J., Makinster, A.S., Coggins, L.G., Yard, M.D., and Persons, W.R., 2012, Estimating recruitment dynamics and movement of rainbow trout (*Oncorhynchus mykiss*) in the Colorado River in Grand Canyon using an integrated assessment model: Canadian Journal of Fisheries and Aquatic Sciences, v. 69, no. 11, p. 1827–1849, accessed January 9, 2022, at https://doi.org/10.1139/f2012-097.

Korman, J., Yard, M.D., and Yackulic, C.B., 2016, Factors controlling the abundance of rainbow trout in the Colorado River in Grand Canyon in a reach utilized by endangered humpback chub: Canadian Journal of Fisheries and Aquatic Sciences, v. 73, no. 1, p. 105–124, accessed January 9, 2022, at https://doi.org/10.1139/cjfas-2015-0101.

Korman, J., Yard, M.D., and Kennedy, T.A., 2017, Trends in rainbow trout recruitment, abundance, survival, and growth during a boom-and-bust cycle in a tailwater fishery: Transactions of the American Fisheries Society, v. 146, no. 5, p. 1043–1057, accessed May 4, 2020, at https://doi.org/10.1080/00028487.2017.1317663.

Korman, J., Yard, M.D., Dzul, M.C., Yackulic, C.B., Dodrill, M.J., Deemer, B.R., and Kennedy, T.A., 2021, Changes in prey, turbidity, and competition reduce somatic growth and cause the collapse of a fish population: Ecological Monographs, v. 91, no. 1, article e01427, 20 p., accessed January 9, 2022, at https://doi.org/10.1002/ecm.1427.

Korman, J., Giardina, M.A., Yard, M.D., Wright, S.A., Kaplinski, M., and Bennett, G., 2024, Colorado River milage system and ancillary attribute data for connecting to hydrodynamic model output in Glen Canyon, AZ: U.S. Geological Survey data release, https://doi.org/10.5066/P9L1XEZO.

Larranaga, N., Valdimarsson, S.K., Linnansaari, T., and Steingrímsson, S.Ó., 2018, Diel activity and foraging mode of juvenile Arctic charr in fluctuating water flow: River Research and Applications, v. 34, no. 4, p. 348–356, accessed May 4, 2020, at https://doi.org/10.1002/rra.3256.

LeRoy Heman, M., Campbell, R.S., and Redmond, L.C., 1969, Manipulation of fish populations through reservoir drawdown: Transactions of the American Fisheries Society, v. 98, no. 2, p. 293–304, accessed May 4, 2020, at https://doi.org/10.1577/1548-8659(1969)98[293:MOFPTR]2.0.CO;2.

Lobón-Cerviá, J., 2007, Numerical changes in stream-resident brown-trout (*Salmo trutta*)—Uncovering the roles of density-dependent and density-independent factors across space and time: Canadian Journal of Fisheries and Aquatic Sciences, v. 64, no. 10, p. 1429–1447, accessed January 9, 2022, at https://doi.org/10.1139/f07-111.

Maddux, H.R., Kubly, D.M., deVos, J.C., Jr., Persons, W.R., Staedicke, R., and Wright, R.L., 1987, Effects of varied flow regimes on aquatic resources of Glen and Grand Canyons—Final report: Flagstaff, Ariz., Bureau of Reclamation, Glen Canyon Environmental Studies, prepared by Arizona Game and Fish Department, under contract no. 4-AG-40-01810, 291 p., accessed January 9, 2022, at http://npshistory.com/publications/grca/varied-flow-regimes-1987.pdf.

McKinney, T., Speas, D.W., Rogers, R.S., and Persons, W.R., 2001, Rainbow trout in a regulated river below Glen Canyon Dam, Arizona, following increased minimum flows and reduced discharge variability: North American Journal of Fisheries Management, v. 21, no. 1, p. 216–222, https://doi.org/10.1577/1548-8675(2001)021<0216:RTIARR>2.0.CO;2.

McKinney, T., Speas, D.W., Rogers, R.W., and Persons, W.R., 1999, Rainbow trout in the Lees Ferry recreational fishery below Glen Canyon Dam, Arizona, following establishment of minimum flow requirements—Final Report: Flagstaff, Ariz., Bureau of Reclamation, Grand Canyon Monitoring and Research Center, prepared by Arizona Game and Fish Department, Phoenix, Ariz., under cooperative agreement no. 1425-98-FC-40-22690, 109 p.

Melis, T.S., Korman, J., and Kennedy, T.A., 2012, Abiotic and biotic responses of the Colorado River to controlled floods at Glen Canyon Dam, Arizona, USA: River Research and Applications, v. 28, no. 6, p. 764–776, accessed January 9, 2022, at https://doi.org/10.1002/rra.1503.

Moreira, M., Hayes, D.S., Boavida, I., Schletterer, M., Schmutz, S., and Pinheiro, A., 2019, Ecologically-based criteria for hydropeaking mitigation—A review: Science of the Total Environment, v. 657, p. 1508–1522, accessed May 4, 2020, https://doi.org/10.1016/j.scitotenv.2018.12.107.

Mueller, G.A., 2005, Predatory fish removal and native fish recovery in the Colorado River mainstem—What have we learned?: Fisheries, v. 30, no. 9, p. 10–19, accessed January 9, 2022, at https://doi.org/10.1577/1548-8446(2005)30[10:PFRANF]2.0.CO;2.

Nagrodski, A., Raby, G.D., Hasler, C.T., Taylor, M.K., and Cooke, S.J., 2012, Fish stranding in freshwater systems—Sources, consequences, and mitigation: Journal of Environmental Management, v. 103, p. 133–141, accessed January 9, 2022, at https://doi.org/10.1016/j.jenvman.2012.03.007.

Nelson, J. M., Shimizu, Y., Abe, T., Asahi, K., Gamou, M., Inoue, T., Watanabe, Y., 2016, The international river interface cooperative—Public domain flow and morphodynamics software for education and applications: Advances in Water Resources, v. 93, p. 62–74. https://doi.org/10.1016/j.advwatres.2015.09.017.

