Technical Papers

Effects of Experimental Ramping Rate on the Invertebrate Community of a Regulated River

By Karen E. Smokorowski[1]

Abstract

In Ontario, Canada, the provincial government regulates water licenses and in recent years has required that all hydroelectric facilities prepare dam operating plans that often include some incorporation of environmental flows. Peaking facilities can be required to implement a minimum flow and (or) have restrictions imposed on ramping rates (rate of change of turbine flow in cubic meters per second per hour) without sound scientific knowledge that these restrictions benefit river health. This paper reports preliminary results from a collaborative, long-term, adaptive management experiment designed to determine if removing all existing operational constraints on ramping rates was detrimental to the downstream riverine ecology, assessed relative to an unregulated river. Invertebrate abundance, diversity, and taxa composition were measured to test the hypothesis that invertebrate communities would be negatively affected by unlimited ramping. During the restricted years, the invertebrate community had greater abundance, diversity, and proportion of sensitive taxa relative to the unregulated river. After unlimited ramping, there was evidence of negative effects on the invertebrate community, implying that the restricted operation was protective of these biota, although results should be viewed with caution because of a confounding climate effect.

Background

Canada has an abundance of freshwater resources, which consequently have been used to a large degree for social and economic benefits, including hydroelectric power generation. In Canada, approximately 60 percent of the total electricity generation is from hydroelectric sources (Canadian Electricity Association, 2006), with many unaltered watersheds holding potential for additional generation. The size of dams can range from a few meters to hundreds of meters, and the operational regime can range from "run-of-the-river" (smaller impoundments, where power generation is largely dictated by inflow volume), which is considered relatively benign, to fully "peaking" where water is released in accordance with electricity demand resulting in large hourly and daily fluctuations (Clarke and others, 2008). Relative to a natural hydrograph, peaking operations greatly alter flow regimes, which have been shown to lead to altered temperature patterns and geomorphology (sediment and physical channel characteristics), reduced habitat diversity, organism physiological stress, and consequently reduced abundance, diversity, and productivity of biota (Cushman, 1985; Richter and others, 1997; Bunn and Arthington, 2002; Sabater, 2008).

Environmental flows (flows prescribed for the benefit of river ecosystem health) traditionally considered only minimum flow levels, but have recently evolved to consider all elements of the flow regime (including magnitude, duration, timing, frequency, and rate of change of flow), largely because of the increasing interest in the importance of natural flows or the natural flow paradigm (NFP; Poff and others, 1997). The NFP theory states that organisms have adapted to the range in variations inherent to natural flows, and that the ecosystem integrity (health) of a river relies on maintaining natural variability (Poff and others, 1997; Richter and others, 2003). Unfortunately, however, it is difficult to run an efficient and profitable hydroelectric dam under the tenets of the NFP, although compromises do potentially exist (Enders and others, 2009).

In Canada, the provincial Ontario Ministry of Natural Resources (OMNR) controls water licensing and now requires that all hydroelectricity producers in the province develop dam operating plans that set operational requirements for management of water flows and levels that are enforceable by law. Often, peaking hydro dams are required to implement a minimum flow regime, but recently some dams have had restrictions imposed on ramping rates (the rate of change of flow passing through the turbines in cubic meters per second per hour, or $m^3 \cdot s^{-1} \cdot h^{-1}$). Ramping rate restrictions mean that peaking dams can, to a degree, still follow the demand in electricity, but at a slower rate, thus reducing magnitude of change, reducing response times, passing excess water, and lowering the facility efficiency (here termed "modified peaking"). However, with the exception of fish stranding

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studies (e.g., Bradford, 1997; Saltveit and others, 2001; Irvine and others, 2009), there is little evidence in the scientific literature that supports the belief that ramping rate restrictions (while systems continue to peak as able, given restrictions) benefit riverine ecology, and direct experimentation is needed.

In order to reduce scientific uncertainties about the effects of ramping rates, Fisheries and Oceans Canada, the OMNR, Brookfield Renewable Power, Inc., and the University of Waterloo are collaborating on a long-term, adaptive management experiment to test whether regulating ramping rates through hydroelectric turbines can provide ecological benefits, while at the same time minimizing production losses. The main purpose of this adaptive management experiment is to determine if removing all operational constraints on ramping rates from a hydroelectric facility that has operated under restricted ramping rates and minimum flows since its initial operation in the early 1990s is detrimental to the downstream riverine ecology.

Benthic Invertebrates as Test Organisms

Macroinvertebrates have long been used as bioindicators for human disturbance because of their widely varying sensitivity to perturbation, short growth rates and generation time (allowing detection of responses to change), and ability to disperse and recolonize disturbed areas (Hodkinson and Jackson, 2005). Invertebrates have been shown to be sensitive to the negative effects of peaking hydroelectric dams and are, therefore, good test subjects for experimental flows. Frequent and rapid fluctuations in flow can contribute to the decrease in macroinvertebrate abundance and diversity in areas close to the dam (Cushman, 1985; Growns and Growns, 2001), with the shifts in species composition observed for kilometers downstream (Céréghino and others, 2002). While periphyton and macroinvertebrates in the varial zone of a peaking river were found to be impaired in terms of density and diversity and were largely represented by tolerant taxa (Fisher and LaVoy, 1972; Blinn and others, 1995; Benenati and others, 1998), invertebrates found in the permanently wetted zone of a “modified peaking” river may experience more favorable environmental conditions because of the lack of rapid change in shear stress (stress of water flow on the river bed that can cause the substrate to move and (or) dislodge material on the river bed) caused by restricted ramping. For example, Parasiewicz and others (1998) introduced a flow constraint that imposed a minimum base flow and reduced peak flows on a regulated river. The result was that invertebrate biomass was found to increase by 60 percent, which the authors attributed to reduced scouring of the substrate during the bed filling (up-ramping) stage (Parasiewicz and others, 1998). This experiment was intended to test the hypothesis that, relative to an unregulated river, invertebrates in the permanently wetted zone would benefit under a restricted ramping rate regime plus the maintenance of a minimum flow (constrained operation), but would respond negatively (via reduced abundance and diversity) to unlimited ramping because of the resulting increased instability (i.e., changing depth and velocity, increased bedload movement) in habitat.

Study Design

We used a before-after-control-impact (BACI) design for this experiment, which in this case involves comparing conditions on a river regulated for peaking hydroelectric power production (impact river) to conditions on an unregulated reference (control) river (i.e., without any hydroelectric dams) before and after implementing a change in ramping rates. This approach should allow detection of a change in invertebrate measures (abundance and diversity) that were caused by the experimental ramping rate changes, since the control river should reflect the influence of temporal changes in regional environmental factors. The experimental site was the Magpie River, Wawa, Ontario, (48°0´N; 84°7´W) on the 40 kilometer (km) stretch between Steephill Falls and the Harris waterpower facilities (WPF) (fig. 1). The reference river was the unregulated Batchawana River (47°0´N; 84°3´W), located approximately 60 km north of Sault Ste. Marie, Ontario. Between 2002 and 2004, data were collected from the regulated Magpie River under the original restricted ramping rate regime: ramping rate could not exceed 1 $m^3 \cdot s^{-1} \cdot h^{-1}$ from October 10 to November 15; 2 $m^3 \cdot s^{-1} \cdot h^{-1}$ from November 16 until spring freshet (early May); from May until early October, the dam was restricted to an increase or decrease of 25 percent of the previous hour’s flow. From 2005 to 2007, data were collected with no restrictions on ramping and while the Steephill Falls plant operated in accordance with water availability and market forces (fig. 2). During the entire study period, through all seasons, the Steephill Falls WPF could not release a discharge lower than 7.5 $m^3 \cdot s^{-1}$, which was the regulated minimum flow. All sampling on the Batchawana River was done contemporaneously.

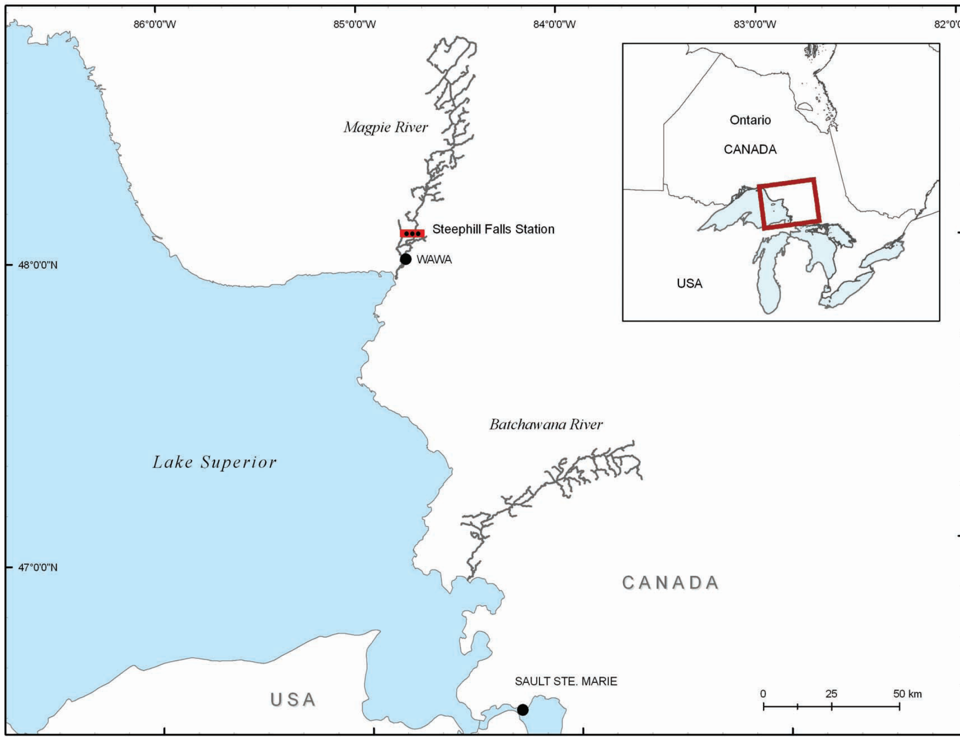

Figure 1. Map showing location of the Magpie and Batchawana Rivers relative to Lake Superior and Sault Ste. Marie, Ontario.

Methods

To assess the benthic invertebrate community, six sites were chosen on the Magpie River, one above the dam outside of the zone of influence and five downstream at distances 2.5, 3, 6, 9.5, and 10.5 km from the dam. The six sites on the Batchawana River were selected to be spatially separated in a similar fashion assuming a hypothetical dam at a point on the river. In each year at each site, five mesh rock bags were randomly placed in a riffle, ensuring a minimum distance of 3 meters (m) apart, and at a depth to maintain a sufficient flow over the bags throughout low-water periods. The rock bags were constructed out of 2-inch net mesh, 48 inches in circumference and 18 inches in length, and were filled with rocks of representative size found along the shoreline at the site of placement until each reached a weight of 7 kilograms (+/– 0.5 kg).The actual number of rocks used, their diameter, and weight of each bag was recorded, as were the depth and velocity (Marsh McBirney Flomate 2000 Portable Flow Meter) in the river at each bag. The bags were left in the river for a period of approximately 60 days (June–August), a sufficient length of time for full colonization to reach fluctuating taxa richness, abundance, and biomass (Mason and others, 1973; Shaw and Minshall, 1980). Once bags were retrieved, the rocks were cleaned and all invertebrates and debris were preserved in 70-percent ethanol. The entire sample was subsampled for identification to taxonomic level of family and enumeration, although in each year a number of samples were identified in their entirety to allow for the calculation of accuracy and precision of subsampling procedure, which were always found to be within acceptable limits (defined as being within 20 percent of true counts, Elliott, 1977).

Invertebrate families were then used to calculate invertebrate diversity (probability of interspecific encounter, PIE; Hurlbert, 1971) and percentage of Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) (%EPT). PIE is an unbiased diversity measure that calculates the chance that two individuals drawn at random from a population represent different families:

$$PIE = \sum_{i=1}^{s} (n_i / n)\left[(n - n_i)/(n-1)\right] \quad (1)$$

where: n = number of all individuals in the sample, n_i = number of individuals of a family in the sample, and s = number of families (Hurlbert, 1971). PIE was selected over other diversity indices because it provides a statistically and biologically understandable probability (out of 100 percent, the higher the number the more diverse the community), unlike more traditional diversity measures (Gottelli and Graves, 1996). The %EPT calculations were completed by summing the number of individuals within the three families and dividing by the total number of individuals in all invertebrate families found in the samples. These three taxa are known to be sensitive to changes in water quality and flow (Mackie, 2004), and a high percentage of EPT signifies a healthy invertebrate community.

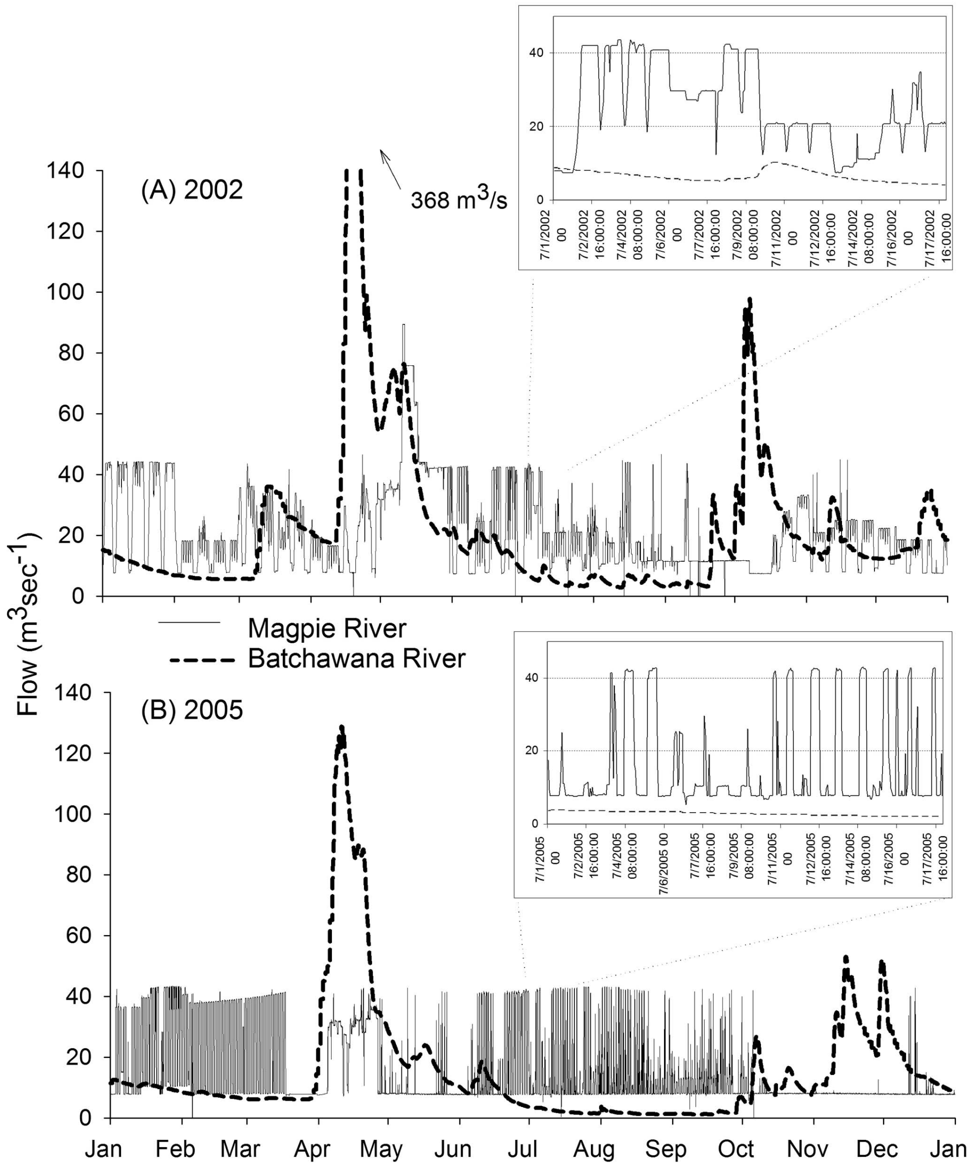

Figure 2. Annual hydrograph of the Magpie (solid lines) and Batchawana (dotted lines) Rivers, (A) in 2002, before ramping change on the Magpie, and (B) in 2005, after unlimited ramping on the Magpie River. Data for the Magpie River from the Steephill Falls waterpower facility (courtesy Brookfield Renewable Power, Inc). (Data for the Batchawana River courtesy of the Water Survey of Canada, Environment Canada.)

Invertebrate abundance, diversity, and %EPT were averaged across all sites and plotted against year for each river. A statistical test (2-way analysis of variance (ANOVA), river by year) was used to determine if there was a significant difference between rivers or years or if the difference between rivers changed through the years (called the interaction term of "river by year"). To simplify the comparison between the years of restricted and unlimited ramping rates, the BACI design was used in a statistical test (2-way ANOVA, treatment by time). In our BACI design, the sites on the Batchawana River plus the one site above the dam outside of the zone of influence of the dam was classified as the "control" treatment, and the sites on the Magpie River downstream from the dam were classified as the "impact" treatment. The years 2002–2004 were classified as the "before" time, and the years 2005–2007 were classified as the "after" time.

For a BACI ANOVA, the statistic of interest is the interaction term (treatment by time), which will be significant if lines defining the differences in before-after samples among rivers cross (or are unparallel to a significant degree). If the lines cross, then the difference between control and impact changes from before to after the treatment, and we can say with some confidence that the change was because of the unlimited ramping. For all statistical tests, p-value of less than 0.05 means that there was a less than 5-percent chance that the difference found was because of chance, and therefore the difference can be considered significant.

Results

It is clear to see in figure 2 that the natural flow of the Batchawana River resulted in much greater peak flows and lower minimum flows relative to the altered Magpie River. In 2002, when ramping rate was restricted, the dam operated on a reduced peaking cycle, "perched" on an elevated minimum during the week (when water supply was high), or did not reach full turbine flow (when water levels were low), and dropped to the minimum flow on weekends (if demand was low). However, in 2005, full ramping from the maximum turbine discharge to minimum regulated flow occurred at a much greater frequency because the speed of change was unrestricted. During the restricted ramping phase between 2002 and 2004, the Magpie River had a significantly greater abundance of invertebrates than the Batchawana River (fig. 3A). After the experimental change to unlimited ramping occurred (2005–2007), however, the Magpie River invertebrate abundance decreased while the Batchawana River invertebrate abundance stayed essentially the same. The change in the difference between the two rivers was enough for the interaction term in the statistical test to be significant, meaning the decrease in the Magpie was much greater than any change on the Batchawana River (fig. 3B).