Olson, F.W., and Metzgar, R.G., 1987, Downramping to minimize stranding of salmonid fry, *in* Clowes, B.W., ed., Water Power '87—International Conference on Hydropower, Portland, Oregon, August 19–21, 1987 [Proceedings]: American Society of Civil Engineers, p. 691–701, accessed May 4, 2020, at https://cedb.asce.org/CEDBsearch/record.jsp?dockey=0055575.

Persons, B., 2002, Final Report—Lees Ferry Creel Data Compilation Project: under Cooperative Agreement 1425-98-FC-40-22690 Modification 008, 90 p., accessed January 9, 2022, at https://www.riversimulator.org/Resources/GCMRC/Aquatic/Persons2002.pdf.

Pflug, D.E., Chantrill, C., Mobrand, L., Swanson, K., and Kurko, K., 1989, Skagit River salmon and steelhead fry stranding studies: Seattle, Wash., Seattle City Light Environmental Affairs Division 411 p., 6 apps., accessed January 9, 2022, at https://www.seattle.gov/light/skagit/Relicensing/cs/groups/secure/@scl.skagit.team/documents/document/cm9k/ntcw/~edisp/prod570975.pdf.

Phinney, L.A., 1974, Further observations on juvenile salmon stranding in the Skagit River, March 1973: Olympia, Wash., State of Washington Department of Fisheries, Progress report no. 26, 68 p.

Porter, M., and Rosenfeld, J., 1999, Microhabitat selection and partitioning by an assemblage of fish in the Nazko River: Victoria, British Columbia, British Columbia Ministry of Fisheries Fish Project Report no. RD77, 28 p.

Puffer, M., Berg, O.K., Hamnes, F.B., Bentsen, V., Koch, W., Ugedal, O., Forseth, T., Arnekleiv, J.V., and Einum, S., 2019, Density-independent use of shallow riverine areas in juvenile Atlantic salmon: Canadian Journal of Fisheries and Aquatic Sciences, v. 76, no. 7, p. 1161–1170, accessed May 4, 2020, at https://doi.org/10.1139/cjfas-2017-0500.

Randle, T.J., and Pemberton, E.L., 1987, Results and analysis of STARS modeling efforts of the Colorado river in Grand Canyon: Flagstaff, Arizona, Glen Canyon Environmental Studies, NTIS Report PB88-183421.

Rempel, L.L., Healy, K., and Lewis, F.J.A., 2012, Lower Fraser River juvenile fish habitat suitability criteria: Vancouver, British Columbia, Canadian Technical Report of Fisheries and Aquatic Sciences 2991, 73 p.

Rolls, R.J., Leigh, C., and Sheldon, F., 2012, Mechanistic effects of low-flow hydrology on riverine ecosystems—Ecological principles and consequences of alteration: Freshwater Science, v. 31, no. 4, p. 1163–1186, accessed May 4, 2020, at https://doi.org/10.1899/12-002.1.

Rubin, D.M., Topping, D.J., Schmidt, J.C., Hazel, J., Kaplinski, M., and Melis, T.S., 2002, Recent sediment studies refute Glen Canyon Dam hypothesis: Eos, v. 83, no. 25, p. 273–278, accessed January 9, 2022, at https://doi.org/10.1029/2002EO000191.

Runge, M.C., Yackulic, C.B., Bair, L.S., Kennedy, T.A., Valdez, R.A., Ellsworth, C., Kershner, J.L., Rogers, R.S., Trammell, M.A., and Young, K.L., 2018, Brown trout in the Lees Ferry reach of the Colorado River—Evaluation of causal hypotheses and potential interventions: U.S. Geological Survey Open-File Report 2018–1069, 83 p., accessed January 9, 2022, at https://doi.org/10.3133/ofr20181069.

Saltveit, S.J., Halleraker, J.H., Arnekleiv, J.V., and Harby, A., 2001, Field experiments on stranding in juvenile Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*) during rapid flow decreases caused by hydropeaking: Regulated Rivers, v. 17, nos. 4–5, p. 609–622, accessed May 4, 2020, at https://doi.org/10.1002/rrr.652.

Sankey, J.B., Ralston, B.E., Grams, P.E., Schmidt, J.C., and Cagney, L.E., 2015, Riparian vegetation, Colorado River, and climate—Five decades of spatiotemporal dynamics in the Grand Canyon with river regulation: Journal of Geophysical Research. Biogeosciences, v. 120, no. 8, p. 1532–1547, accessed May 4, 2020, at https://doi.org/10.1002/2015JG002991.

Sauterleute, J.F., Hedger, R.D., Hauer, C., Pulg, U., Skoglund, H., Sundt-Hansen, L.E., Bakken, T.H., and Ugedal, O., 2016, Modelling the effects of stranding on the Atlantic salmon population in the Dale River, Norway: Science of the Total Environment, v. 573, p. 574–584, accessed May 4, 2020, at https://doi.org/10.1016/j.scitotenv.2016.08.080.

Schmidt, J.C., and Grams, P.E., 2011, The high flows—Physical science results, chap. 3, *in* Melis, T.S., ed., Effects of three high-flow experiments on the Colorado River ecosystem downstream from Glen Canyon Dam, Arizona: U.S. Geological Survey Circular 1366, p. 53–91.

Schmutz, S., Bakken, T.H., Friedrich, T., Greimel, F., Harby, A., Jungwirth, M., Melcher, A., Unfer, G., and Zeiringer, B., 2015, Response of fish communities to hydrological and morphological alterations in hydropeaking rivers of Austria: River Research and Applications, v. 31, no. 8, p. 919–930, accessed May 5, 2020, at https://doi.org/10.1002/rra.2795.

Shi, K., Dong, S., Zhou, Y., Gao, Q., Li, L., Zhang, M., and Sun, D., 2018, Comparative evaluation of toleration to heating and hypoxia of three kinds of salmonids: Journal of Ocean University of China, v. 17, no. 6, p. 1465–1472, accessed January 9, 2022, at https://doi.org/10.1007/s11802-018-3673-9.

Thompson, J.S., 1970, The effect of water flow regulation at Gorge Dam on stranding of salmon in the Skagit River 1969–1970: F.P.C. 553, State of Washington Department of Fisheries, Management and Research Division, 50 p, accessed January 9, 2022, at https://www.seattle.gov/light/skagit/relicensing/cs/groups/secure/@scl.skagit.team/documents/document/cm9k/ntuz/~edisp/prod553741.pdf.

Tiffan, K.F., Garland, R.D., and Rondorf, D.W., 2002, Quantifying flow-dependent changes in subyearling fall Chinook salmon rearing habitat using two-dimensional spatially explicit modeling: North American Journal of Fisheries Management, v. 22, no. 3, p. 713–726, accessed May 4, 2020, at https://doi.org/10.1577/1548-8675(2002)022<0713:QFDCIS>2.0.CO;2.

Tuhtan, J.A., Noack, M., and Wieprecht, S., 2012, Estimating stranding risk due to hydropeaking for juvenile European grayling considering river morphology: KSCE Journal of Civil Engineering, v. 16, no. 2, p. 197–206, accessed May 4, 2020, at https://doi.org/10.1007/s12205-012-0002-5.

Ugedal, O., Næsje, T.F., Thorstad, E.B., Forseth, T., Saksgård, L.M., and Heggberget, T.G., 2008, Twenty years of hydropower regulation in the River Alta—Long-term changes in abundance of juvenile and adult Atlantic salmon: Hydrobiologia, v. 609, no. 1, p. 9–23, accessed May 4, 2020, at https://doi.org/10.1007/s10750-008-9404-2.

U.S. Department of Interior, 1995, Operation of Glen Canyon Dam Final Environmental Impact Statement, Colorado River Storage Project, Coconino County, Arizona: Salt Lake City, Utah, Bureau of Reclamation, Upper Colorado Regional Office, 337 p.

U.S. Department of Interior, 1996, Record of Decision, Operation of Glen Canyon Dam Final Environmental Impact Statement: Washington, D.C., Office of the Secretary of the Interior, 15 p.

U.S. Department of Interior, 2016a, Attachment E—U.S. Fish and Wildlife Service Biological Opinion for the Glen Canyon dam long-term experimental and management plan, *in* Record of decision for the Glen Canyon Dam long-term experimental plan final environmental impact statement: Salt Lake City, Utah, Bureau of Reclamation, Upper Colorado Region, National Park Service, Intermountain Region, p. E-1–E-94, accessed January 9, 2022, at https://ltempeis.anl.gov/documents/docs/LTEMP_ROD.pdf.

U.S. Department of Interior, 2016b, Glen Canyon Dam long-term experimental and management plan final environmental impact statement: U.S. Department of the Interior, Bureau of Reclamation, Upper Colorado Region, National Park Service, Intermountain Region, 8 chapters plus 17 appendixes, accessed January 9, 2022, at https://ltempeis.anl.gov/documents/final-eis/.

U.S. Department of Interior, 2016c, Record of decision for the Glen Canyon Dam long-term experimental and management plan final environmental impact statement: Salt Lake City, Utah, U.S. Department of the Interior, Bureau of Reclamation, Upper Colorado Region, National Park Service, Intermountain Region, 196 p., accessed January 9, 2022, at https://ltempeis.anl.gov/documents/docs/LTEMP_ROD.pdf.

U.S. Fish and Wildlife Service, 2021, Reclassification of the humpback chub from endangered to threatened with a section 4(d) rule: Federal Register, v. 86, no. 198, p. 57588–57610, accessed January 9, 2022, at https://www.federalregister.gov/documents/2021/10/18/2021-20964/endangered-and-threatened-wildlife-and-plants-reclassification-of-the-humpback-chub-from-endangered.

U.S. Geological Survey [USGS], 2022, USGS water data for the Nation: U.S. Geological Survey National Water Information System database, accessed January 9, 2022, at https://waterdata.usgs.gov/nwis/inventory/?site_no=09380000.

Van Haverbeke, D.R., Stone, D.M., Dodrill, M.J., Young, K.L., and Pillow, M.J., 2017, Population expansion of humpback chub in western Grand Canyon and hypothesized mechanisms: The Southwestern Naturalist, v. 62, no. 4, p. 285–292, accessed January 9, 2022, at https://doi.org/10.1894/0038-4909-62.4.285.

Vehanen, T., Bjerke, P.L., Heggenes, J., Huusko, A., and Mäki–Petäys, A., 2000, Effect of fluctuating flow and temperature on cover type selection and behaviour by juvenile brown trout in artificial flumes: Journal of Fish Biology, v. 56, no. 4, p. 923–937, accessed May 4, 2020, at https://doi.org/10.1111/j.1095-8649.2000.tb00882.x.

Wiele, S.M., and Smith, J.D., 1996, A reach-averaged model of diurnal discharge wave propagation down the Colorado River through the Grand Canyon: Water Resources Research, v. 32, no. 5, p. 1375–1386, https://doi.org/10.1029/96WR00199.

Wright, S.A., Kaplinski, M., and Grams, P.E., 2024, Hydrodynamic model of the Colorado River, Glen Canyon Dam to Lees Ferry in Glen Canyon National Recreation Area, Arizona—Tables of model results and accuracy assessment: U.S. Geological Survey data release, accessed May 1, 2024, at https://doi.org/10.5066/P1QTRNEB.

Yackulic, C.B., Dodrill, M., Dzul, M., Sanderlin, J.S., and Reid, J.A., 2020, A need for speed in Bayesian population models—A practical guide to marginalizing and recovering discrete latent states: Ecological Applications, accessed January 9, 2022, at https://esajournals.onlinelibrary.wiley.com/doi/am-pdf/10.1002/eap.2112.