Similar to the abundance results, our invertebrate diversity PIE and %EPT measurements were both significantly greater on the Magpie River compared to the Batchawana River during the limited ramping period (fig. 4A and C). However, contrary to the abundance results, these measurements increased on the Batchawana River during 2005–2007 while they decreased on the Magpie River, so that they were actually greater on the control river after the change to unlimited ramping (fig. 4B and D).

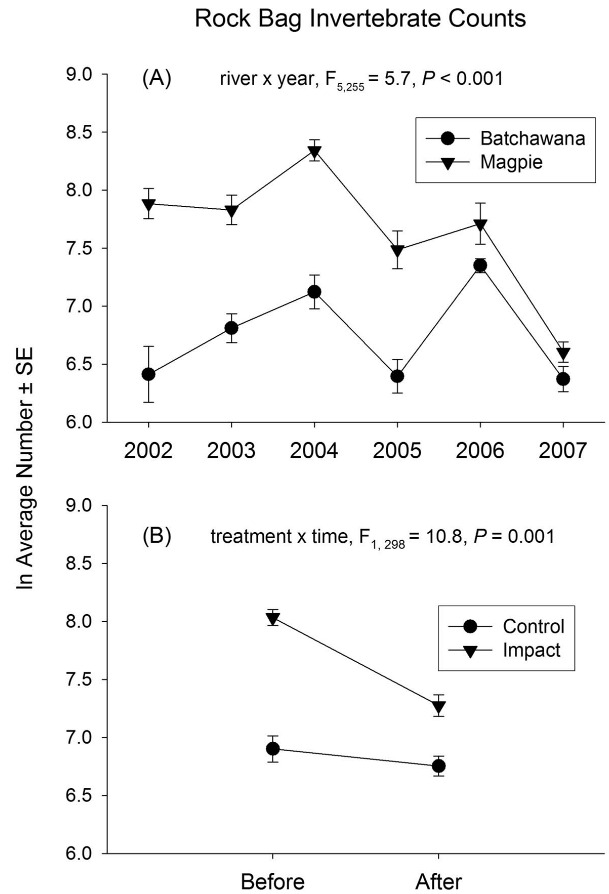

Figure 3. Average abundance (log + 1 transformed) of invertebrates per rock bag ± standard error (SE) plotted as (A) average across sites for each year, and (B) as the before-after-control-impact plot.

Discussion

During the period of constrained ramping rate, although the hydrograph of the Magpie River was still considerably altered relative to a natural flow regime, the invertebrate community remained healthy in terms of abundance, diversity, and proportion of sensitive taxa relative to the unregulated river. Yet once the operation of the waterpower facility was unconstrained (unlimited ramping, maintained minimum flow), there was evidence of negative effects on the invertebrate community, implying that the restricted operation was protective of these biota. Without the experimental change in flow regime to unlimited ramping rate, it would have been unclear whether the minimum flow or ramping rate was of greater benefit.

The maintenance of a minimum flow has been shown to be important for the protection of river ecosystems, including invertebrates, below hydroelectric facilities. For example, Bednarek and Hart (2005) found a significantly improved invertebrate family richness and proportion of intolerant taxa (%EPT) below dams that implemented a minimum flow regime and increased dissolved oxygen concentrations. The natural flow regime of the Batchawana River allowed minimum summer flows to drop considerably lower than the Magpie River, which could have resulted in elevated peak summer temperatures (Sinokrot and Gulliver, 2000) and cause stress to biota. It is likely that the combination of a minimum flow improving invertebrate habitat conditions mid-summer and restricted ramping alleviating shear stress and bedload movement on the Magpie River allowed the invertebrate community to proliferate relative to the unregulated river during the phase of constrained operations.

The onset of unlimited ramping resulted in decreased invertebrate abundance, diversity, and proportion of sensitive taxa relative to the unaltered Batchawana River. There are a number of potential reasons why unlimited ramping may

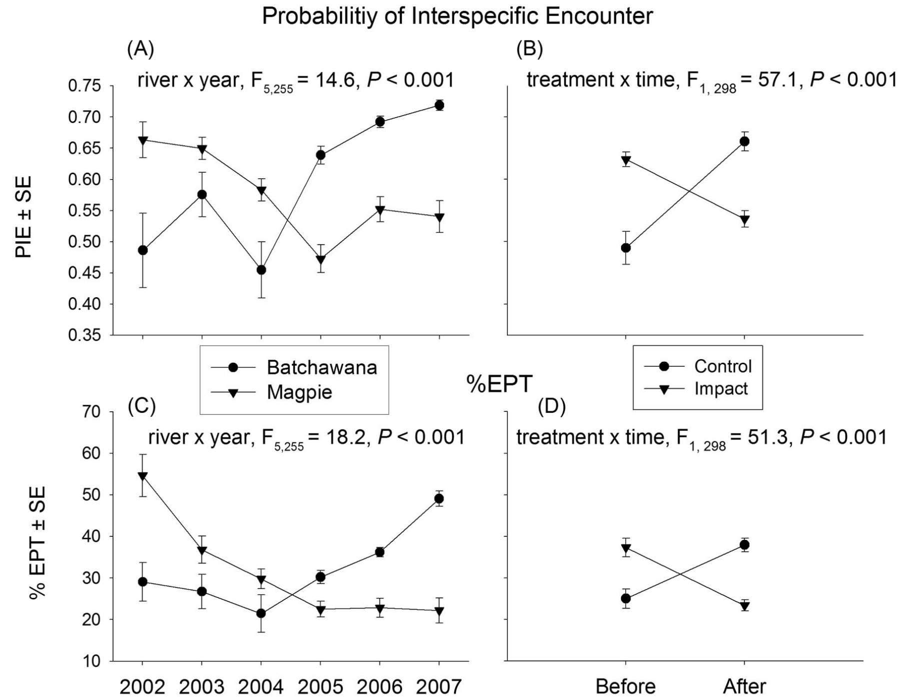

Figure 4. Average diversity (PIE) of invertebrates per rock bag ± standard error (SE) plotted as (A) average across sites for each year, and (B) as the before-after-control-impact plot. Average % EPT invertebrates per rock bag ± standard error (SE) plotted as (C) average across sites for each year, and (D) as the before-after-control-impact plot.

be considered detrimental to aquatic invertebrates, the most probable candidates including stranding, flushing (catastrophic drift), and rapid and extreme temperature fluctuations. Stranding refers to the separation of an organism from the flowing surface water caused by the rapid decrease in flows, resulting in isolation in pools, side channels, or desiccation on formerly wet substrate. During experimental flows, a greater number of insects were found stranded when the rate of decrease in flow was rapid (Perry and Perry, 1986), implicating unlimited down ramping as a potential cause for increased invertebrate mortality. Because invertebrates are continually moving and drifting to different positions in the river, stranding a significant number of invertebrates in the varial zone would reduce the overall abundance in the river including those in the permanently wetted zone. Rapid increases in flow could result in rapid increases in shear stress, potentially causing catastrophic drift, or the large scale displacement of invertebrates from the sediment during increases in river discharge (Gibbins and others, 2007). While these displaced invertebrates may be able to recolonize the riverbed further downstream, they are more vulnerable to predation by fish while drifting. Finally, rapid and frequent changes in flow below a peaking hydroelectric dam are often accompanied by rapid fluctuations in water temperature (Cushman, 1985), which can be highly stressful, if not lethal, to organisms (Stanford and Hauer, 1992). All of these potential negative consequences of unlimited ramping could be more detrimental to sensitive taxa (i.e., EPT) than tolerant taxa, leading to the increased dominance of tolerant species and reduced diversity.

In 2005, when the rate of change of flow occurred as rapidly and frequently as the electricity market and water availability dictated, the Steephill Falls waterpower facility was still required to maintain a minimum flow below the dam. Therefore, any negative effects detected on the invertebrate community between 2005 and 2007 should have been clearly attributable to unlimited ramping. Unfortunately, however, there was a confounding factor affecting our ability to definitively implicate the change in ramping rate as the causative factor. With the change to unlimited ramping in the fall of 2004, the region experienced the onset of a 3-year drought, confounding the clarity of our results (fig. 5). The drought resulted in above-average temperatures and lower-than-normal flows on all rivers, including the reference river, and the ability of the Steephill Falls reservoir on the Magpie River to store the complete spring freshet, which reduced the magnitude and frequency of ramping relative to a normal water-level year. A spring freshet, although reduced, still occurred on the reference river, and the importance of the complete loss of the freshet on the Magpie River is unclear. Therefore, any results need to be viewed with some caution as the study is ongoing to attempt to clarify causation: are observed effects the result of changes in ramping or drying conditions?

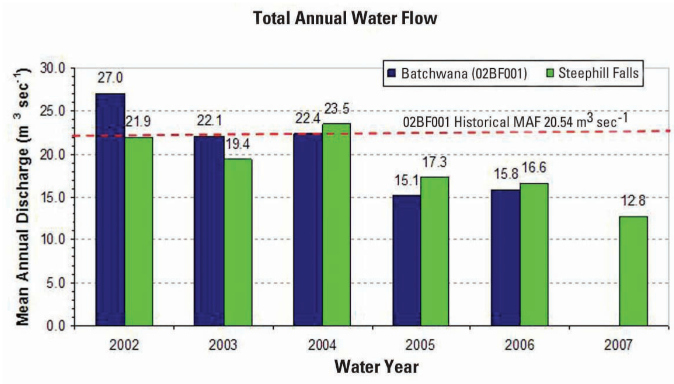

Figure 5. Total annual flow ($m^3 \cdot s^{-1}$) on the Magpie and Batchawana Rivers both before and after unlimited ramping was implemented in 2005. The red dotted line indicates the mean annual flow for the Batchawana River as calculated from historical water survey of Canada data.

Implications for Management

This research project constitutes a significant undertaking, and establishing cooperative partnerships and shared financial support among all partners was essential to success. Many challenges were encountered, including sampling methodology difficulties specific to working on peaking systems. Subsequent field method refinement resulted in an important methodological contribution to future research and monitoring of peaking hydrofacilities in the form of standardized sampling protocols. Other challenges include the modification and fine tuning of data exploration and analyses to best understand stressors and effects and the challenge of unpredictable climate changes.

Results of this and ongoing studies will help inform Canadian provincial and Federal waterpower guidelines and policy, facilitating science-based decisions regarding ramping at hydrofacilities. In addition, methodologies developed will be used to help establish effectiveness monitoring programs for dam operating plans at existing and new hydrofacilities in Ontario. This project generated several successes, including cooperative management, field and data-sharing partnerships, assurance of independent scientific integrity through the design team structure, and development of standardized protocols across a suite of ecosystem measures (including hydrology, geomorphology, invertebrates, fish, and food web) that show a response to subtle flow changes. It is anticipated that these successes will serve as a model for future collaborations to address large-scale, long-term, and complex ecological questions related to resource management.

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Projecting Temperature in Lake Powell and the Glen Canyon Dam Tailrace

By Nicholas T. Williams[1]

Abstract

Recent drought in the Colorado River Basin reduced water levels in Lake Powell nearly 150 feet between 1999 and 2005. This resulted in warmer discharges from Glen Canyon Dam than have been observed since initial filling of Lake Powell. Water quality of the discharge also varied from historical observations as concentrations of dissolved oxygen dropped to levels previously unobserved. These changes generated a need, from operational and biological resource standpoints, to provide projections of discharge temperature and water quality throughout the year for Lake Powell and Glen Canyon Dam. Projections of temperature during the year 2008 were done using a two-dimensional hydrodynamic and water-quality model of Lake Powell. The projections were based on the hydrological forecast for the Colorado River Basin and initial conditions from limnological field surveys. Results from the projection simulations are presented and compared with 2008 field observations. Post-simulation comparisons of projected results with field data were done to assess the accuracy of projection simulations.

Introduction

Drought in the Colorado River Basin from 1999 to 2005 greatly reduced the inflow to Lake Powell and brought about changes to temperature and water quality of the river below the dam. Reservoir elevations steadily dropped to an elevation of 3,555 feet in April 2005, just before the snowmelt runoff of that year. The powerplant intakes, which were then only 85 feet below the reservoir water surface, withdrew warmer water from the reservoir, and river temperatures below the dam peaked at 61 degrees Fahrenheit (°F) (16 degrees Celsius; °C) in October 2005 (fig. 1). While it was expected that temperatures in the river below the dam would warm with decreasing reservoir elevations, it was not the only factor contributing to warmer temperatures. Spring runoff volume and the local climate were also significant factors affecting the magnitude of warming in dam discharges (Bureau of Reclamation, 2007).

During the period of warmest river temperatures, the dissolved oxygen content of discharges from the dam declined to concentrations lower than any previously observed (fig. 1). Operations at Glen Canyon Dam were modified by running turbines at varying speeds, which artificially increased the dissolved oxygen content of discharges; however, these changes also resulted in decreased power generation and possibly damaged the turbines (Bureau of Reclamation, 2005). The processes in the reservoir creating the low dissolved oxygen content in the reservoir had been observed in previous years, but before 2005 the processes had never affected the river below the dam to this magnitude (Vernieu and others, 2005). As with the warmer temperatures, the low dissolved oxygen concentrations could not be explained solely by the reduced reservoir elevations. Other contributing factors include interactions with exposed sediment delta and spring runoff volume (Wildman, 2009).

The low dissolved oxygen content of Glen Canyon Dam discharges during 2005 resulted in increased efforts to provide better information on potential water-quality issues in the reservoir and on changes to temperature or water quality of

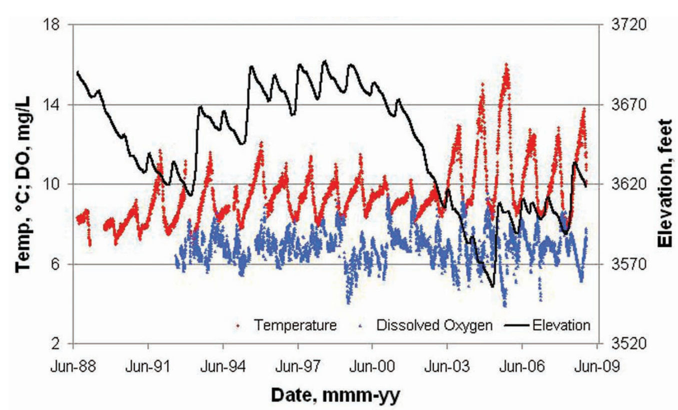

Figure 1. Daily water temperature and dissolved oxygen concentration below Glen Canyon Dam with Lake Powell water-surface elevations, 1988–2008 (adapted from Vernieu and others, 2005).

[1] Bureau of Reclamation, Upper Colorado Region, Water Quality Group, 125 South State Street, Salt Lake City, UT 84138.

dam discharges. Studying the conditions of the drought and reservoir processes has increased understanding of the causes of warmer temperatures and other water-quality changes in discharges from the dam (Vernieu and others, 2005; Williams, 2007; Wildman, 2009). Quarterly lake-wide monitoring of the reservoir provides information about conditions in the reservoir in advance of such events, but projecting the routing of water through the dam to the river below is difficult to determine from the reservoir monitoring data alone. A computer model has been developed and tested to reproduce historical hydrodynamics and water-quality characteristics of Lake Powell and the discharges from Glen Canyon Dam (Williams, 2007). Using this model in combination with monitoring data and hydrological forecasts allows for projection simulations of temperature in and below Lake Powell several months in advance. The objective of this paper was to use the existing model and develop methods for simulating reservoir and dam discharge temperatures that can be replicated for repeated simulations at later dates.

Glen Canyon Dam and Lake Powell

Glen Canyon Dam is located in north-central Arizona just south of the Utah-Arizona border near the town of Page, Arizona (fig. 2). The dam was constructed between 1957 and 1964 and formed the reservoir known as Lake Powell. At full capacity the lake's elevation is 3,700 feet, the volume is 26.5 million acre-feet, and its deepest point is more than 500 feet. Water is released from the dam through the hydroelectric powerplant. The intake for the powerplant is at elevation 3,470 feet, 230 feet below the water surface of a full reservoir. The large lake and deep intake for the powerplant altered the temperature and water quality of the Colorado River below the dam. Large seasonal fluctuations from 32 °F to 80 °F (0 °C to 27 °C) in river temperatures were replaced with temperatures ranging from 44 °F to 54 °F (7 °C to 12 °C) after the reservoir filled and stayed within this range while reservoir water-surface elevations were maintained above approximately 3,600 feet (Vernieu and others, 2005).

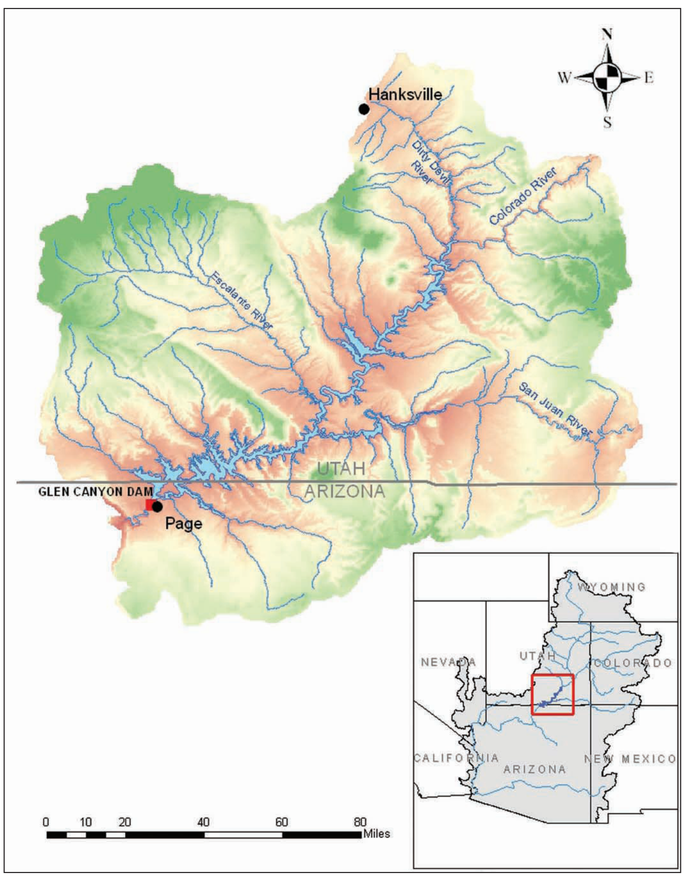

Figure 2. Lake Powell and immediate watershed showing location of Glen Canyon Dam; inset shows the location of Lake Powell in reference to the Colorado River Basin

Methods

Hydrologists and meteorologists develop forecasts to project runoff and weather that are intended to be an educated guess at what the future might bring. These forecasts are based on current conditions and assumptions of future conditions. Forecasts are not 100 percent accurate in their predictions, but the information they provide is still useful for planning purposes. Similarly, current conditions in Lake Powell and assumptions about future inputs to the reservoir during 2008 were simulated in a model to project characteristics of Lake Powell and the Colorado River below Glen Canyon Dam. Detailed results from the simulations were used to support quarterly monitoring and provide information for dam operations and resource management.

Hydrodynamic and Water-Quality Model

Temperature in and below Lake Powell is simulated using the two-dimensional hydrodynamic and water-quality model, CE-QUAL-W2, version 3.2 (Cole and Wells, 2003). CE-QUAL-W2 was developed by the U.S. Army Corps of Engineers and Portland State University and has evolved over three decades. It assumes lateral homogeneity and is ideal for long, narrow waterbodies such as Lake Powell. The model is capable of predicting water-surface elevations, velocities, temperatures, and a number of water-quality constituents. Water is routed through cells in a computational grid where each cell acts as a completely mixed reactor in each time step. Geometrically complex waterbodies are represented through multiple branches and cells. Multiple inflows and outflows are represented through point/nonpoint sources, branches, precipitation, and other methods. Output from the model provides options for detailed and convenient analyses.

Lake Powell Model Description

The Lake Powell CE-QUAL-W2 model was developed and tested by the Bureau of Reclamation (Reclamation), Upper Colorado Regional Office (Williams, 2007). The particular model discussed here simulated hydrodynamics, temperature, salinity, dissolved oxygen, phytoplankton, and organic matter decay in Lake Powell from January 1990 through December 2005. It is hereafter referred to as the calibration model so as to distinguish it from the projection simulation models of Lake Powell. The calibration model uses a geometric, computational grid and various input data to simulate these processes. The model computational grid, inputs, and calibration process and results are briefly discussed in the sections below.

Lake Powell Bathymetry

The bathymetry of Lake Powell is represented in the CE-QUAL-W2 model as a two-dimensional computational grid. The two dimensions represented are the length and depth, which are divided into longitudinal segments and vertical layers. The lateral dimension, or width, is not represented in the grid, but an average width is computed and used to determine volume. Because the model grid is two-dimensional, all modeled parameters, such as temperature, velocity, and water-quality constituents, can only vary in the longitudinal and vertical directions. This assumes that modeled parameters do not vary significantly in the lateral direction, and this assumption is considered appropriate for Lake Powell.

The Lake Powell CE-QUAL-W2 computational grid consists of nine branches that represent the following channels and bays: Colorado River or main channel, Bullfrog and Halls Creek Bay, Escalante River channel, San Juan River channel, Rock Creek Bay, Last Chance Bay, Warm Creek Bay, Navajo Canyon, and Wahweap Bay (fig. 3). The nine branches are further subdivided into 90 segments between 800 and 17,000 meters in length. Each segment consists of up to 97 layers, which are each 1.75 meters in height. Figure 4 is an image of the computational grid showing a plan view of the entire reservoir, a side view of the segment above Glen Canyon Dam, and a profile view of the Colorado River or main channel. In the computational grid, the color green indicates the upstream segment of a branch, blue indicates the downstream segment of a branch, and red indicates the segment where one branch connects to another branch.

Model Inputs

Model inputs are time sequences of data that describe meteorological conditions, inflows, outflows, and water-quality parameters at Lake Powell. The time sequence inputs also provide the model boundary conditions. Meteorological data measured and recorded at the Page Municipal Airport were obtained through the National Climate Data Center (NCDC). Inflow records for all gaged tributaries of Lake Powell were obtained from the U.S. Geological Survey (USGS) National Water Information System (NWIS) for the Colorado River. The number and location description of these stream sites are presented in table 1. For inflows where little or no data are available, estimates are made to approximate base flows. Data for outflow from Lake Powell through Glen Canyon Dam were obtained from historical reservoir data recorded by Reclamation. Water-quality data for tributary inflows, including temperature, total dissolved solids (TDS), dissolved oxygen, and nutrients, were obtained from measurements collected by several different agencies, including USGS, Reclamation, and the Utah Division of Water Quality (Utah DWQ).

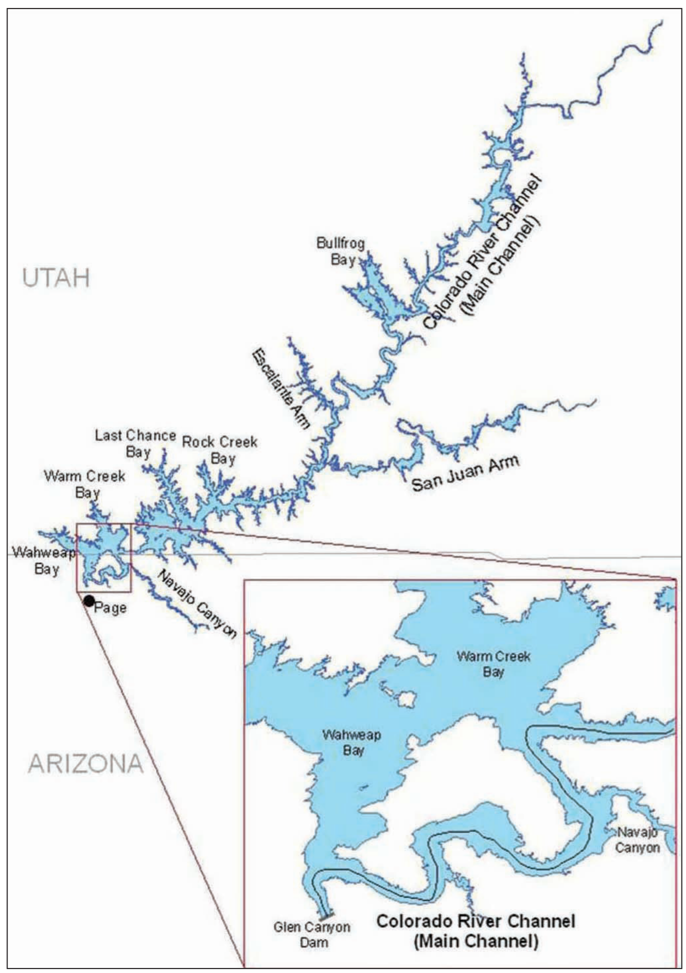

Figure 3. Lake Powell channels and bays.

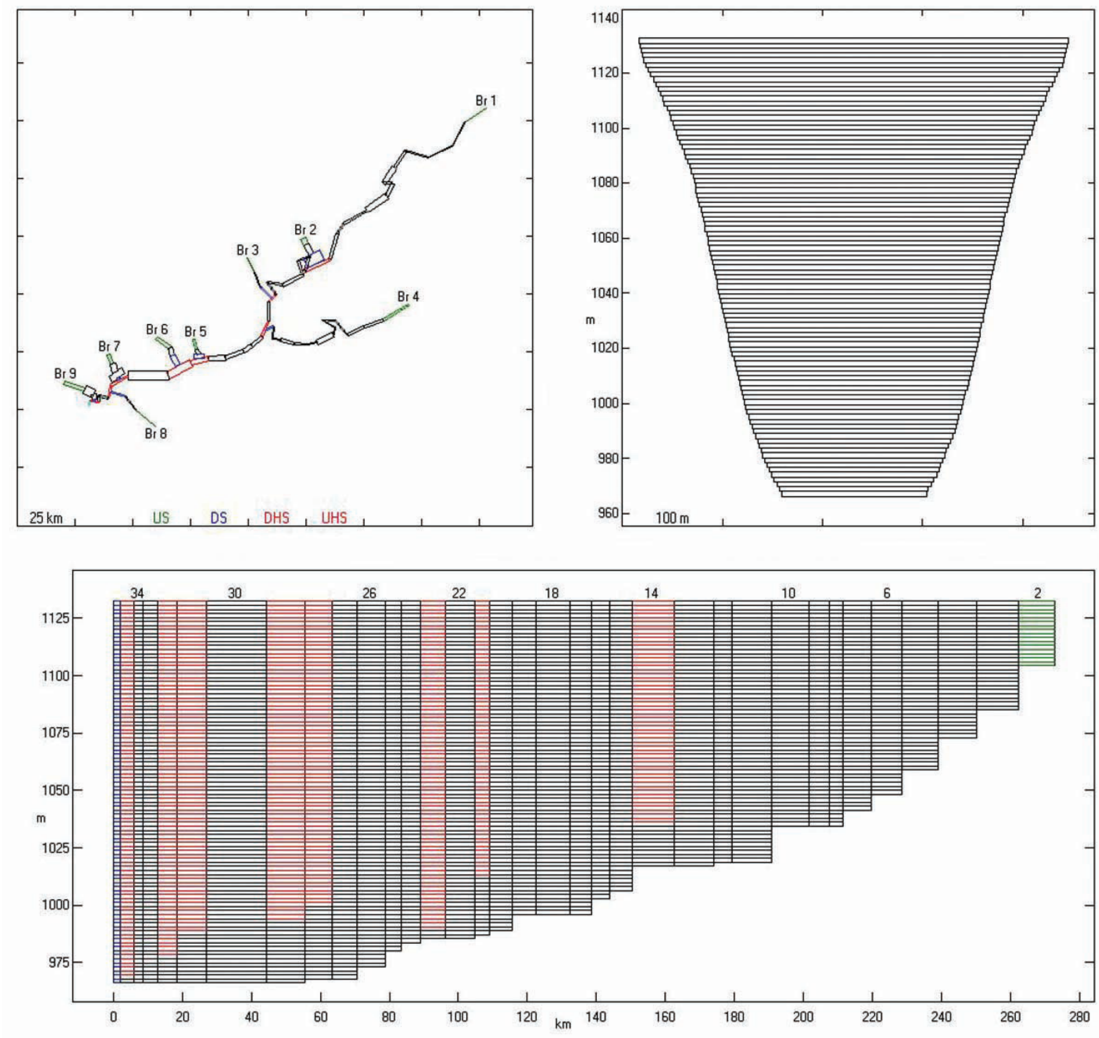

Figure 4. Lake Powell computational grid displaying plan, side, and profile views of the grid.

Table 1. U.S. Geological Survey (USGS) National Water Information System streamgages on tributaries of Lake Powell.

USGS streamgage number	Location description
09180500	Colorado River near Cisco, UT
09315000	Green River at Green River, UT
09328500	San Rafael River near Green River, UT
09379500	San Juan River near Bluff, UT
09333500	Dirty Devil River above Poison Springs Wash near Hanksville, UT

Calibration

The calibration model was calibrated for the historical period, 1990–2005, by comparing field observations of reservoir water-surface elevation (WSE), temperature, TDS, and dissolved oxygen with simulated model results. The quality of model calibration was measured by using the absolute mean error (AME) statistic (eq. 1). Model calibration statistics are presented in table 2 for the reservoir and in table 3 for the dam discharge. The mean of discharge temperatures and TDS are also presented in table 3. Statistics of dissolved oxygen concentration for dam discharges are not included because power generation increases dissolved oxygen in the river below the dam slightly depending on several factors (Williams, 2007). The model does not account for those factors; therefore, a comparison of dissolved oxygen content of the discharge with model results would not reflect actual processes.

$$AME = \frac{\sum \left| Predicted - Observed \right|}{NumberofObservations} \qquad (1)$$

Projection Model

Four projection simulations were run during 2008, which simulated reservoir and discharge temperatures. The projection simulation models were based on the calibration model, meaning kinetic coefficients and parameters determined by the 1990–2005 calibration were used in the projection simulation.

Table 2. Lake Powell CE-QUAL-W2 model, reservoir calibration statistics, 1990–2005 (Williams, 2007).

[m, meters; °C, degrees Celsius; mg/L, milligrams per liter]

Parameter	Absolute mean error
Water-surface elevation	0.08 m
Temperature	0.74 °C
Total dissolved solids	31.3 mg/L
Dissolved oxygen	1.09 mg/L

Table 3. Lake Powell CE-QUAL-W2 model, dam discharge calibration statistics, 1990–2005 (Williams, 2007).

[°C, degrees Celsius; mg/L, milligrams per liter]

Parameter	Mean		Absolute mean error
	Observed	Modeled	
Temperature	9.69 °C	9.22 °C	0.46 °C
Total dissolved solids	501 mg/L	492 mg/L	16.1 mg/L

The first step in setting up the projection simulations was determining the model simulation period. The starting date of model simulation was determined by the quarterly lake-wide monitoring surveys that provided data for the model initial conditions. The ending date of all simulations was December 31, 2008. Next, input data were added to the model. The inputs included reservoir initial conditions; forecasted hydrology, including inflows and outflows; meteorology; inflow temperatures; and water quality.

Reservoir initial conditions were obtained from quarterly lake-wide monitoring surveys conducted by the USGS Grand Canyon Monitoring and Research Center (GCMRC). Surveys used for initial conditions were conducted from February 26 to March 2, 2008, and from June 14 to June 18, 2008. During the surveys, data were collected for physical, chemical, and biologic characteristics of the reservoir at more than two dozen locations throughout the reservoir. The temperature, TDS, and dissolved oxygen data collected during the surveys were used as reservoir initial conditions and were interpolated across the model computational grid to create the input for the model.

Next, reservoir inflows and outflows for the projected period of time were obtained from the 24-Month Study reports (Bureau of Reclamation, 2009) that are hydrological forecasts of inflows to and operations of major reservoirs in the Colorado River Basin for a period of 24 months beginning with the month the report was issued. The reports provide monthly projections of Lake Powell inflow, outflow, and water-surface elevations. Inflow and outflow data in the reports are given as monthly volumes in acre-feet. Elevation data are given as end-of-month elevations in feet. The 24-Month Study reports provided total monthly inflow, but the Lake Powell projection simulation models require that the total inflow volume be allocated among the major tributaries. The allocation to the major tributaries was based on historical average ratios of tributary inflow to total reservoir inflow, which were 79 percent for the Colorado River, 13 percent for the San Juan River, <1 percent for the Dirty Devil River, and 2 percent for ungaged inflows.

Meteorological data required by the model include air and dewpoint temperature, wind speed and direction, and cloud cover recorded at the Page Municipal Airport. Typically hourly or sub-hourly observations of these parameters are used, but detailed forecasts of meteorology were not available; therefore, an hourly average of meteorological data for 1990–2005 from the Page Municipal Airport was used for the corresponding model simulation dates and times.

The inflow temperature and water-quality inputs to the projection simulations were developed from empirical and statistical relations. The program W2Met, developed by Environmental Resource Management, Inc. (ERM), was used to develop inflow temperatures on the basis of meteorological inputs (E.M. Buchak and others, ERM Group, Inc., unpub. data, 2004). The same method was used to derive the inflow temperatures for the calibration model of Lake Powell (Williams,

Table 4. Projection simulation name and associated dates for the 24-Month Study report, starting date, lake-wide survey, period of observed data input to the model, and period of projected data input to the model.

Projection name	24-Month Study	Model starting date	Lake-wide survey used for initial conditions	Period of actual data inputs	Period of projected data inputs
April 2008	April 2008	2/29/2008	February/March 2008	2/29/2008 to 4/15/2008	4/16/2008 to 12/31/2008
June 2008	June 2008	2/29/2008	February/March 2008	2/29/2008 to 6/4/2008	6/5/2008 to 12/31/2008
July 2008	July 2008	6/16/2008	June 2008	6/16/2008 to 7/28/2008	7/29/2008 to 12/31/2008
October 2008	October 2008	6/16/2008	June 2008	6/16/2008 to 10/16/2008	10/17/2008 to 12/31/2008

2007). Inflow TDS was developed from power regressions with streamflow for the major tributaries to Lake Powell (Liebermann and others, 1987). The dissolved oxygen content of the inflows was assumed to be at saturation levels based on data collected by the USGS (Williams, 2007). Other water-quality inputs to the model were developed similar to the inputs of the calibration model (Williams, 2007).

Four projection simulations were run during the spring and summer of 2008. These simulations are referred to as the April, June, July, and October 2008 projection simulations and are named on the basis of the month from which the 24-Month Study data were used (table 4). For example, the April 2008 projection simulation used hydrological forecast data from the April 24-Month Study. The model starting date of each projection simulation depended on the initial condition data collected during the quarterly lake-wide surveys. Each simulation had a period of time between the model starting date and the actual calendar day when the model was executed. During this period, observed data for inflow, outflow, and meteorology, rather than forecasted or average data, were used for the model inputs.

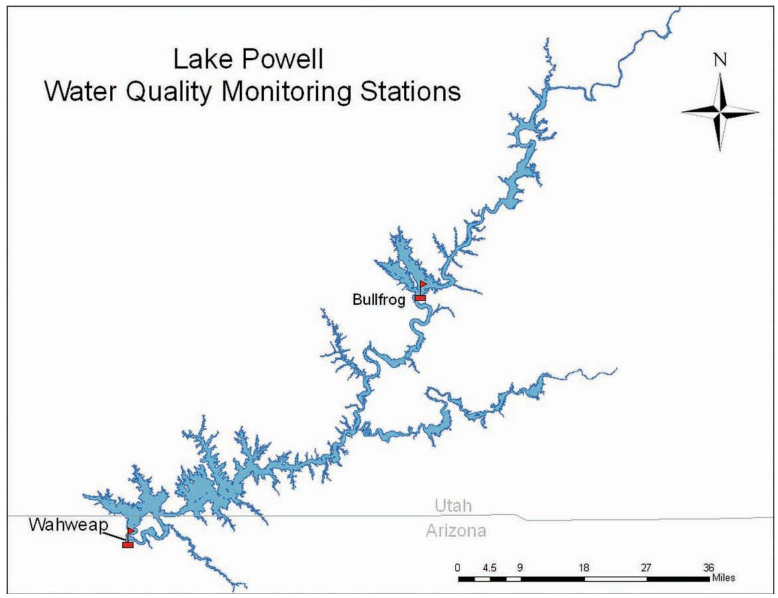

Figure 5. Lake Powell showing Wahweap and Bullfrog monitoring locations.

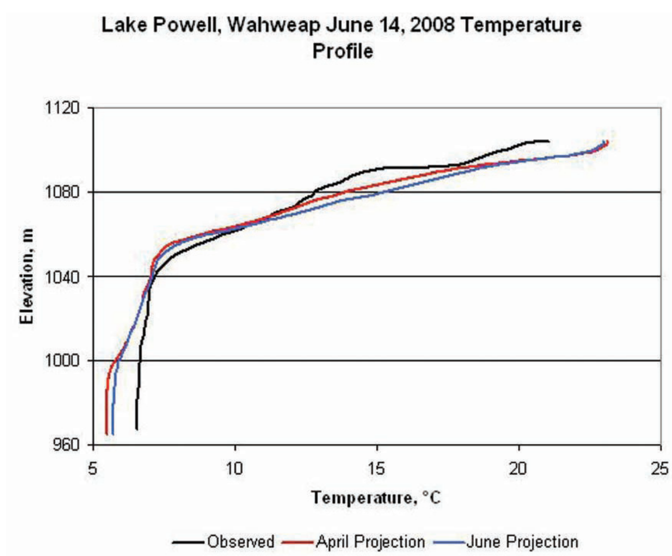

Figure 6. Wahweap, Lake Powell, June 14, 2008, temperature profile comparing projection simulation and observed temperatures.