Yackulic, C.B., Korman, J., Yard, M.D., and Dzul, M.C., 2018, Inferring species interactions through joint mark-recapture analysis: Ecology, v. 99, no. 4, p. 812–821, accessed May 4, 2020, at https://doi.org/10.1002/ecy.2166.

Yackulic, C.B., Yard, M.D., Korman, J., and Van Haverbeke, D.R., 2014, A quantitative life history of endangered humpback chub that spawn in the Little Colorado River—Variation in movement, growth, and survival: Ecology and Evolution, v. 4, no. 7, p. 1006–1018, accessed January 9, 2022, at https://doi.org/10.1002/ece3.990.

Yard, M.D., Bennett, G.E., Mietz, S.N., Coggins, L.G., Jr., Stevens, L.E., Hueftle, S., and Blinn, D.W., 2005, Influence of topographic complexity on solar insolation estimates for the Colorado River, Grand Canyon, AZ: Ecological Modelling, v. 183, nos. 2–3, p. 157–172, accessed May 4, 2020, at https://doi.org/10.1016/j.ecolmodel.2004.07.027.

Yard, M.D., Coggins, L.G., Jr., Baxter, C.V., Bennett, G.E., and Korman, J., 2011, Trout piscivory in the Colorado River, Grand Canyon—Effects of turbidity, temperature, and fish prey availability: Transactions of the American Fisheries Society, v. 140, no. 2, p. 471–486, accessed January 9, 2022, at https://doi.org/10.1080/00028487.2011.572011.

Yard, M.D., Korman, J., Walters, C.J., and Kennedy, T.A., 2016, Seasonal and spatial patterns of growth of rainbow trout in the Colorado River in Grand Canyon, Arizona: Canadian Journal of Fisheries and Aquatic Sciences, v. 73, no. 1, p. 125–139, accessed January 9, 2022, at https://doi.org/10.1139/cjfas-2015-0102.

Young, P.S., Cech, J.J., Jr., and Thompson, L.C., 2011, Hydropower-related pulsed-flow impacts on stream fishes—A brief review, conceptual model, knowledge gaps, and research needs: Reviews in Fish Biology and Fisheries, v. 21, no. 4, p. 713–731, accessed May 4, 2020, at https://doi.org/10.1007/s11160-011-9211-0.

Appendix

Appendix 1. Stranding Papers for Literature Review

This appendix includes all sources utilized for the Literature review on fish stranding. These sources were evaluated on whether they quantified factors leading to stranding (see table 1).

Literature Review References

Anglin, D.R., Haeseker, S., Skalicky, J., Schaller, H., Tiffan, K., Hatten, J., Rondorf, D., Hoffarth, P., Nugent, J., Benner, D., and Yoshinaka, M., 2006, Effects of hydropower operations on spawning habitat, rearing habitat, and stranding/entrapment mortality of fall Chinook salmon in the Hanford Reach of the Columbia River: U.S. Fish and Wildlife Service, 260 p., accessed April 4, 2020, at https://pubs.er.usgs.gov/publication/70198141.

Auer, S., Zeiringer, B., Fuhrer, S., Tonolla, D., and Schmutz, S., 2017, Effects of river bank heterogeneity and time of day on drift and stranding of juvenile European grayling (*Thymallus thymallus* L.) caused by hydropeaking: Science of the Total Environment, v. 575, p. 1515–1521, accessed March 30, 2020, at https://doi.org/10.1016/j.scitotenv.2016.10.029.

Bauersfeld, K., 1978, Stranding of juvenile salmon by flow reductions at Mayfield Dam on the Cowlitz River, 1976: Olympia, Wash., Washington Department of Fisheries, Tech. Rep. 36, 36 p.

Becker, C.D., Neitzel, D.A., and Carlile, D.W., 1986, Survival data for dewatered rainbow trout (*Salmo gairdneri* Rich.) eggs and alevins: Journal of Applied Ichthyology, v. 2, no. 3, p. 102–110, accessed May 5, 2020, at https://doi.org/10.1111/j.1439-0426.1986.tb00436.x.

Bell, E., Kramer, S., Zajanc, D., and Aspittle, J., 2008, Salmonid fry stranding mortality associated with daily water level fluctuations in Trail Bridge reservoir, Oregon: North American Journal of Fisheries Management, v. 28, no. 5, p. 1515–1528, accessed May 20, 2020, at https://doi.org/10.1577/M07-026.1.

Berland, G., Nickelsen, T., Heggenes, J., Okland, F., Thorstad, E.B., and Halleraker, J., 2004, Movements of wild atlantic salmon parr in relation to peaking flows below a hydropower station: River Research and Applications, v. 20, no. 8, p. 957–966, accessed May 15, 2020, at https://doi.org/10.1002/rra.802.

Bradford, M.J., 1997, An experimental study of stranding of juvenile salmonids on gravel bars and in sidechannels during rapid flow decreases: Regulated Rivers, v. 13, no. 5, p. 395–401, accessed March 30, 2020, at https://doi.org/10.1002/(SICI)1099-1646(199709/10)13:5<395::AID-RRR464>3.0.CO;2-L.

Bradford, M.J., Taylor, G.C., Allan, J.A., and Higgins, P.S., 1995, An experimental study of the stranding of juvenile coho salmon and rainbow trout during rapid flow decreases under winter conditions: North American Journal of Fisheries Management, v. 15, no. 2, p. 473–479, accessed May 2, 2020, at https://doi.org/10.1577/1548-8675(1995)015<0473:AESOTS>2.3.CO;2.

Carty, D.G., 1985, Potential impacts of altering discharge pattern from Hauser Dam, Missouri River, on young-of-the-year brown trout and rainbow trout: Bozeman, Mont., Montana State University, Ph.D dissertation, 84 p., accessed January 9, 2022, at https://scholarworks.montana.edu/xmlui/handle/1/4916.