Results

The results of reservoir temperatures from the projections simulation models are presented as depth profiles of temperature and are compared with actual reservoir temperature profiles measured during monitoring surveys during June 2008 and October-November 2008. Two reservoir monitoring locations were selected to present simulation and observed temperatures—Wahweap and Bullfrog (fig. 5). The June profiles for Wahweap (fig. 6) and Bullfrog (fig. 7) compare temperature results from the April 2008 and June 2008 projection simulations with the observed reservoir temperatures. The accuracy of the projections is determined from the AME statistic (eq. 1). The AME statistics of the projection simulations compared with the June observed data are shown in table 5.

The October-November profiles for Wahweap (fig. 8) and Bullfrog (fig. 9) compare temperature results from each projection simulation with the observed reservoir

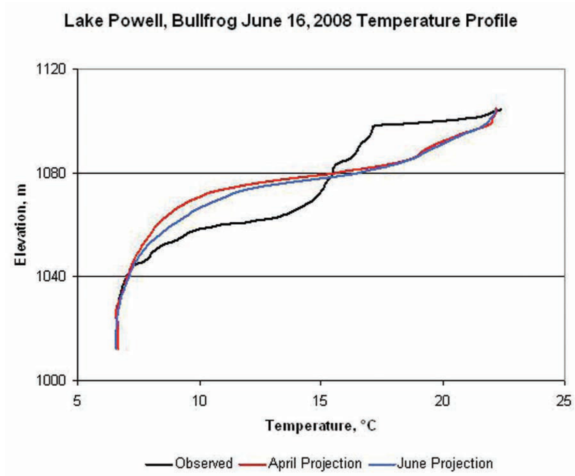

Figure 7. Bullfrog, Lake Powell, June 16, 2008, temperature profile comparing projection simulation and observed temperatures.

Table 5. Reservoir temperature profile absolute mean error statistics for 2008 projection simulations.

[°C, degrees Celsius; NA, not applicable]

Projection simulation	Wahweap profile June 14, 2008	Bullfrog profile June 16, 2008	Wahweap profile October 29, 2008	Bullfrog profile November 1, 2008
April 2008	0.99 °C	1.85 °C	0.84 °C	1.06 °C
June 2008	1.17 °C	1.64 °C	0.56 °C	1.04 °C
July 2008	NA	NA	0.48 °C	0.77 °C
October 2008	NA	NA	0.57 °C	0.83 °C

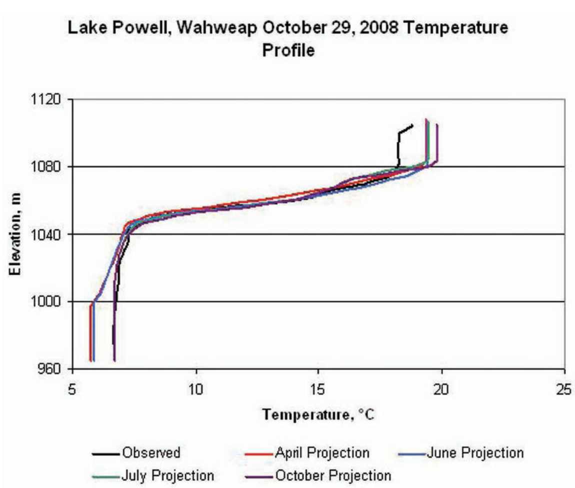

Figure 8. Wahweap, Lake Powell, October 29, 2008, temperature profile comparing projection simulation and observed temperatures.

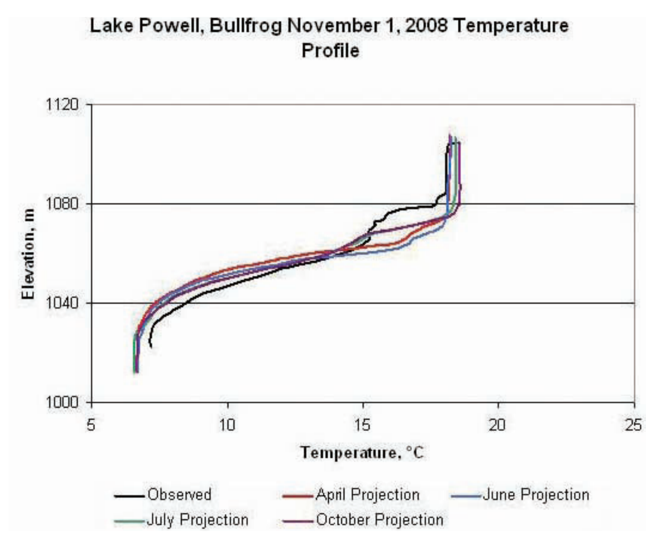

Figure 9. Bullfrog, Lake Powell, November 1, 2008, temperature profile comparing projection simulation and observed temperatures.

temperatures. The AME statistics of the projection simulations compared with the October-November observed data are shown in table 5.

The results of dam discharge temperatures from the projection simulation models are presented as daily average temperatures and compared with actual water temperatures from Glen Canyon Dam discharges between April and December 2008. The actual water temperatures are labeled "Below Dam DCP" (Data Collection Platform) in the figures displaying results. Results from the April 2008 projection simulation are presented in figure 10, results from the June 2008 projection simulation are presented in figure 11, results from the July 2008 projection simulation are presented in figure 12, and results from the August 2008 projection simulation are presented in figure 13.

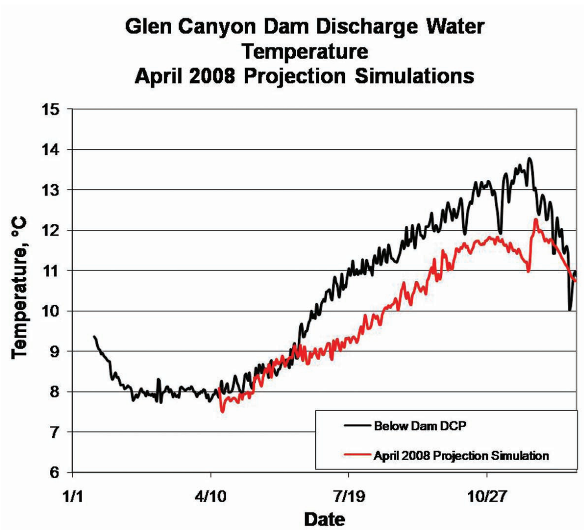

Figure 10. Glen Canyon Dam discharge water temperature, April 2008 projection simulation temperatures compared to Below Dam DCP temperatures.

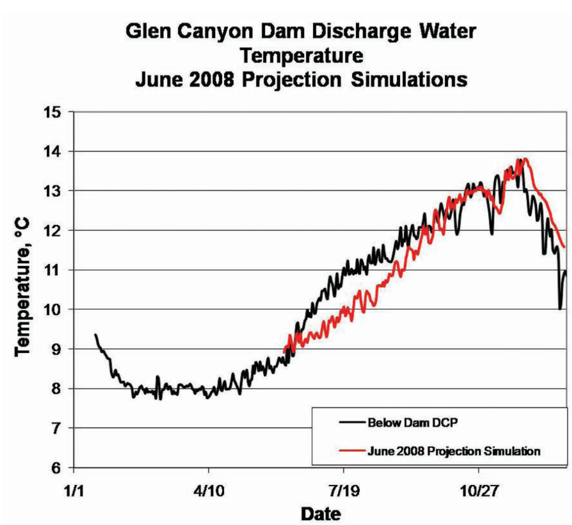

Figure 11. Glen Canyon Dam discharge water temperature, June 2008 projection simulation compared to Below Dam DCP temperatures.

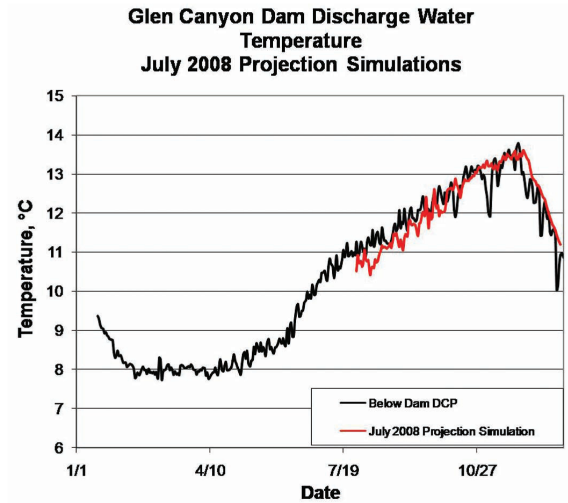

Figure 12. Glen Canyon Dam discharge water temperature, July 2008 projection simulation compared to Below Dam DCP temperatures.

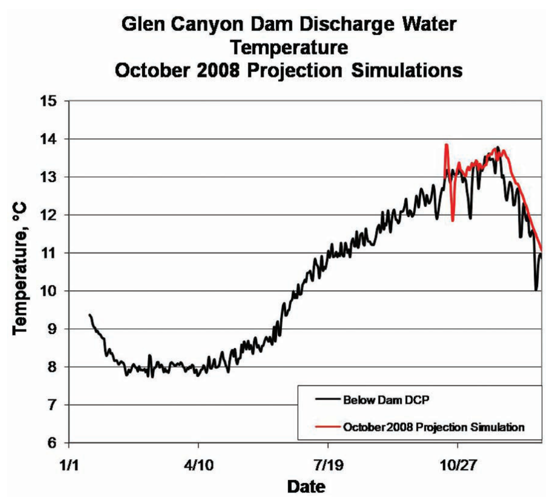

Figure 13. Glen Canyon Dam discharge water temperature, October 2008 projection simulation compared to Below Dam DCP temperatures.

Discussion

The results of water temperature in Glen Canyon Dam discharges using projection simulations are encouraging. As expected, projections are more accurate in the late season simulations as can be seen by comparing the April and June results with the July and October results. The April projections, in particular, do not adequately project the warmest discharge temperatures. The differences in the projections can be explained by several factors.

Warming is variable during spring months at Lake Powell and in the inflows. The July and October simulations capture this warming through the June initial conditions and actual meteorology between June and the date of the projection simulation (July or October). The April and June simulations rely on assumptions during the most critical time of reservoir warming, which is the spring runoff period.

Hydrological forecasts are subject to assumptions for snowpack accumulation, melting patterns, and other hydrologic factors. The forecasts are most variable during the periods of highest inflows, which are April through July. Base flows during the other months do not have as much variability. The April and June simulations use forecasts of spring runoff into the lake while the July and October simulations are done after spring runoff, thereby removing the uncertainty associated with runoff assumptions.

The projection simulations did not capture the development of stratification, especially in the upper reservoir as is illustrated by the June Bullfrog temperature profile. Based on the differences between the modeled results and the observed temperatures, the use of average meteorological data to represent meteorological conditions in the projection simulations may not be an appropriate assumption. Future projection simulations could explore alternate methods of representing meteorological conditions. Methods to disaggregate inflow volumes from monthly average flow rates to daily average flow rates could also be investigated.

Implications for Management

Reliable forecasts of water temperatures below Glen Canyon Dam are important to scientists and natural resource professionals involved in aquatic habitat studies in Grand Canyon. Results from the Lake Powell CE-QUAL-W2 model are input to a model of the Colorado River in Grand Canyon maintained by the GCMRC. The results from this model include water temperatures at several key locations along the river. The data from the two models allow professionals to know of temperatures conditions in advance and adapt studies accordingly. Accurate results from the CE-QUAL-W2 model are crucial to the Colorado River model and to resource management planning. Because the application of the model for projection simulations is still being developed and refined, a value of +/– 1 °C has been arbitrarily used to define accurate results. Continued development and experience with the projection simulations are expected to reduce that value.

It is anticipated that the model simulation results will continue to be used concurrently with the Colorado River model. Future uses will build on the knowledge and experience gained from this first year of model projections. Specifically, the early spring model projection will be considered qualitative, and recommendations to the GCMRC will include delaying detailed analysis and planning until a projection can be made in June or July. Subsequent projections in a given year will be used to confirm previous projection results or provide information in the event projections differ significantly.

Acknowledgments

I would like to thank Mr. Jerry Miller, retired Reclamation water-quality scientist, for his mentoring, input on the modeling, and insight into Lake Powell processes; Mr. Robert Radtke, Reclamation physical scientist, who supplied several images that were used during the presentation to illustrate reservoir water quality and processes; Mr. Rich Wildman for sharing his knowledge and insight into geochemical processes at Lake Powell; and finally, Mr. Bill Vernieu, USGS hydrologist, for providing monitoring data from Lake Powell limnological surveys that were used to calibrate the model and compare results.

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Bed Incision and Channel Adjustment of the Colorado River in Glen Canyon National Recreation Area Downstream from Glen Canyon Dam

By Paul E. Grams,[1] John C. Schmidt,[2] and David J. Topping[1]

Abstract

Closure of Glen Canyon Dam in 1963 reduced the magnitude and duration of spring floods, increased the magnitude of base flows, and trapped fine sediment upstream from the dam. These changes resulted in bed incision, bed armoring, and channel narrowing downstream in Glen Canyon. Channel-change measurements spanning over 45 years demonstrate that channel adjustment is directly related to both natural processes associated with sediment deficit and human decisions about dam operations. Most bed incision occurred in 1965 during pulsed high flows that scoured an average of 2.6 meters of sediment from the center of the channel. The average grain size of bed material increased from 0.25 millimeters in 1956 to over 20 millimeters in 1999. The decreased magnitude of peak discharges, extremely low sediment supply, and channel incision have resulted in erosion of sandbars and pre-dam flood deposits and the transformation of active bare sandbars and gravel bars to abandoned deposits that are stabilized by vegetation and no longer inundated. Erosion along the channel margins has been isolated to a few pre-dam flood deposits that eroded rapidly for brief periods and have since stabilized. Channel narrowing has resulted from decreased magnitude of peak discharges and minor post-dam deposition in the downstream part of the study area where riffles have not incised. These physical changes to the aquatic and riparian systems have supported the establishment and success of an artifact ecosystem dominated by nonnative species.

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Introduction

Large dams and their associated reservoirs typically trap upstream sediment supplies and drastically alter downstream flow regimes (Petts, 1979; Williams and Wolman, 1984). These changes in the driving variables that determine river channel form can result in sediment deficit, sediment surplus, or approximate sediment balance. River systems that have large post-dam peak flows and low tributary sediment supply are, consequently, in severe sediment deficit and typically exhibit signs of sediment evacuation (Schmidt and Rubin, 1995). In contrast, segments of regulated rivers that have low post-dam peak flows coupled with significant tributary sediment input may experience sediment surplus and post-dam sediment accumulation (Andrews, 1986; Grams and Schmidt, 2002, 2005). A deficit condition downstream from Glen Canyon Dam and associated bed incision were first documented just over a decade after dam closure in 1963 (Pemberton, 1976). However, the full range of effects of Glen Canyon Dam on the 25-kilometer (km) segment of the Colorado River between the dam and Grand Canyon National Park was not described until recently (Grams and others, 2007). This paper summarizes the findings of Grams and others (2007), which extends the record of change in bed elevation to May 2000, examines the pattern of bed scour, and evaluates the spatial pattern of erosion and deposition along the channel margins.

The Colorado River in Glen Canyon

Glen Canyon was named by John Wesley Powell on his exploratory journeys and is just one in the series of canyons carved by the Colorado River in its course across the Colorado Plateau. The canyon extends approximately 200 km from Hite, Utah, downstream to Lees Ferry, Arizona. Presently, all but the lowermost 25 km of Glen Canyon is flooded by Lake Powell, the reservoir formed by Glen Canyon Dam (fig. 1). Hereafter, we use "Glen Canyon" to refer to the portion of the canyon that is downstream from Glen Canyon Dam. In this reach, the

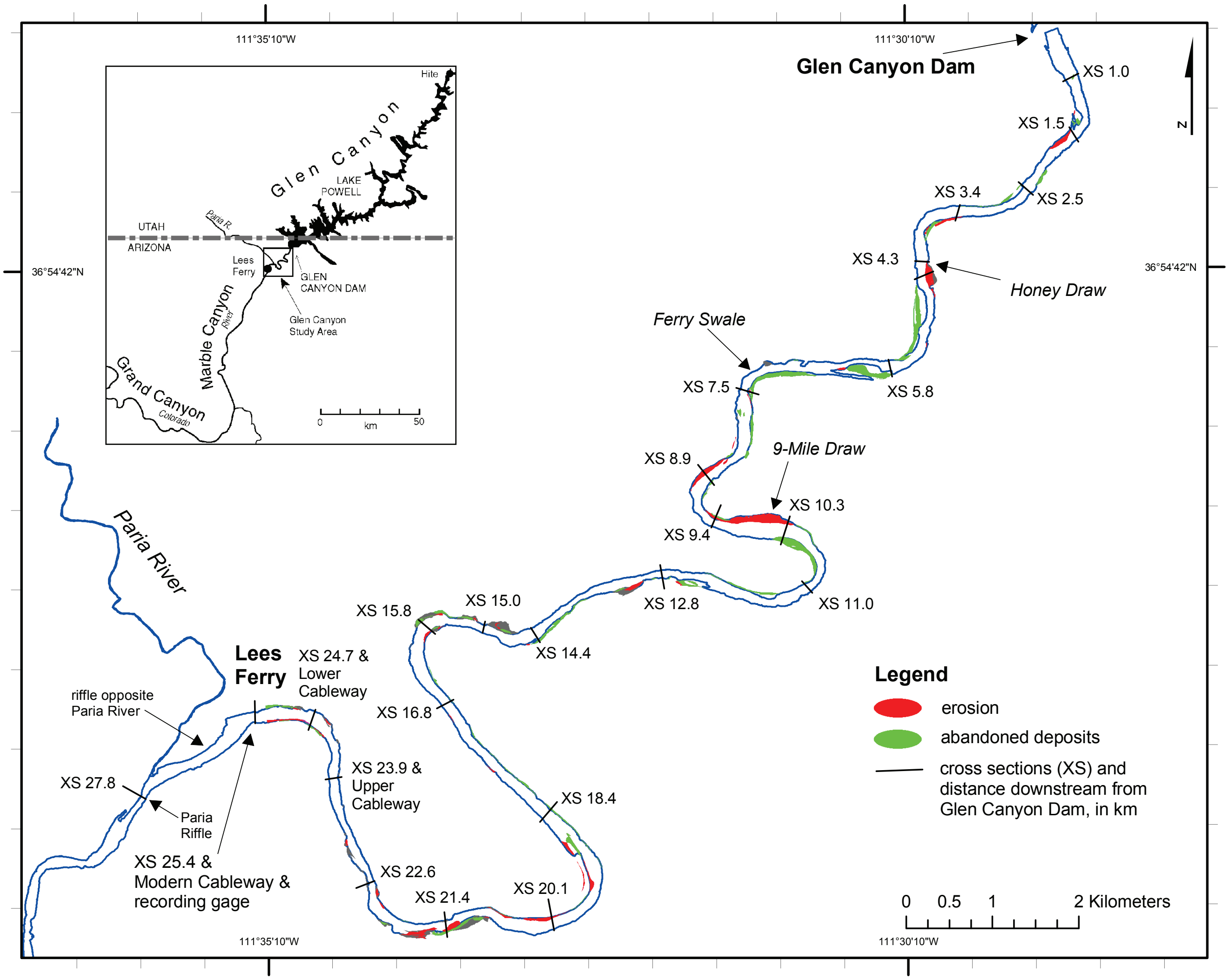

Figure 1. The Colorado River in Glen Canyon downstream from Glen Canyon Dam showing locations of monitoring cross sections and areas of erosion and deposition of channel-side deposits.

river is confined within mostly vertical sandstone walls, has a low average gradient of about 0.0003, occasional small riffles, and very few tributary debris fans. The average width of the channel inundated during the post-dam 2-year recurrence flow is about 146 meters (m), and the total width of the canyon bottom, including pre- and post-dam alluvial deposits, is about 183 m.

Peak Flows on the Colorado River: Pre- and Post-Glen Canyon Dam

Before the construction of Glen Canyon Dam, the Colorado River in Glen Canyon was free flowing with snowmelt floods that typically peaked in May or June. Smaller secondary peaks occurred at any time of year, but most frequently from July to October when summer thunderstorms triggered floods in tributary watersheds. In February 1959, a coffer dam that allowed the passage of floods was completed, and the river was diverted around the dam construction site. Flow regulation officially began in March 1963 when Glen Canyon Dam was completed, resulting in a 63 percent reduction in the average peak flow (2-year recurrence interval) from 2,407 cubic meters per second (m^3/s) to 892 m^3/s (Topping and others, 2003), slightly less than the 940 m^3/s maximum operating capacity of the Glen Canyon Dam powerplant (fig. 2).

Between dam closure and 2000, flows exceeded powerplant capacity in 7 years: 1965, 1980, 1983, 1984, 1985, 1986, and 1996. In May 1965, the dam's river-diversion tunnel, outlet works, and partially completed powerplant were used to release a large volume of water rapidly. These releases consisted of 14 pulsed flows with durations of a few days to more than 1 week (fig. 2). The pulses increased progressively in peak discharge from 435 m^3/s in February to 1,700 m^3/s in June 1965. After 1965, dam releases were at or below powerplant capacity until the early 1980s, when Lake Powell first reached full capacity. Soon thereafter, wet conditions in the Colorado River Basin required use of the spillway,

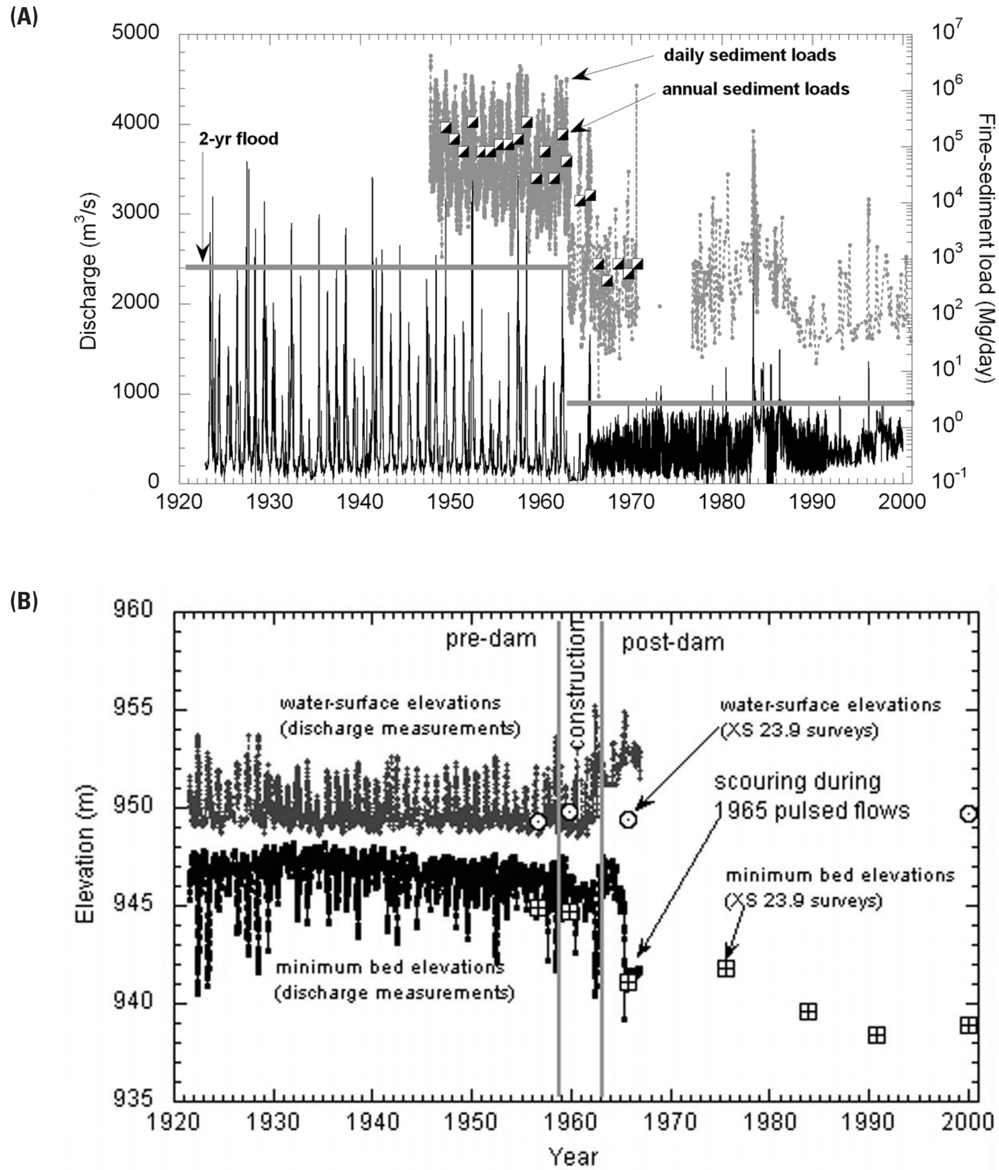

Figure 2. (A) Instantaneous discharge of the Colorado River at Lees Ferry, Arizona, 1921 to 2000, and measured sediment load for the same location, 1947 to 2000. The gray points connected by the dashed line are the computed loads for each day that sediment concentration was measured. The black and white boxes are the annual loads (expressed in megagrams per day (Mg/day)) computed by Topping and others (2000) for the years with sufficient data. The thick horizontal line indicates the magnitude of the pre- and post-dam 2-year recurrence peak flow. (B) The time series of water-surface elevations and minimum bed elevation for the upper cableway of the Lees Ferry gage from August 14, 1921, to December 1, 1966.

including a June 1983 release of 2,755 m^3/s, the highest flow in the post-dam period. The high release of 1996 was part of management efforts to restore components of the river ecosystem in Glen Canyon and in Grand Canyon National Park (Webb and others, 1999).

In addition to altering the flow regime, Glen Canyon Dam also resulted in almost complete elimination of the upstream sediment supply, which in the pre-dam period was $57 \pm 3 \times 10^6$ megagrams (Mg) per year (Topping and others, 2000). Measurements made between 1966 and 1970 at Lees Ferry indicate a post-dam annual load of about $0.24 \pm 0.01 \times 10^6$ Mg, a reduction of more than 99 percent (Topping and others, 2000).

The Timing and Pattern of Post-Dam Sediment Evacuation and Bed Incision

Sediment evacuation is the gross channel response to a deficit in sediment supply and may include erosion of material from the channel bed, from sandbars and gravel bars, and from the channel margins. Incision is the specific process of sediment evacuation that results in lowering of the river bed such that for similar discharges the water surface is also lowered. Distinction between these evacuation processes is especially important in systems where water-surface elevations are controlled at discrete locations by particular channel features, such as rapids or riffles.

The spatial distribution of sediment evacuation in Glen Canyon is well documented by repeat measurements of the elevation of the channel bed made periodically from 1956 to 2000 by the Bureau of Reclamation at 24 monumented channel cross sections established at approximately 1-km intervals between the dam and Lees Ferry (fig. 1). Because measurements at the cross sections were made infrequently, precise timing of sediment evacuation is best shown by repeated discharge measurements made from 1921 to 2000 at the U.S. Geological Survey streamgaging cableways near Lees Ferry (fig. 1). The methods used to analyze these records and construct time series of bed elevation change are described in Grams and others (2007).

The measurements of bed elevation made during discharge measurements at the upper cableway show that the bed was very dynamic in the pre-dam period, often scouring and refilling several meters in a single season (fig. 2). This pattern continued during dam construction, but once flow regulation began, bed elevation was stable until the 1965 pulsed flows that rapidly resulted in about 4 m of erosion. The measurements made at the cross sections located throughout the study area show that the bed lowering, constrained precisely in time at the upper cableway, also occurred throughout Glen Canyon. Some of the cross sections near the dam began eroding during dam construction when the coffer dam was partially regulating flow, but most of the erosion occurred between the time of the 1959 measurement and the measurement made after the flow pulses in 1965 (fig. 3). These measurements demonstrate that bed lowering occurred both in pools and riffles, resulting in a significant change in the water-surface profile from the pre- to post-dam period (fig. 4). Notably, the magnitude of lowering of riffles decreased with increasing distance downstream from the dam whereas the amount of bed lowering and sediment evacuation from pools is not correlated with distance downstream (fig. 3). This is consistent with observations that no channel controls (riffles or rapids) downstream from XS 20.1 (cross sections are labeled with the abbreviation XS followed by distance downstream from Glen Canyon Dam, in kilometers) have scoured, whereas pools downstream from this point have scoured. The observed longitudinal pattern of bed incision caused the reach-average gradient to decrease by about 25 percent, from 0.0004 to 0.0003 at a low-flow discharge of 150 m^3/s (fig. 4).

The sand-bed surface and some underlying gravel were eroded in the process of sediment evacuation. At the time of the initial cross-section measurements in 1956, the bed was mostly sand, and the average bed-surface grain size was about 0.25 millimeters (mm). This sand was underlain at depths of up to 4 m by mixed sand and gravel that had a median grain size of about 20 mm. During evacuation, all of the sand and between 0 and 8 m of gravel was eroded from the bed (fig. 4). Evidence for this erosion into the underlying gravel is based on measurements of the depth to gravel made in 1956 at XS 4.3, XS 5.8, XS 12.8, and XS 16.8. These data indicate that approximately 50 percent of the material evacuated between 1956 and 2000 was derived from beneath the sand veneer.

Based on analysis of the cross-section measurements, an estimated 12.6 x 10^6 cubic meters (m^3) (21.6 x 10^6 Mg) of sand and gravel were evacuated from the study reach between the beginning of dam construction and 2000. Approximately 37 percent of the total evacuation measured and 64 percent

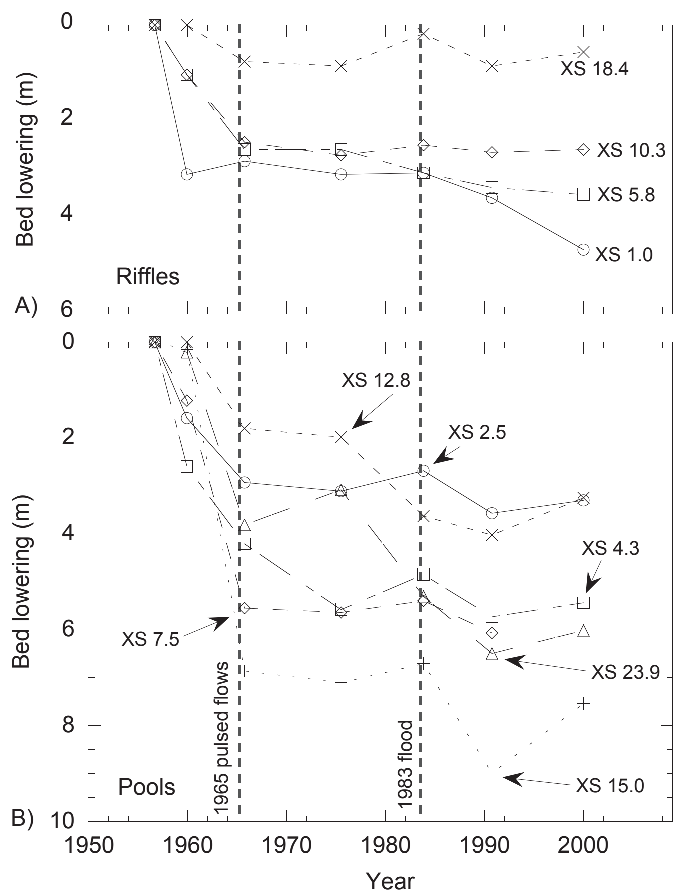

Figure 3. Magnitude of decrease in minimum bed elevation from 1956 to 2000 at the 10 monitoring cross sections grouped by (A) riffles and (B) pools. Each cross section is labeled by distance downstream from Glen Canyon Dam, and the times of the 1965 pulsed flows and the 1983 flood are indicated.

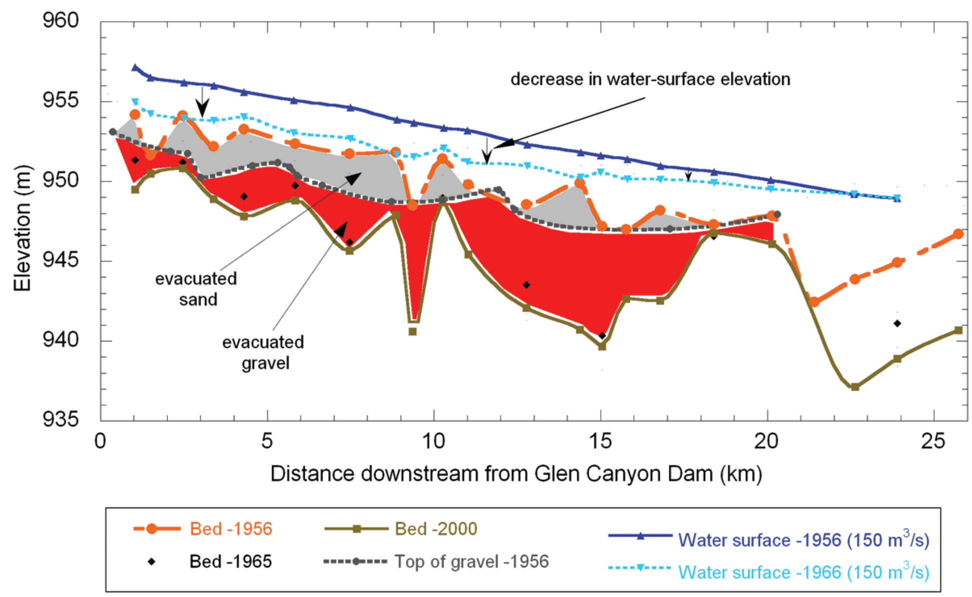

Figure 4. Longitudinal profile showing minimum bed elevation for each of the Bureau of Reclamation surveys and elevation of the top of the gravel layer determined by bore-hole and jet-probe measurements made in 1956. Water-surface profiles for a low-flow discharge of 150 m^3/s are also shown. For the distance of 7.5 km downstream from the dam, a measurement made in 1990 was used for the 2000 bed elevation because that station was not measured in 2000.

of the evacuation that occurred after the dam was completed occurred between March 1963 and September 1965. Because dam releases from March 1963 through February 1965 were extremely low, it is likely that the majority of the erosion occurred during the 1965 pulsed flows. Bed lowering and sediment evacuation continued after the 1965 pulsed flows but at lowered rates.

Sand to Gravel: Changes in the Alluvial Deposits in Glen Canyon

In the pre-dam period, Glen Canyon was characterized by abundant channel bar deposits. These sand and gravel deposits were exposed above the water surface during low flow and discontinuously located in or near the edges of the channel, in eddies, and along the inside of bends. The bars were mostly unvegetated. Comparison of pre- and post-dam photographs (figs. 5 and 6) from two sites located 15 and 22.6 km downstream from the dam shows erosion of pre-dam deposits, widespread abandonment of pre-dam deposits resulting from incision, post-dam deposition, and vegetation encroachment. These key changes are diagrammed in cross-section view in figure 7.

These changes were evaluated throughout the study area by comparing maps made from 1952 aerial photographs and 1984 aerial photographs in a geographic information system. On average, active-channel width in the study area decreased by 6 percent, from 156 m to 146 m, between 1952 and 1984. In the upstream 20 km of the study area, inundation frequency of the pre-dam flood deposits decreased because of bed incision and decreased magnitude of annual high flows. This change resulted in the abandonment of alluvial deposits not inundated by post-dam high flows, an increase in the area of alluvial deposits inundated at discharges between 300 and 600 m^3/s, and an overall narrowing of the active channel. Deposits left by the post-dam high flows have also contributed to channel narrowing because they are rarely inundated and have been colonized by vegetation, consisting primarily of tamarisk. Although this invasive shrub has been present in the region since the 1930s (Clover and Jotter, 1944), it increased in abundance after 1952 (Turner and Karpiscak, 1980). Despite sediment evacuation, the area of channel-side and mid-channel sand deposits exposed at flows of similar recurrence has not changed significantly. However, the proportion of the alluvial valley that is covered by deposits with perennial, riparian vegetation has increased while the area of bare sand has decreased.

Erosion of pre-dam deposits along the channel margins also occurred but was not widespread. The largest area of erosion between 1952 and 1984 occurred near XS 10.3, where a large part of a pre-dam flood deposit was eroded (fig. 1). Thus, with the exception of these isolated areas of erosion, deposits along the channel margins have maintained or increased stability whereas the channel bed incised.

(A)

(B)

Figure 5. Looking downstream at a small debris fan and sandbar on the left bank of the Colorado River about 3 km upstream from Lees Ferry between XS 21.4 and XS 22.6. (A) The first photograph was taken by Robert Brewster Stanton on December 26, 1889. (B) The second photograph was taken by Tom Wise on October 28, 1992. The discharge for the date of the original photograph is not known, but the mean daily discharge for the months of December and January in the pre-dam period was 156 m^3/s. Flow at the time of the 1992 repeat was 275 m^3/s. Note the much smaller area of bare sand and much larger area occupied by woody riparian vegetation (tamarisk) in 1992.