Casas-Mulet, R., Saltveit, S.J., and Alfredsen, K., 2015, The survival of Atlantic salmon (*Salmo salar*) eggs during dewatering in a river subjected to hydropeaking: River Research and Applications, v. 31, no. 4, p. 433–446, accessed May 5, 2020, at https://doi.org/10.1002/rra.2827.

Chamberlain, C.D., 2003, Trinity River juvenile fish stranding evaluation, May to June 2002: U.S. Fish and Wildlife Service, Arcata Fish and Wildlife Office, Calif., Report Number AFWO-F-01-03, 42 p., 3 app., accessed January 9, 2022, at https://www.trrp.net/DataPort/doc.php?id=234.

Choi, S.U., Kim, S.K., Choi, B., and Kim, Y., 2017, Impact of hydropeaking on downstream fish habitat at the Goesan Dam in Korea: Ecohydrology, v. 10, no. 6, p. e1861, accessed May 7, 2020, at https://doi.org/10.1002/eco.1861.

Cocherell, S.A., Chun, S.N., Cocherell, D.E., Thompson, L.C., Klimley, A.P., and Cech, J.J., Jr., 2012, A lateral-displacement flume for fish behavior and stranding studies during simulated pulsed flows: Environmental Biology of Fishes, v. 93, no. 1, p. 143–150, accessed May 6, 2020, at https://doi.org/10.1007/s10641-011-9894-z.

Connor, E.J., and Pflug, D.E., 2004, Changes in the distribution and density of pink, chum, and Chinook salmon spawning in the upper Skagit River in response to flow management measures: North American Journal of Fisheries Management, v. 24, no. 3, p. 835–852, accessed May 3, 2020, at https://doi.org/10.1577/M03-066.1.

Dauwalter, D., Vidergar, D., and Kozfkay, J., 2013, A pilot study of fish stranding on the South Fork Boise River, 2012: Boise, Idaho, Trout Unlimited, 14 p., accessed May 3, 2020, at http://www.southforkboise.org/wp-content/uploads/2013/07/2012SFBoise_StrandingSummary.pdf.

Dudley, P.N., 2019, Insights from an individual based model of a fish population on a large regulated river: Environmental Biology of Fishes, v. 102, no. 8, p. 1069–1095, accessed May 3, 2020, at https://doi.org/10.1007/s10641-019-00891-6.

Ewertzh, J., 2017, Frequency regulation and its effects on fish stranding—A hydrological modelling example of a Swedish river: Stockholm, Sweden, Royal Institute of Technology, M.S. thesis, 59 p., 1 app., accessed May 3, 2020, at http://www.diva-portal.org/smash/get/diva2:1140145/FULLTEXT01.pdf.

Faulkner, S., Sparling, M., Parsamanesh, A., and Lewis., A., 2019, BC Hydro Seton Generating Station—BRGMON-9 Addendum 1—Lower Fraser River fish stranding risk assessment year 8 (2019): Castlegar, British Columbia, BC Hydro, prepared by Ecofish Research Ltd., Consultant's draft report, 50 p., accessed January 9, 2022, at https://www.bchydro.com/content/dam/BCHydro/customer-portal/documents/corporate/environment-sustainability/water-use-planning/lower-mainland/BRGMON-09%20Yr1%202019-12-17.pdf.

Gadomski, D.M., and Wagner, P.G., 2009, Factors Affecting the Age-0 Resident Fish Community Along Shorelines of the Hanford Reach of the Columbia River: Northwest Science, v. 83, no. 3, p. 180–188, accessed May 4, 2020, at https://doi.org/10.3955/046.083.0302.

Gibbins, C.N., and Acornley, R.M., 2000, Salmonid habitat modelling studies and their contribution to the development of an ecologically acceptable release policy for Kielder Reservoir, North-east England: Regulated Rivers, v. 16, no. 3, p. 203–224, accessed May 5, 2020, at https://doi.org/10.1002/(SICI)1099-1646(200005/06)16:3<203::AID-RRR579>3.0.CO;2-8.

Golder Associates Ltd, 2016, Lower Columbia River (CLBMON #42[A]) and Kootenay River fish stranding assessments—Annual summary (April 2015 to April 2016): Castlegar, British Columbia, BC Hydro, prepared by Golder Associates Ltd. [report no. 1407618], 28 p., 2 app., accessed January 9, 2022, at https://www.bchydro.com/content/dam/BCHydro/customer-portal/documents/corporate/environment-sustainability/water-use-planning/southern-interior/clbmon-42a-yr10-2017-05-30.pdf.

Golder Associates Ltd, and Poisson Consulting Ltd., 2019, DDMMON-16 Lower Duncan River fish stranding impact monitoring—Year 10 report (April 2017 to April 2018): Castlegar, British Columbia, BC Hydro, prepared by Golder Associates Ltd. [report no. 1535517F], 50 p., 3 app., accessed January 9, 2022, at https://www.bchydro.com/content/dam/BCHydro/customer-portal/documents/corporate/environment-sustainability/water-use-planning/lower-mainland/DDMMON-16%20Yr10%202019-12-04.pdf.

Halleraker, J.H., Alfredsen, K., Arnekleiv, J.V., Fjeldstad, H.P., Harby, A., and Saltveit, S.J., 1999, Environmental impacts of hydro peaking with emphasis on River Nidelva in Trondheim, Norway—Optimum use of run-of-river hydropower schemes *in* Proceedings for hydro power schemes, Trondheim, Norway, June 21–23, 1999: International Center for Hydropower, 7 p., accessed January 9, 2022, at https://www.researchgate.net/publication/237549128_Environmental_impacts_of_hydropeaking-with_emphasis_on_River_Nidelva_in_Trondheim_Norway/link/0046352cf10a047021000000/download.