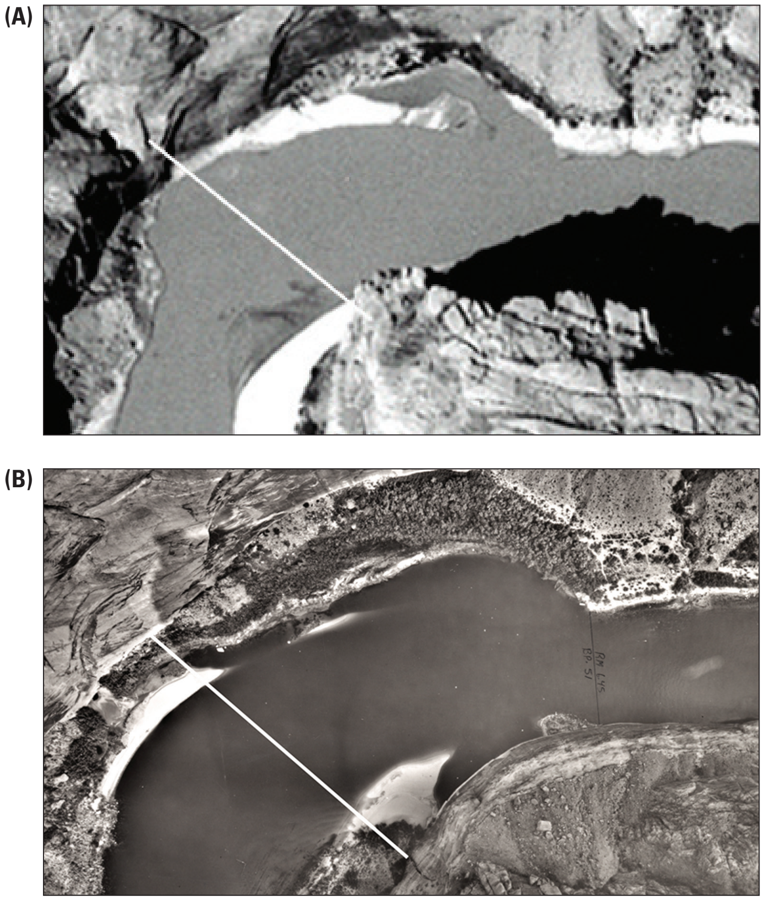

Figure 6. Clips from aerial photographs taken in (A) 1952 and (B) 1984 about 16 km downstream from Glen Canyon Dam near XS 15.8 (white line). Discharge was 290 m³/s at the time of the 1952 photograph and 141 m³/s at the time of the 1984 photograph. Note the bare sandbars and narrow strips of vegetation in the 1952 photograph. Streamflow is from right to left.

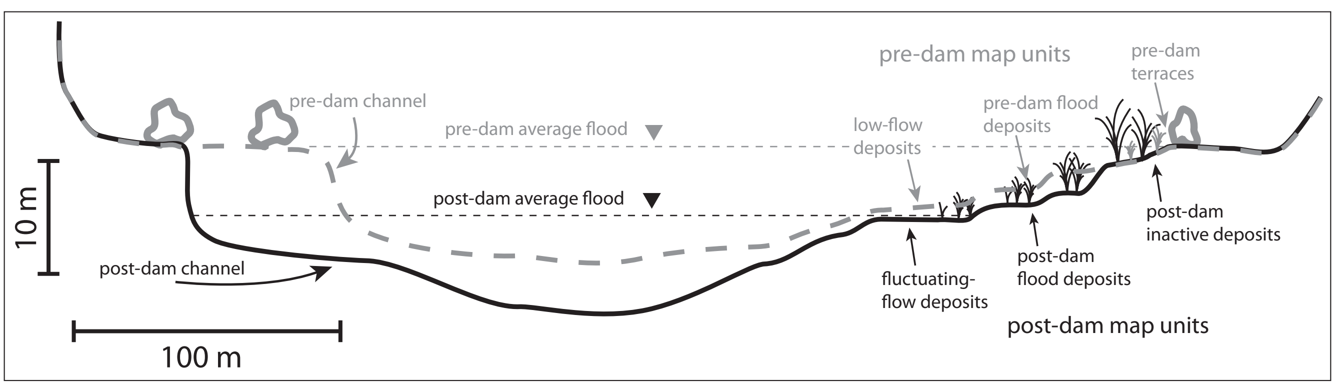

Figure 7. Relative elevations of mapped deposits in relation to the pre- and post-dam Colorado River channel. The approximate stages of the pre- and post-dam average (2-year recurrence interval) high flows of 2,407 m³/s and 892 m³/s, respectively, are also shown. The post-dam average high flow is approximately 7 m lower than the pre-dam average high flow. Vegetation on the post-dam deposits is mostly tamarisk, but other riparian species occur. Vegetation on the post-dam inactive deposits is mostly upland grasses and shrubs.

The volume of material eroded from deposits along the channel margins outside the low-flow channel throughout the study area was small compared to the volume of sediment eroded from the bed. We estimate that $3 \pm 1 \times 10^6$ Mg of sand and gravel was eroded from channel-side sand and gravel deposits (based on a specific gravity of 2.65 and a porosity of 35 percent), equivalent to about 14 percent of the estimated mass eroded from the bed. This estimate is based on extrapolating the thickness of eroded deposits from the locations where cross-section surveys show eroded pre-dam deposits to all areas where comparison of the 1952 and 1984 photographs showed erosion (fig. 1). Based on those cross sections, 6 ± 1 m eroded from pre-dam deposits along the channel margins and 2 ± 1 m eroded from pre-dam low-flow deposits.

Conclusions

The closure of Glen Canyon Dam and subsequent flow regulation caused average peak flows in Glen Canyon to decrease by about 63 percent and essentially eliminated the fine sediment supply for the 25-km reach downstream from the dam. These changes resulted in bed-sediment evacuation, channel incision, channel narrowing, vegetation encroachment, and the transformation from a sand-dominated to a gravel-dominated river channel. The highest rate of sediment evacuation occurred in 1965 during a series of pulsed dam releases. Whereas the magnitude of bed lowering was predictable, the rate and timing of lowering were determined by management decisions about dam operations. The magnitude of bed lowering of riffles was greatest near the dam and decreased downstream, resulting in a lowered post-dam reach-average water-surface gradient that extended more than 20 km downstream from the dam. This decrease in gradient coupled with an increase in the average bed-material grain size from about 0.25 mm to about 20 mm provides a negative feedback that reduces the likelihood of further bed incision at riffles (Grams and others, 2007). This joint adjustment of bed-material grain size and gradient has resulted in the transformation of an adjustable-bed alluvial channel to a stable channel with an infrequently mobilized bed.

In contrast to the response measured at riffles, the magnitude of sediment evacuated from pools did not decrease systematically downstream. The pools continued to exhibit sediment evacuation after incision at riffles had ceased. This demonstrates that riffle controls do not limit the downstream extent of scour and that pools can scour even where riffle scour does not occur. Thus, even though the riffles that control the channel gradient are likely stable, continued scour in pools is possible.

The lowering of the bed and water surface coupled with decreased peak-flow magnitude and low post-dam sediment supply have caused isolated erosion of sandbars and pre-dam flood deposits, but more importantly, widespread areas of previously active sandbars and gravel bars have become disconnected from the channel and abandoned because they are no longer inundated by post-dam flows. Whereas hillslope processes and gullying may result in future local erosion of pre-dam deposits, large-scale erosion associated with channel incision is no longer evident. The abandoned deposits are above the low-discharge water-surface elevation and are stabilized by riparian vegetation. In the downstream part of the study area where incision has not occurred at channel controls, channel narrowing has been caused by decreased peak-flow magnitude and vegetation encroachment. These physical changes to the aquatic and riparian environments in Glen Canyon have supported the establishment of an ecosystem of largely nonnative plant species.

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Water Velocity of the Colorado River: Implications for Native Fishes

By Christopher S. Magirl[1] and Matthew E. Andersen[2]

Abstract

The native aquatic biota in bedrock-controlled reaches of the Colorado River and its tributaries evolved in highly variable conditions of streamflow and habitat structure. Water velocity in the river is governed by pool-and-rapid sequences, with generally slower water in pools and faster water in rapids. For example, while velocity values as great as 6.5 meters per second were measured in rapids in Cataract Canyon, flow velocity within 0.3 meter of the riverbed was, on average, 60 percent slower than the velocity measured near the water surface. Maximum velocities in slower sections between the rapids range from 0.5 to 2 meters per second. In the modern era when dams have altered physical aquatic environments, management of native fishes may be improved with a better understanding of how organisms interact with the altered hydraulic regime. Different river reaches may be available to various life stages of endangered native fishes depending on local conditions of flows released from dams. Newly collected velocity data from pools and rapids in the Colorado River give some insight into whether fish may negotiate different reaches of the river under changing flow regimes, though specific conclusions are not possible with the current dataset. This article summarizes the hydraulic data that have thus far been collected and suggests where future research is needed to better understand the interactions between aquatic ecology and hydraulics in the Colorado River.

Introduction

Rapids are widespread in many canyons of the Colorado River and its tributaries, including Cataract and Grand Canyons (fig. 1). Almost all rapids in the Colorado River were formed by the deposition of boulders at tributary mouths from flash flooding and debris flows. Over time, alluvial fans at these tributaries build, constricting the river and forming turbulent, high-velocity rapids. These constrictions also create pools upstream from rapids. This character of interspaced pools and rapids is almost immediately apparent to anyone who floats the river and is well described in the literature (Leopold, 1969; Melis and others, 1995; Webb and others, 2004). The hydraulic character of the river also has implications for the movements of native and nonnative fishes, especially at younger life stages.

Following closure of Glen Canyon Dam in 1963, the physical characteristics of the Colorado River in Grand Canyon changed. Before the dam, the river was sediment laden, with large snowmelt floods in spring (discharges above 2,000 cubic meters per second (m^3/s) were common) and small flows (≈50 m^3/s) at other times of the year. Regulated releases from Glen Canyon Dam of 283–566 m^3/s are typical today. Similarly, the river temperature fluctuated between 0 and 25 degrees Celsius (°C) in the pre-dam era; fluctuations between 8 and 12 °C are typical under flow regulation, though regional drought has resulted in warmer temperatures in recent years (Voichick and Wright, 2007).

Native fishes adapted to the turbulent and variable nature of flows of the rapid-rich Colorado River (Douglas and Marsh, 1996). For example, the humpback chub, *Gila cypha*, a long-lived and federally listed endangered native fish found only in the Colorado River Basin, reaches 50 centimeters (cm) in length and possesses features that distinguish native Colorado River fishes: large adult body size, large predorsal hump, a streamlined caudal peduncle, and a relatively large caudal fin. While the adult population of humpback chub in Grand Canyon declined steadily through the 1990s, recent improvement to an estimated 7,650 adult individuals was observed in 2008 (Coggins and Walters, 2009).

Analysis of long-term monitoring data suggests the majority of humpback chub below Glen Canyon Dam are found in the vicinity of the Little Colorado River (Paukert and others, 2006). Valdez and Masslich (1999) found adult and young-of-year humpback chub in the mainstem Colorado River (upstream from the Little Colorado River) near the in-stream Fence Fault Springs around river mile 30, suggesting adults can move upstream against rapids in the current dam-release regime. Upstream movement of young-of-year

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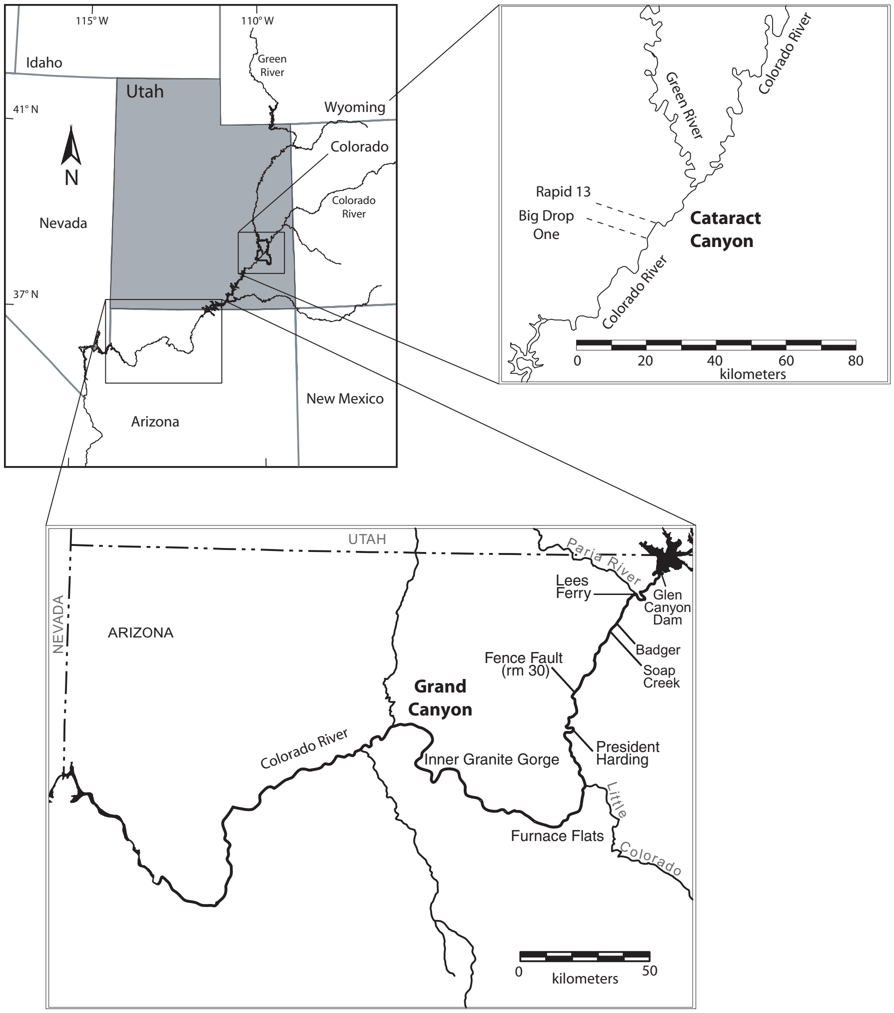

Figure 1. The Colorado River in the Southwestern United States. Water velocity was measured at locations in Cataract Canyon in Utah and Grand Canyon in northern Arizona.

and juvenile humpback chub 1 to 3 years old has not been documented in the Colorado River, though laboratory research has shown age-0 humpback chub can maintain a swimming speed of 0.4 meter per second (m/s) up to 2 hours (Berry and Pimentel, 1985); larger fish swim faster than younger fish and all fish swim faster in warmer water. Humpback chub that appear to be in juvenile size classes (1 or more years old) were captured in the vicinity of Fence Fault in 2006 and 2007 (Andersen and others, 2010), prompting an important question: Can juvenile humpback chub swim upstream from the Little Colorado River to the Fence Fault reach or were observed juveniles reared locally? Moreover, a broader research opportunity exists to better quantify the hydraulics in the Colorado River and assess the response of both native and nonnative fishes to changes in hydraulic regime.

Water Velocity in the Colorado River

Water velocity in the pool sections of the Colorado River generally ranges from 0.5 to 2.0 m/s. Graf (1997) used dye tracer studies to determine mean velocity in Grand Canyon was about 1.0 m/s at 425 m^3/s and 1.8 m/s at 1,270 m^3/s. Mean velocity increased about 15 percent in narrow, confined reaches of the canyon like Inner Granite Gorge and decreased about 15 percent in wide unconfined reaches like Furnace Flats. In the 1980s, Kieffer (1987, 1988) made pioneering measurements of water velocity in rapids by using floating tracer particles. Velocities at the water surface of rapids ranged from 5.0 to 7.0 m/s, and one measurement of 10.0 m/s was recorded. While Kieffer's work was insightful, research questions concerning the speed of water below the surface remained.

Recent studies with flow-measurement instruments (including an acoustic Doppler current profiler and a pitot-static tube) in rapids and riffles in Cataract Canyon (Magirl and others, 2009) and in Grand Canyon (Magirl and others, 2006) give better insight into the nature of water velocity within rapids. Recent computer modeling of the Colorado River at higher discharge further extends our understanding of how water moves in the river system (Magirl and others, 2008). Specifically, we have a much better understanding of how water velocity and hydraulics in the river change as a function of location, time, and discharge.

Water Velocity as a Function of Location

Water velocities in the tranquil sections of the Colorado River at low discharge, particularly upstream from constricting debris fans at rapids, can be relatively small. Velocity values of 0.5–2.0 m/s are common in pools, and velocities are usually less than 2.0 m/s for discharge less than 500 m^3/s. Near the shoreline and along the bottom of the bed in tranquil reaches, flow velocities can be almost zero, and there are broad spatial regions on the benthic substrate where velocity at low discharges is less than 0.25 m/s. Figure 2 graphically shows velocities at 280 m^3/s and 1,110 m^3/s measured at a transect near river mile 30 in Grand Canyon. Peak velocities in this pool section of the river during the lower discharge were on the order of 1.5 m/s, and flow velocity was generally less than 0.5 m/s along the bed of the river. At higher discharges, velocity on the order of 2.0–3.0 m/s was common, and slow regions of flow were present near the river bed, though these regions were less extensive than the slow regions observed at lower discharge. Regions of low velocity can act as migratory pathways for fish moving upstream.

In contrast, flow velocities in rapids can be large. Figure 3 shows mean flow velocities on the order of 5.0 m/s were readily measured in Big Drop One Rapid in Cataract Canyon in eastern Utah with a peak instantaneous velocity of 6.5 m/s (Magirl and others, 2009). But even within rapids, regions of relatively slow-moving water exist along the shorelines and near the bed. In Rapid 13 in Cataract Canyon, for example, the velocity within 0.3 m of the riverbed was, on average, 60 percent slower than the velocity measured near the water surface. More importantly, large boulders (many larger than 1.0 meter (m)) stabilize rapid-forming debris fans and create localized eddies of slower velocity that, presumably, act as refuges for migrating fish. These pockets of slow water are prevalent along the shoreline of a rapid.

Water Velocity as a Function of Time

Velocity in the river is also a strong function of time. Flow in all rivers is turbulent, even in seemingly tranquil reaches, and this turbulence is readily seen on the water surface as boils and seam lines. Velocity in turbulent flow is not constant, but fluctuates around an average value. In the Colorado River, turbulent eddies sweep sediment and nutrients off the bed and tend to keep the water well mixed. Analogous to gusts on a windy day, turbulent eddies also push high-velocity eddies of water down to the river bed disrupting sands and other organisms that might otherwise collect in slower water. Measurements by Magirl and others (2009) of water velocity at fixed points above Big Drop One Rapid in Cataract Canyon show how turbulent fluctuations in the flow velocity behave near the rapid (fig. 3). At 150 m upstream from the core of the rapid, flow velocities were on the order of 2.0–3.0 m/s with moderate turbulent fluctuations. Further downstream, at 110 m and 130 m upstream from the core of the rapid, flow velocity increased while turbulent fluctuations seemed to lessen.