Halleraker, J., Saltveit, S.J., Harby, A., Arnekleiv, J.V., Fjeldstad, H.-P., and Kohler, B., 2003, Factors influencing stranding of wild juvenile brown trout (*Salmo trutta*) during rapid and frequent flow decreases in artificial stream: River Research and Applications, v. 19, no. 5–6, p. 589–603, accessed May 4, 2020, at https://doi.org/10.1002/rra.752.

Harnish, R.A., Sharma, R., McMichael, G.A., Langshaw, R.B., and Pearsons, T.N., 2014, Effect of hydroelectric dam operations on the freshwater productivity of a Columbia River fall Chinook salmon population: Canadian Journal of Fisheries and Aquatic Sciences, v. 71, no. 4, p. 602–615, accessed May 4, 2020, at https://doi.org/10.1139/cjfas-2013-0276.

Hauer, C., Unfer, G., Holzapfel, P., Haimann, M., and Habersack, H., 2014, Impact of channel bar form and grain size variability on estimated stranding risk of juvenile brown trout during hydropeaking: Earth Surface Processes and Landforms, v. 39, p. 1622–1641, accessed May 5, 2020, at https://doi.org/10.1002/esp.3552.

Hedger, R.D., Sauterleute, J., Sundt-Hansen, L.E., Forseth, T., Ugedal, O., Diserud, O.H., and Bakken, T.H., 2018, Modelling the effect of hydropeaking-induced stranding mortality on Atlantic salmon population abundance: Ecohydrology, v. 11, no. 5, p. e1960, accessed March 30, 2020, at https://doi.org/10.1002/eco.1960.

Higgins, P.S., and Bradford, M.J., 1996, Evaluation of a large-scale fish salvage to reduce the impacts of controlled flow reduction in a regulated river: North American Journal of Fisheries Management, v. 16, no. 3, p. 666–673, accessed May 3, 2020, at https://doi.org/10.1577/1548-8675(1996)016<0666:EOALSF>2.3.CO;2.

Hoffarth, P., 2004, 2004 evaluation of juvenile fall Chinook salmon entrapment in the Hanford Reach of the Columbia River: Olympia, Wash., Washington Department of Fish and Wildlife, 56 p.

Huntingford, F.A., Aird, D., Joiner, P., Thorpe, K.E., Braithwaite, V.A., and Armstrong, J.D., 1999, How juvenile Atlantic salmon, *Salmo salar* L., respond to falling water levels—Experiments in an artificial stream: Fisheries Management and Ecology, v. 6, no. 5, p. 357–364, accessed May 6, 2020, at https://doi.org/10.1046/j.1365-2400.1999.00161.x.

Hvidsten, N.A., 1985, Mortality of pre-smolt Atlantic salmon, *Salmo salar* L., and brown trout, *Salmo trutta* L., caused by fluctuating water levels in the regulated River Nidelva, central Norway: Journal of Fish Biology, v. 27, no. 6, p. 711–718, accessed May 5, 2020, at https://doi.org/10.1111/j.1095-8649.1985.tb03215.x.

ICF Jones and Stokes, 2009, Lower Yuba River redd dewatering and fry stranding study—2008 annual report: Marysville, Calif., Yuba County Water Agency, prepared by IFC Jones & Stokes, 24 p., accessed January 9, 2022, at https://www.yubawater.org/Archive/ViewFile/Item/239.

Irvine, R.L., Oussoren, T., Baxter, J.S., and Schmidt, D.C., 2009, The effects of flow reduction rates on fish stranding in British Columbia, Canada: River Research and Applications, v. 25, no. 4, p. 405–415, accessed May 5, 2020, at https://doi.org/10.1002/rra.1172.

Irvine, R.L., Thorley, J.L., Westcott, R., Schmidt, D., and DeRosa, D., 2015, Why do fish strand?—An analysis of ten years of flow reduction monitoring data from the Columbia and Kootenay rivers, Canada: River Research and Applications, v. 31, no. 10, p. 1242–1250, accessed May 5, 2020, at https://doi.org/10.1002/rra.2823.

Jones and Stokes Associates, 1985, Salmon and steelhead fry trapping and stranding in potholes on the Skagit River, 1984: Seattle, Wash., Seattle Department of Lighting, prepared by Jones & Stokes Associates, 200 p, accessed January 9, 2022, at https://www.seattle.gov/light/skagit/relicensing/cs/groups/secure/@scl.skagit.team/documents/document/cm9k/ntuz/~edisp/prod553737.pdf.

Juárez, A., Adeva-Bustos, A., Alfredsen, K., and Dønnum, B.O., 2019, Performance of a two-dimensional hydraulic model for the evaluation of stranding areas and characterization of rapid fluctuations in hydropeaking rivers: Water (Basel), v. 11, no. 2, 26 p., accessed May 2, 2020, at https://doi.org/10.3390/w11020201.

Maddux, H.R., Kubly, D.M., deVos, J.C., Jr., Persons, W.R., Staedicke, R., and Wright, R.L., 1987, Effects of varied flow regimes on aquatic resources of Glen and Grand Canyons—Final report: Flagstaff, Ariz., Bureau of Reclamation, Glen Canyon Environmental Studies, prepared by Arizona Game and Fish Department, under contract no. 4-AG-40-01810, 291 p., accessed January 9, 2022, at https://www.gcmrc.gov/library/reports/Biological/Fish_Studies/Maddux1987.pdf.

May, C.L., and Lee, D.C., 2004, The relationships among in-channel sediment storage, pool depth, and summer survival of juvenile salmonids in Oregon Coast Range streams: North American Journal of Fisheries Management, v. 24, no. 3, p. 761–774, accessed May 3, 2020, at https://doi.org/10.1577/M03-073.1.

Moog, O., 1993, Quantification of daily peak hydropower effects on aquatic fauna and management to minimize environmental impacts: Regulated Rivers, v. 8, no. 1–2, p. 5–14, accessed May 4, 2020, at https://doi.org/10.1002/rrr.3450080105.

Nelson, J. M., Shimizu, Y., Abe, T., Asahi, K., Gamou, M., Inoue, T., and Watanabe, Y., 2016, The international river interface cooperative—Public domain flow and morphodynamics software for education and applications: Advances in Water Resources, v. 93, p. 62–74. https://doi.org/10.1016/j.advwatres.2015.09.017.

Normandeau Associates, Inc., and Gomez and Sullivan Engineers, P.C., 2012, Final study report—Downstream flow ramping and stranding study RSP 3.8 Conowingo hydroelectric project (FERC project number 405): Exelon Generation Company, LLC, prepared by Normandeau Associates, Inc., and Gomez and Sullivan Engineers, P.C., 53 p, accessed January 9, 2022, at https://mde.maryland.gov/programs/Water/WetlandsandWaterways/Documents/ExelonMD/FERC/Conowingo-FRSP-3.08.pdf.

Nugent, J., Nugent, M., and Brock, W., 2002, Evaluation of juvenile fall Chinook salmon stranding on the Hanford Reach of the Columbia River, 2001 annual report: Grant County, Wash., Public Utility District Number 2 prepared by Washington Department of Fish and Wildlife, [report no. DOE/BP-00004294-3], 85 p., accessed January 9, 2022, at https://www.osti.gov/biblio/798759.

Nugent, J., 2002, Evaluation of juvenile fall Chinook salmon stranding on the Hanford Reach of the Columbia River, 1999 Annual Report: Grant County, Wash., Bonneville Power Administration and Public Utility District Number 2, prepared by Washington Department of Fish and Wildlife, [report no. DOE/BP-00004294-1], 215 p., accessed January 9, 2022, at https://digital.library.unt.edu/ark:/67531/metadc739411/m1/91/.

Olson, F.W., and Metzgar, R.G., 1987, Downramping to minimize stranding of salmonid fry, *in* Clowes, B.W., ed., Water Power ‘87—International Conference on Hydropower, Portland, Oregon, August 19–21, 1987 [Proceedings]: American Society of Civil Engineers, p. 691–701, accessed May 4, 2020, at https://cedb.asce.org/CEDBsearch/record.jsp?dockey=0055575.

Patterson, A., and Hawker, S., 2018, Site C Clean Energy Project—Site C Fish Stranding Monitoring Program (Mon-12), Construction Year 3 (2017): Castlegar, British Columbia, BC Hydro, prepared by Ecora, 45 p., accessed January 9, 2022, at https://www.sitecproject.com/sites/default/files/mon-12-site-c-fish-stranding-2017-annual-report.pdf.

Perkins, W.A., Richmond, M.C., and McMichael, G.A., 2004, Two-dimensional modeling of time-varying hydrodynamics and juvenile Chinook salmon habitat in the Hanford Reach of the Columbia River *in* Sehlke, G., Hayes, D.F., and Stevens, D.K., eds., Critical transitions in water and environmental resources management—World Water and Environmental Resources Congress 2004, Salt Lake City, Utah, June 27–July 1, 2004 [Proceedings]: Reston, Va., American Society of Civil Engineers, p. 1–8, accessed May 4, 2020, at https://doi.org/10.1061/40737(2004)180.

Pflug, D.E., Chantrill, C., Mobrand, L., Swanson, K., and Kurko, K., 1989, Skagit River salmon and steelhead fry stranding studies: Seattle, Wash., Seattle City Light Environmental Affairs Division, 411 p., 6 apps., accessed January 9, 2022, at https://www.seattle.gov/light/skagit/Relicensing/cs/groups/secure/@scl.skagit.team/documents/document/cm9k/ntcw/~edisp/prod570975.pdf.

Phinney, L.A., 1974, Further observations on juvenile salmon stranding in the Skagit River, March 1973: State of Washington Department of Fisheries, Progress report no. 26, 68 p.

Premstaller, G., Cavedon, V., Pisaturo, G.R., Schweizer, S., Adami, V., and Righetti, M., 2017, Hydropeaking mitigation project on a multi-purpose hydro-scheme on Valsura River in South Tyrol/Italy: Science of the Total Environment, v. 574, p. 642–653, accessed May 4, 2020, at https://doi.org/10.1016/j.scitotenv.2016.09.088.

Puffer, M., Berg, O.K., Hamnes, F.B., Bentsen, V., Koch, W., Ugedal, O., Forseth, T., Arnekleiv, J.V., and Einum, S., 2019, Density-independent use of shallow riverine areas in juvenile Atlantic salmon: Canadian Journal of Fisheries and Aquatic Sciences, v. 76, no. 7, p. 1161–1170, accessed May 4, 2020, at https://doi.org/10.1139/cjfas-2017-0500.

Reiser, D.W., and White, R.G., 1983, Effects of complete redd dewatering on salmonid egg-hatching success and development of juveniles: Transactions of the American Fisheries Society, v. 112, no. 4, p. 532–540, accessed May 5, 2020, at https://doi.org/10.1577/1548-8659(1983)112<532:EOCRDO>2.0.CO;2.

Richmond, M.C., and Perkins, W.A., 2009, Efficient calculation of dewatered and entrapped areas using hydrodynamic modeling and GIS: Environmental Modelling & Software, v. 24, no. 12, p. 1447–1456, accessed May 3, 2020, at https://doi.org/10.1016/j.envsoft.2009.06.001.

Robertson, M.J., Pennell, C.J., Scruton, D.A., Robertson, G.J., and Brown, J.A., 2004, Effect of increased flow on the behaviour of Atlantic salmon parr in winter: Journal of Fish Biology, v. 65, no. 4, p. 1070–1079, accessed May 4, 2020, at https://doi.org/10.1111/j.0022-1112.2004.00516.x.

Saltveit, S.J., Halleraker, J.H., Arnekleiv, J.V., and Harby, A., 2001, Field experiments on stranding in juvenile Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*) during rapid flow decreases caused by hydropeaking: Regulated Rivers, v. 17, no. 4–5, p. 609–622, accessed May 4, 2020, at https://doi.org/10.1002/rrr.652.