At longer time scales, the river channel itself also changes with time. In the canyons of the Colorado River, frequent flash floods and debris flows from tributaries dump

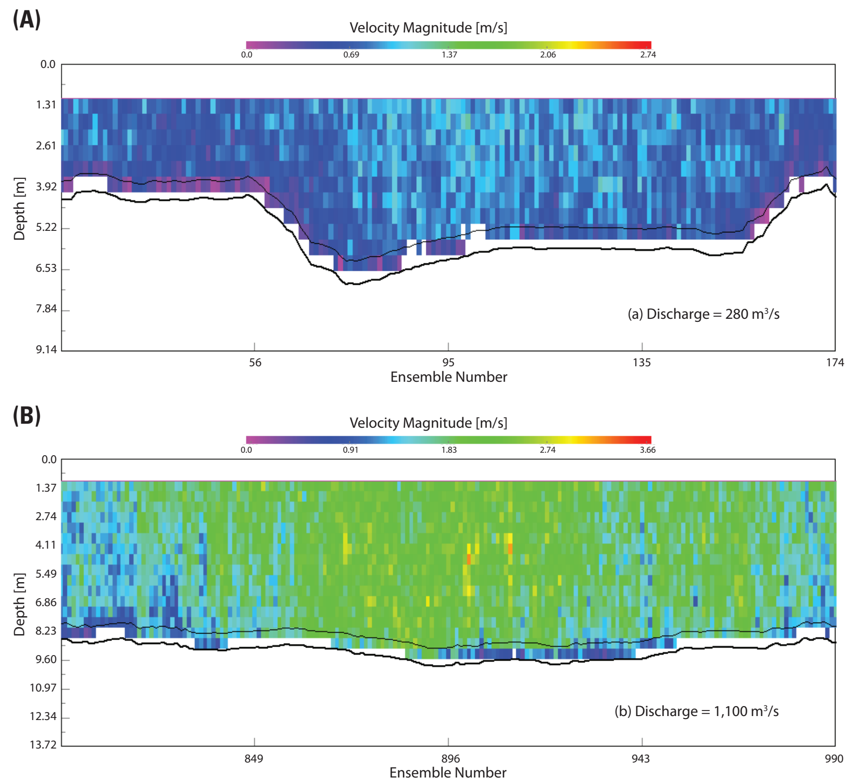

Figure 2. Flow velocities as measured with an acoustic Doppler current profiler are shown in a pool section of the Colorado River in Grand Canyon near river mile 30 for (A) low discharge of about 280 m^3/s and (B) larger discharge of about 1,100 m^3/s. The term “ensemble” refers to serial measurements from the instrument and represents a proxy for position along the river-wide transect from left shoreline to right shoreline.

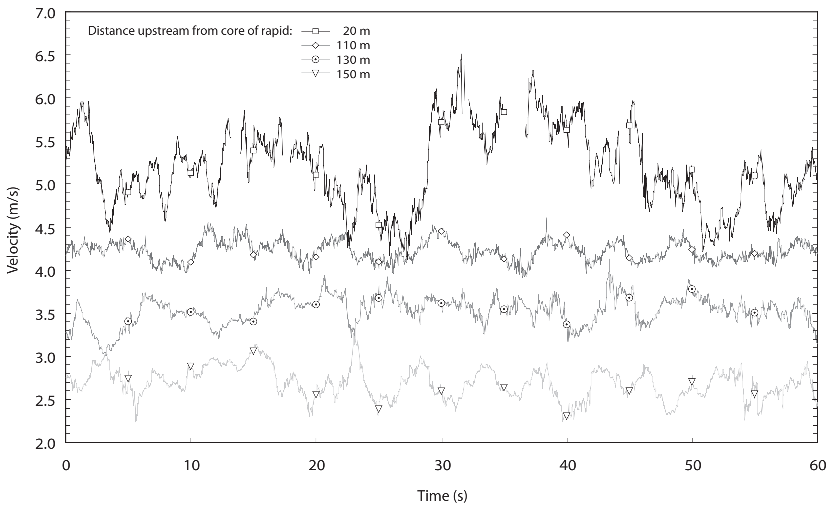

Figure 3. Turbulent flow velocities measured at the top of Big Drop One Rapid on the Colorado River in Cataract Canyon.

release (≈1,100 m^3/s). Water velocity increased from about 1.0 m/s to almost 2.0 m/s with this increasing discharge. In fact, consistent with the findings of Graf (1997), the flow velocity in all pool sections of the Colorado River increased with increasing discharge. The nature of flow velocity in rapids is more complex.

As flow in the Colorado River in Grand Canyon rises from 227 m^3/s to 850 m^3/s, the water velocity in most rapids becomes faster. As the discharge increases beyond 850 m^3/s, however, many rapids “drown out” or become less severe as downstream hydraulic control reduces water slope within the rapid. For example, in the reach of river below Lees Ferry, computer modeling (Magirl and others, 2008) with large floods shows Paria Riffle, Badger Rapid, and Soap Creek Rapid all get much less severe as hydraulic features for discharges above 2,000 m^3/s (fig. 5). These three rapids completely drown out for flows above 4,800 m^3/s. This is a surprising result to those unfamiliar with large floods in Grand Canyon because Badger Rapids and Soap Creek Rapids are large, significant rapids at most modern discharges. However, historical accounts of these rapids and photographs from the early 20th century support the model predictions (Schmidt, 1990).

Further downstream, in the reach between river mile 30 and the confluence with the Little Colorado River (river mile 62), the computer model predicts many moderately sized rapids lessen in severity at discharges between 1,100 m^3/s and 2,500 m^3/s, although the bigger rapids (for example, President Harding Rapid, Kwagunt Rapid, and 60-Mile Rapid) remain prominent hydraulic steps in the river profile. This is an intriguing observation, possibly suggesting that native fish may have used spring floods as windows of opportunity to migrate upriver when the relative severity of some rapids is reduced.

coarse-grained alluvium in the river corridor making rapids steeper and more severe (Webb and others, 1989; Magirl and others, 2005). An aggraded debris fan can increase water velocity within a rapid and concurrently slow the water velocity in the pool above the rapid. In turn, flooding on the Colorado River removes accumulated alluvial material from debris fans slowing the water in the rapid and reducing the severity of the rapid.

Water Velocity as a Function of Discharge

Finally, water velocity and hydraulics in the Colorado River change with discharge. Figure 2 shows water velocity at different depths for the pool section of river near river mile 30 in Grand Canyon. The range of water velocities as a function of depth (with turbulent fluctuations) is shown in figure 4 at low flow (≈280 m^3/s) and during the 2008 controlled

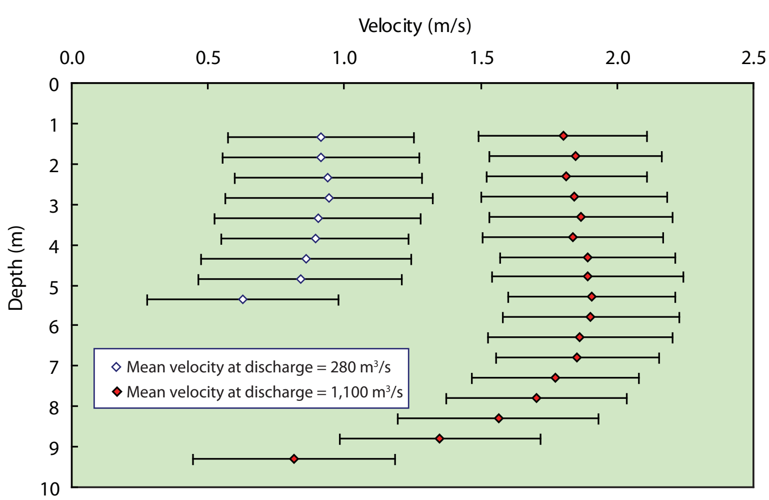

Figure 4. Average flow velocity in a pool section of the Colorado River in Grand Canyon near river mile 30 during low discharge and the controlled release of March 2008. The error bars represent the total range of turbulent fluctuations of velocity measured in the flow. Mean velocity in the pools almost doubled with the higher discharge.

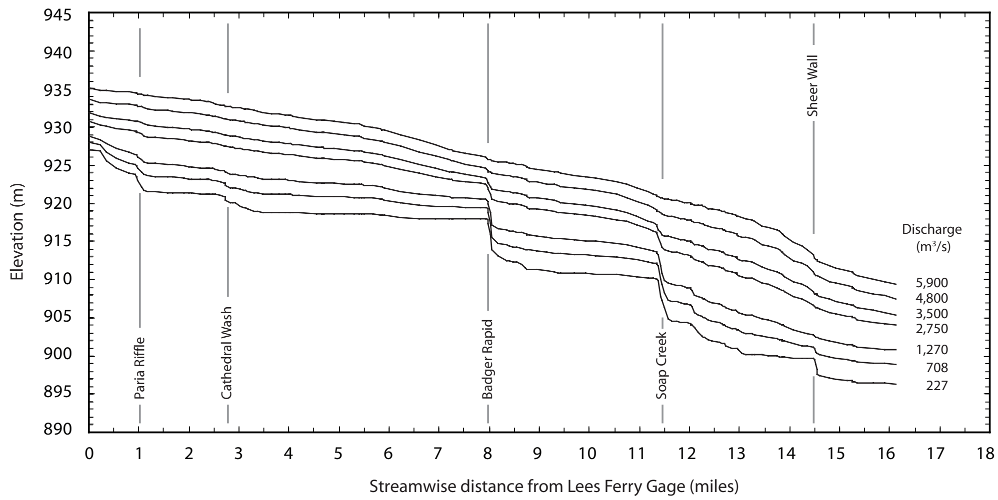

Figure 5. Predicted water-surface elevation profiles of the Colorado River downstream from Lees Ferry for discharges ranging from 227 m^3/s to 5,900 m^3/s. Badger Rapid and Soap Creek Rapids, both large rapids, completely drown out at a discharge of 4,800 m^3/s.

Fish

The ability of fish to move is critical for feeding, spawning, and predator avoidance, among others. Water velocity is a principal environmental factor that limits or aids fish movement. The evolution of fish native to the Colorado River forced swimming styles and behavior optimized for a muddy river prone to annual swings in discharge and temperature. Introduced fish may not have evolved strategies to navigate high, turbulent flows (Minckley and Meffe, 1987; Valdez and others, 2001). Rapids on the Colorado River are predominately formed by debris-flow processes from tributaries, resulting in flowing water that tumbles down and around collections of rounded boulders. These boulder piles create regions of variable flow and multiple pathways for the possible upstream movement of adult fish, though upstream movement of juveniles is less likely.

With observations of juvenile humpback chub in the Fence Fault reach and knowledge of fish swimming capabilities measured by Berry and Pimentel (1985), we postulate juvenile humpback chub observed in the Fence Fault reach were reared locally as opposed to migrating 50 kilometers (km) upstream from the Little Colorado River. However, available hydraulic data, which include observations and modeled estimates of mean velocity across a channel cross section and detailed observations of instantaneous velocity at specific locations within rapids, do not include enough detailed observations in potential low-velocity areas to permit an assessment of whether upstream navigation by juvenile chub is possible. These data do suggest, however, that if upstream navigation by these fish did occur, the fish would have to utilize shallow near-shore or near-bed areas because velocities near the center of the channel generally exceed their swimming ability.

Needed Research

Recent studies, coupled with previous research, tell us something about the nature of water velocity in the Colorado River. While these new data offer insight, better understanding of the interactions between ecology and water velocity is needed. More velocity data are needed within rapids specifically focusing on three-dimensional flow structures, velocity magnitude throughout the water column, flow strength near the bed, and interactions between flowing water and native and nonnative fishes. These velocity data need to be collected at varying discharge; a thorough understanding of water velocity at different discharges informs us about the potential for movement of native and nonnative fishes as well as the river’s impact on species success. Because of limits of the instrumentation and safety concerns when working in fast-flowing water, flow-measurement data should also be augmented with hydrodynamic computer models that enable detailed analysis of flow structures in the river. These models are most valuable when calibrated with velocity data. While three-dimensional modeling is needed to fully characterize flow structures in a pool-and-rapid sequence, much insight could come from simulations using widely available two-dimensional models.

If specifically attempting to answer the question of the upstream mobility of juvenile and adult humpback chub between the Little Colorado River and Fence Fault, a hydraulic and ecologic study of the entire river reach would be necessary. Such a study, however, would be time consuming, logistically challenging, and expensive. In contrast, a detailed study that is spatially limited to a smaller subreach of river, though still spanning multiple rapids and pools, would be scientifically useful and cost efficient. For example, the reach of river near President Harding Rapid (river mile 43) could be an excellent study site for such work. This reach has been studied by biologists and physical scientists over the past two decades, facilitating the construction of new numerical models and the collection of new hydraulic data directly comparable with the rich historical dataset. The reach is also home to native and nonnative fishes. Hydraulically, the reach contains smaller rapids that drown out during larger discharges and a large anchor rapid (President Harding) that does not drown out. Studying the hydraulic response of both types of rapids is important to test and assess the ability of fish movement during larger flows. For full benefit of the research, these velocity studies would need to be combined with biological studies of the aquatic ecology in the river, specifically assessing the response of native and nonnative fishes to different hydraulic regimes and evaluating the ability of different age classes of fish to navigate and use the river. The results of such a study, in addition to providing important insight into the interactions of fish and river hydraulics, could then be extrapolated to the larger river to begin to assess the ability of native and nonnative fishes to migrate long distances.

Acknowledgments

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Entrainment of Semi-Buoyant Beads as a Surrogate for Larval Razorback Sucker, *Xyrauchen texanus*, into Flood-Plain Wetlands of the Middle Green River, Utah

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Abstract

Razorback sucker (*Xyrauchen texanus*) reproduction in the middle Green River occurs before spring snowmelt peak flows, when riverine habitats and flood-plain wetlands connect. Warmer temperatures and greater food production in wetlands promote faster growth and higher survival of razorback sucker larvae than the cold, food-poor mainstem river; thus, increased access to wetlands may increase recruitment of this endangered species. We undertook this study to determine the flows needed to maximize entrainment of razorback sucker larvae into wetlands to better manage spring releases from Flaming Gorge Dam, which have specifically been designed to enhance access to flood-plain wetlands. In 2005 and 2006, we used drift nets to estimate entrainment of biodegradable beads and marked razorback sucker larvae released into the Green River, though issues with sample preservation made interpretation of larval fish results difficult. In 2005, released beads were recaptured at all sampling locations and as far as 50 miles downstream. In 2006, beads were released immediately upstream from three wetlands at three or four flow levels after wetlands had connected with the river. Entrainment of beads into all sites was positively correlated with river flow volume. Results suggest that entrainment would be highest at flows greater than 18,000 cubic feet per second (ft^3/s), and that entrainment would continue to increase with increasing flows. Optimizing the peak and duration of spring flows and timing flows with the appearance of wild larvae may increase recruitment and enhance the recovery of razorback sucker.

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Introduction

The razorback sucker (*Xyrauchen texanus*) was formerly widespread throughout warmwater reaches of the Colorado River Basin, but is currently rare and as a result is federally listed as endangered because of negative impacts from physical habitat alteration and introduction and proliferation of nonnative fishes (U.S. Fish and Wildlife Service, 1991). Razorback suckers reproduce in the middle Green River, near Jensen, Utah (fig. 1); however, juvenile razorback suckers are rare, and recruitment of young fish to adulthood is limited despite annual reproduction (McAda and Wydoski, 1980; Modde and others, 1996; Bestgen and others, 2002).

It is hypothesized that flood-plain wetlands are essential for survival of early life stages of razorback sucker in the middle Green River (Modde and others, 1996; Muth and others, 1998; Wydoski and Wick, 1998). Razorback sucker larvae enter the drift in spring, usually during or just after the peak of snowmelt runoff and are entrained into flood-plain habitats. These habitats are warmer and more productive than riverine habitats and may enhance survival of larval fish (Tyus and Karp, 1991). Because of the limitations of riverine habitats in early spring, access to flood-plain wetlands after entering the drift may enhance survival of larval razorback sucker.

Because spring peak flows in the Green River were lower (on average) after construction of Flaming Gorge Dam, flood-plain wetlands connected with the river less often and only during the highest flow years (FLO Engineering, 1996). To increase frequency of river–flood-plain connections during the 1990s, levees surrounding high-priority flood plains were breached (referred to as the “levee removal study”). Flood-plain connections were either a single upstream or downstream entry or had multiple breaches (e.g., “flowthrough” wetlands; Birchell and others, 2002). These flood plains were originally breached to connect with the river at approximately 13,000 ft^3/s, a level that was expected to achieve connection in most years.

However, uncertainties arose regarding the flow magnitude and breach design that would maximize entrainment of larval razorback suckers. Thus, this study was initiated

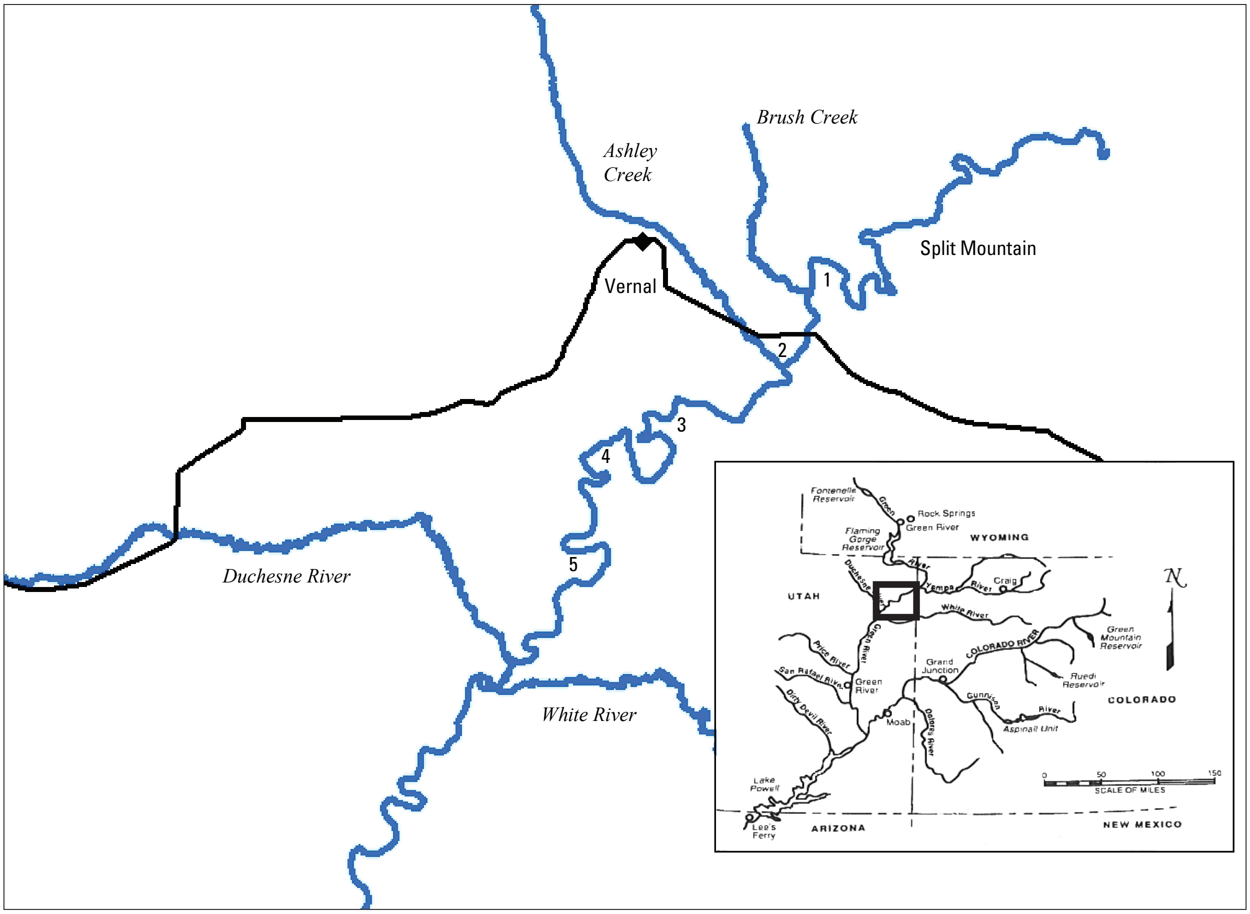

Figure 1. The middle Green River study area and flood-plain sites. 1 = Thunder Ranch, 2 = Stewart Lake, 3 = Bonanza Bridge, 4 = Stirrup, and 5 = Leota. Box in inset shows the extent of the project area within the larger regional area.

to better understand those uncertainties and better manage middle Green River flood-plain wetlands. The objectives for this study were to:

1. Evaluate larval drift and entrainment patterns downstream from known razorback sucker spawning bars in the Green River at multiple spring flow magnitudes;

2. Evaluate drift and entrainment of larvae into flood plains from other potential spawning locations at multiple spring flow magnitudes; and

3. Continue to evaluate the effectiveness of breach connections for entraining drift at various flows over the spring hydrograph.