Sauterleute, J.F., Hedger, R.D., Hauer, C., Pulg, U., Skoglund, H., Sundt-Hansen, L.E., Bakken, T.H., and Ugedal, O., 2016, Modelling the effects of stranding on the Atlantic salmon population in the Dale River, Norway: Science of the Total Environment, v. 573, p. 574–584, accessed May 4, 2020, at https://doi.org/10.1016/j.scitotenv.2016.08.080.

Scruton, D.A., Pennell, C.J., Robertson, M.J., Ollerhead, L.M.N., Clarke, K.D., Alfredsen, K., Harby, A., and McKinley, R.S., 2005, Seasonal response of juvenile Atlantic salmon to experimental hydropeaking power generation in Newfoundland, Canada: North American Journal of Fisheries Management, v. 25, no. 3, p. 964–974, accessed May 4, 2020, at https://doi.org/10.1577/M04-133.1.

Strickland, R.R., Hayes, J.W., and Boubee, J., 2002, Project Aqua—Environmental Study—Aquatic Ecosystems—Fish stranding: New Zealand, Cawthorn Institute, 41 p., accessed January 9, 2022, at https://www.worldcat.org/title/155853766.

Sykes, G., Bio, R.P., Liebe, R., and Milne, M., 2010, Middle Columbia River juvenile fish stranding assessment CLBMON-53—2011 report (year 3 of 4)—Prepared for BC Hydro: British Columbia, Revelstoke.

Thompson, J.S., 1970, The Effect of Water Flow Regulation at Gorge Dam on Stranding of Salmon Fry in the Skagit River 1969–1970: F.P.C. 553, State of Washington Department of Fisheries, Management and Research Division, 50 p, accessed January 9, 2022, at https://www.seattle.gov/light/skagit/relicensing/cs/groups/secure/@scl.skagit.team/documents/document/cm9k/ntuz/~edisp/prod553741.pdf.

Tiffan, K.F., Garland, R.D., and Rondorf, D.W., 2002, Quantifying flow-dependent changes in subyearling fall Chinook salmon rearing habitat using two-dimensional spatially explicit modeling: North American Journal of Fisheries Management, v. 22, no. 3, p. 713–726, accessed May 4, 2020, at https://doi.org/10.1577/1548-8675(2002)022<0713:QFDCIS>2.0.CO;2.

Tonolla, D., Bruder, A., and Schweizer, S., 2017, Evaluation of mitigation measures to reduce hydropeaking impacts on river ecosystems—A case study from the Swiss Alps: Science of the Total Environment, v. 574, p. 594–604, accessed May 4, 2020, at https://doi.org/10.1016/j.scitotenv.2016.09.101.

Tuhtan, J.A., Noack, M., and Wieprecht, S., 2012, Estimating stranding risk due to hydropeaking for juvenile European grayling considering river morphology: KSCE Journal of Civil Engineering, v. 16, no. 2, p. 197–206, accessed May 4, 2020, at https://doi.org/10.1007/s12205-012-0002-5.

U.S. Fish and Wildlife Service, 2006, Relationships between flow fluctuations and redd dewatering and juvenile stranding for Chinook salmon and Steelhead in the Sacramento River between Keswick Dam and Battle Creek—Prepared by the Energy Planning and Instream Flow Branch: Sacramento, Calif., U.S. Fish and Wildlife Service, 94 p.

Van Winkle, W., Jager, H.I., Railsback, S.F., Holcomb, B.D., Studley, T.K., and Baldrige, J.E., 1998, Individual-based model of sympatric populations of brown and rainbow trout for instream flow assessment—Model description and calibration: Ecological Modelling, v. 110, no. 2, p. 175–207, accessed May 4, 2020, at https://doi.org/10.1016/S0304-3800(98)00065-9.

Vanzo, D., Tancon, M., Zolezzi, G., Alfredsen, K., and Siviglia, A., 2016, A modeling approach for the quantification of fish stranding risk—The case of Lundesokna River, Norway, *in* 11th International Symposium on Ecohydraulics (ISE 2016), Melbourne, Australia, 2016, Proceedings: Melbourne, Australia, Engineers Australia, 326 p., accessed January 9, 2022, at https://search.informit.org/doi/10.3316/INFORMIT.687014727486689.

Vollset, K.W., Skoglund, H., Wiers, T., and Barlaup, B.T., 2016, Effects of hydropeaking on the spawning behaviour of Atlantic salmon *Salmo salar* and brown trout *Salmo trutta*: Journal of Fish Biology, v. 88, no. 6, p. 2236–2250, accessed May 3, 2020, at https://doi.org/10.1111/jfb.12985.

Walker, R.H., Maitland, B.M., LaSharr, T.N., Rosing, M.N., and Ben-David, M., 2018, Fate of juvenile salmonids stranded in off-channel pools—Implications for nutrient transfers: Aquatic Sciences, v. 80, article 10, 9 p., accessed May 4, 2020, at https://doi.org/10.1007/s00027-017-0562-z.

Wilson, J., and Duston, R.A., 2006, 2005 Evaluation of Chum, Chinook and Coho Salmon Entrapment near Ives Island in the Columbia River—2004–2005 Annual Report [report no. DOE/BP-00004287-4]: Bonneville Power Administration, prepared by Pacific States Marine Fisheries Commission, 61 p., accessed January 9, 2022, at https://doi.org/10.2172/897379.

Woodin, R.M., Crumley, S.C., Stober, Q.J., and Engman, G., 1984, Salmon and steelhead studies, v. 2 *of* Skagit River Interim Agreement Studies: Seattle, Wash., University of Washington, 61 p., accessed January 9, 2022, at https://www.seattle.gov/light/skagit/Relicensing/cs/groups/secure/@scl.skagit.team/documents/document/cm9k/nta1/~edisp/prod505528.pdf.

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