The Green River study area is near the town of Vernal in northeastern Utah (fig. 1). Green River flow is partially controlled by Flaming Gorge Dam, located near the Utah-Wyoming border. Green River flow is supplemented by tributary flow, particularly that from the Yampa River, which is confluent with the Green River within Dinosaur National Monument. The Green River downstream from the Yampa River is designated critical habitat for recovery of the razorback sucker (U.S. Fish and Wildlife Service, 1991). The flow pattern of the Green River near Jensen, Utah, is dominated by a large spring peak generated from snowmelt runoff in the headwaters of the Green and Yampa Rivers and has a relatively low base flow during the rest of the year. Post-dam Green River flows, as measured at the Jensen gage (station 09261000), are on average lower and are consistently shorter duration peaks than during the pre-dam period (fig. 2). The middle Green River from the Yampa River to the White River is predominantly an alluvial reach with two known spawning areas and many well-developed flood-plain areas considered important for survival and recruitment of razorback sucker larvae. The two known spawning bars in this reach are at Razorback Bar (river mile (RM) 311.0 as measured upstream from the confluence with the Colorado River) and Escalante Bar (RM 306.8), both of which are just upstream from the Thunder Ranch (RM 305.8) flood-plain wetland (fig. 1). Over the course of the study, five flood-plain sites were sampled: Thunder Ranch, Stewart Lake (RM 300.0), Bonanza Bridge (RM 289.6), Stirrup (RM 275.5), and Leota (RM 257.8). Stewart Lake connects at the lowest river flow, at approximately 8,000 ft^3/s, while Bonanza Bridge connects at the

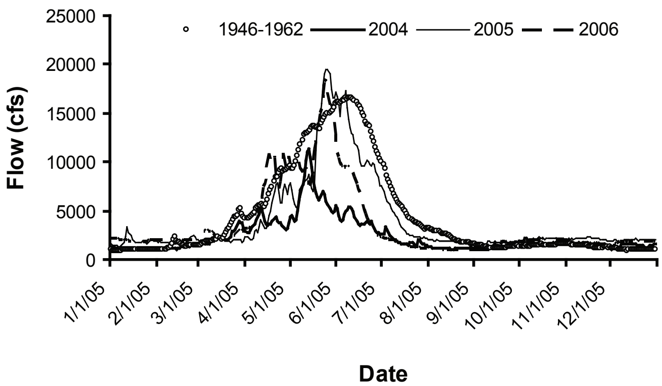

Figure 2. Mean daily average flows for the Green River near Jensen, Utah (station 09261000) for the study period, 2004–2006. Mean daily average flows for the period 1946–1962 (pre-Flaming Gorge Dam) are shown for comparison.

highest flow, about 16,000 ft^3/s. The other three flood plains connect at about 13,000–14,000 ft^3/s. Leota is the largest flood plain sampled (over 1,000 acres inundated at 18,600 ft^3/s river flow), while Bonanza Bridge and Stirrup are the smallest (28 acres each at 18,600 ft^3/s river flow). Thunder Ranch and Stewart Lake acreage values are in between these amounts at 330 acres and 570 acres, respectively.

Methods

From pilot studies in previous years, we knew that wild-spawned razorback sucker larvae were rare (Hedrick and others, 2009). Therefore, we released hatchery-reared, tetracycline-marked razorback sucker larvae (produced at Ouray National Fish Hatchery) and biodegradable, nearly neutrally buoyant beads (Key Essentials, Inc.; fig. 3) into the river. In previous studies, beads were captured at similar rates to hatchery larval fish (0.30 percent bead capture rate versus 0.36 percent larvae capture rate), although over a shorter time period (1 hour versus 4 hours, respectively; Hedrick and others, 2009). Drift net sampling occurred at flood-plain locations that were part of the levee removal study.

Figure 3. A 5-gallon bucket filled with orange beads from Key Essentials, Inc.

In 2005, approximately 1.5 million orange beads and 100,000 marked larvae were released at three different river flow levels at Razorback Bar on river right (as facing downstream; table 1). Approximately 1.5 million yellow beads and 100,000 marked larvae were also released at the same flows at Escalante Bar on river left. Releases occurred in mid to late May at 13,800 ft^3/s on the ascending limb of the hydrograph, 19,100 ft^3/s (the peak), and at 16,700 ft^3/s on the descending limb of the hydrograph. Drift material from both spawning bars was tracked over 50 river miles. Drift nets (4 meters long, 500 micron mesh size) were set in the main channel 1 mile below Razorback Bar and at four flood-plain sites: Thunder Ranch, Stewart Lake, Stirrup, and Leota (table 1). At each flood-plain location, nets were set within the levee breach and in the main river channel on the near shore, mid channel, and far shore. Net sets within most breaches were channel bottom net sets and because of the shallow nature of the breach, sampled the entire water column. Main channel nets were set from floating stations and sampled only the top portion of the water column.

River flow was measured at the U.S. Geological Survey (USGS) gaging station at Jensen, Utah (station 09261000), although at some downstream sampling sites, the substantial tributary inflows from Ashley Creek (station 09266500) and Brush Creek (station 09261700) were added to flow totals. Each site was sampled for nearly 5 hours, and the entire sampling period (release of beads and sampling of all four sites) lasted 36 hours from the release to the final sampling location.

Drift nets were emptied frequently during sampling to prevent clogging with fine debris. Samples were taken to the laboratory, and beads and larvae were picked from debris. Beads and larvae were counted and recorded for further analysis. Although samples to be processed for larval fish were preserved using 100 percent ethanol, many of the samples degraded over time, and fish were lost. In addition, flowmeter malfunction or low river and breach flows sometimes yielded inaccurate results, meaning total bead entrainment could not be extrapolated in 2005.

In 2006, we sampled only at flowthrough flood-plain wetlands because 2005 data showed these wetlands were most efficient at entraining water, beads, and larvae. We sampled at various times on the ascending and descending limbs of the hydrograph at three sites: Thunder Ranch (also sampled in 2005), Stewart Lake (sampled in 2005 as a single breach wetland, but was a flowthrough site in 2006), and Bonanza Bridge (table 1). We released 540,000 beads 1 mile above each flood plain to increase sample sizes and improve our ability to detect patterns of entrainment into flood-plain breaches. Marked larvae of different batch sizes were released as available only at Thunder Ranch.

Table 1. Date, river flow, and number/placement of nets for all sampling occasions.

[ft^3/s, cubic feet per second]

Flood plain (year)	Dates sampled	Number and location of nets	Flows sampled (ft^3/s)
Thunder Ranch (2005)	May 20, 24, and 30	2 breach, 1 far shore, 1 near shore, 1 mid-channel	13,800 ft^3/s; 19,100 ft^3/s; 16,700 ft^3/s (descending)
Stewart Lake (2005)	May 20, 24, and 30	2 breach, 1 far shore, 1 near shore, 2 mid-channel	13,800 ft^3/s; 19,100 ft^3/s; 16,700 ft^3/s (descending)
Stirrup (2005)	May 21, 25, and 31	2 breach, 1 far shore, 1 near shore, 1 mid-channel	13,800 ft^3/s; 19,100 ft^3/s; 16,700 ft^3/s (descending)
Leota (2005)	May 21, 25, and 31	2 breach, 1 far shore, 1 near shore, 1 mid-channel	13,800 ft^3/s; 19,100 ft^3/s; 16,700 ft^3/s (descending)
Thunder Ranch (2006)	May 21, 23, 24, and 30	4 breach, 4 near shore	15,200 ft^3/s; 17,200 ft^3/s; 18,600 ft^3/s; 14,500 ft^3/s (descending)
Stewart Lake (2006)	May 17, 18, 21, and 24	4 breach, 2 near shore	11,450 ft^3/s; 12,200 ft^3/s; 15,200 ft^3/s; 18,600 ft^3/s
Bonanza Bridge (2006)	May 23, 25, and 27	3–4 breach (dependant upon size of breach), 2 near shore	17,200 ft^3/s; 18,900 ft^3/s; 16,000 ft^3/s (descending)

In 2006, we added additional nets to each location. All sites except Bonanza Bridge were sampled with four within-breach drift nets. The number of nets at Bonanza Bridge varied depending on the size of the breach at the time of sampling (which increased as flow scoured the breach), but was always three or four. At Thunder Ranch, where we released larval fish, we sampled the near shore with four nets; at other sites we used two near-shore nets. On one sampling occasion, at the Stewart Lake site, we used additional nets to sample the lower (deepest) portion of the inlet channel, in addition to the usual mid-column nets. This was done because in the slow-flowing and nonturbulent Stewart Lake inlet, beads tended to sink. Ratios of captures in each zone were used to calibrate captures of beads at times when only upper zone sampling was conducted, and resultant estimated capture rates of beads in the lower and upper zones were both used to estimate total bead entrainment (Hedrick and others, 2009).

We used a different flowmeter to more reliably measure flow rates in drift net mouths in order to estimate entrainment rates. Reliable measures of net flows allowed us to determine rates of drift and water entrainment and to extrapolate total entrainment into breaches in addition to further assessing patterns of bead and water entrainment. Unfortunately, some samples remained unsorted for too long and any fish present may have degraded and were unavailable for analysis.

Total number of beads entrained in the breach was estimated by dividing the breach flow volume by the total volume of flow sampled by drift nets and multiplying that number by the total number of beads captured in the nets (2006 only). Percentage of river flow entrained and percentage of released beads that were entrained were calculated (2005 and 2006). Effectiveness of the breach to entrain drift was portrayed as the percentage of total beads captured in breach samples compared to the total number of beads captured in all main channel (near shore) and breach nets at that site (2005 and 2006).

Results

2005

Because we sampled over 50 river miles this year and incorporated two different release locations on different sides of the river, we detected patterns of bead drift within the river. We did not see complete cross-channel mixing (orange beads released on river right reaching the left river bank or vice versa for yellow beads) until downstream from the Stewart Lake flood plain, which is 11 river miles below the orange bead release site and 6 miles below the yellow bead release site. This pattern was especially prevalent at lower flows. For example, at Stewart Lake, we did see yellow beads on the near shore (opposite of their release) at the peak flow, though we did not see this at the two lower flows sampled (table 2). The

Table 2. Number of yellow beads released on river left captures per minute of sampling at the Stewart Lake flood plain (on river right) at all three sampling times, 2005.

	Stewart Lake, 2005		
	Near shoreline	Mid-channel	Far shoreline
First release	0	0.60	2.60
Second release (peak)	3.25	1.29	1.52
Third release	0.05	1.04	1.50

pattern was similar, though not as pronounced for orange beads released on river right and captured at the near shore of Thunder Ranch, which is on river left. Beads were well mixed across the channel at the two sites furthest downstream from the release locations (tables 3 and 4).

In addition to channel distribution, we compared the number of beads entrained between all of the sampling sites. Thunder Ranch entrained a larger percentage of both beads and flow than any of the other flood plains (table 5), although Stewart Lake did entrain a large percentage of flow during the first two sampling occasions. In addition, Thunder Ranch entrained the most beads and flow at the peak, suggesting that entrainment would continue to rise as flow continued to rise. This was not the case at the other flood plains, which were single breach flood plains, including the largest site, Leota. At these locations, entrainment was highest at the initial sampling occasion and dropped as flows rose, likely because of flood-plain filling. While the first pattern does apply to Leota, the site was not filling during the first or third release and was entraining water and beads only during the peak.

In addition to the correlation between percentage of flows and beads entrained, we saw a correlation between flows entrained and the number of captured beads entrained. Breaches at flowthrough sites became more effective at entraining drift material at higher flows. For example, at Thunder Ranch during the first release, the near-shore nets captured more beads than the breach nets (61 percent versus 38 percent, respectively). At the peak flow, the overall number of beads captured increased substantially, but more importantly, the number of yellow beads captured in the breach increased, while the number captured in near-shore nets declined. The percentage captured in the breach relative to the total number captured increased dramatically from the first release from 38 percent to 96 percent on the second release (table 6). This pattern was not observed at Stewart Lake or the Stirrup (single breach wetlands), but was seen at Leota (which was connected to the river only at the peak), although the number of beads captured at Leota was relatively low.

Table 3. Bead captures per minute of sampling in the near-shore and far-shore nets at the Stirrup sampling site, 2005.

	The Stirrup, 2005			
	Far shore		Near shore	
	Orange	Yellow	Orange	Yellow
First release	0.03	0.01	0.07	0.14
Second release	0.03	0.02	0.08	0.13
Third release	0.04	0.06	0.13	0.20

Table 4. Bead captures per minute of sampling in the near-shore and far-shore nets at the Leota sampling site, 2005.

	Leota, 2005			
	Far shore		Near shore	
	Orange	Yellow	Orange	Yellow
First release	0.36	0.44	0.47	0.64
Second release	0.11	0.14	0.22	0.27
Third release	0.10	0.09	0.34	0.52

Table 5. Percentage of Green River flow and released beads entrained at various Jensen gage (station 09261000) measurements at all flood-plain sites in 2005.

[ft^3/s, cubic feet per second]

2005 release	River flow (ft^3/s)	Thunder Ranch		Stewart Lake		Stirrup		Leota	
		Percent flow entrained	Percent beads entrained	Percent flow entrained	Percent beads entrained	Percent flow entrained	Percent beads entrained	Percent flow entrained	Percent beads entrained
First	13,800	0.17	0.04	0.42	0.02	0.04	0.002	0.00	0.00
Second	19,100	0.37	0.20	0.22	0.002	0.03	0.000	0.09	0.01
Third	16,700	0.22	0.14	0.12	0.0005	0.02	0.002	0.00	0.00

Table 6. Percentage of total beads captured in the breach versus those captured in near-shore nets at Thunder Ranch over all flows, 2005.

	Thunder Ranch, 2005			
	Breach		Near shore	
	Orange	Yellow	Orange	Yellow
First release	1%	38%	0%	61%
Second release	9%	96%	6%	2%
Third release	3%	62%	2%	35%

2006

Similar to results in 2005, Thunder Ranch entrained the most beads at the highest flow sampled; Stewart Lake, a flowthrough site in 2006, also entrained the most beads at the highest flow sampled (figs. 4 and 5). Bonanza Bridge, however, did not show this same pattern. In fact, we observed the highest number of beads entrained at the first flow sampled (17,200 ft^3/s; fig. 6); however, Bonanza Bridge did not connect to the river until very near the peak and was not sampled as extensively as the other two flood plains.

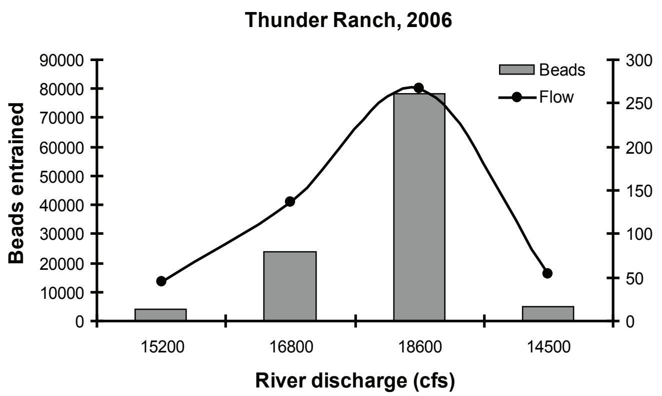

Figure 4. Bead and flow entrainment at Thunder Ranch in 2006 at four flows sampled: three on the ascending limb/peak and the last on the descending limb of the spring hydrograph.

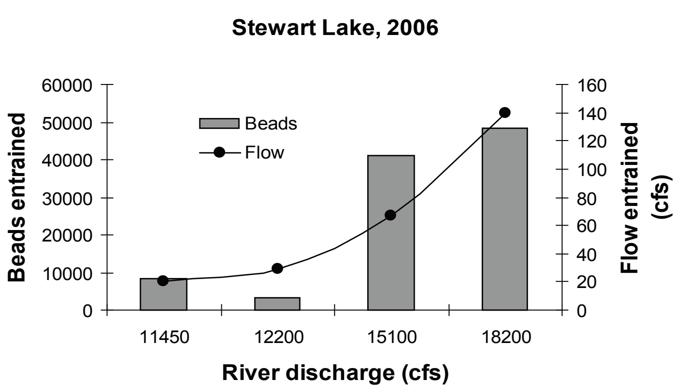

Figure 5. Bead and flow entrainment at Stewart Lake in 2006 at four flows sampled, all on the ascending limb or peak of the spring hydrograph.

Similar to what we observed in 2005, the percentage of released materials entrained was higher in 2006 at higher flows. This was true for both Thunder Ranch and Stewart Lake, but not Bonanza Bridge (tables 7, 8, and 9). Rates of entrainment mirrored the percentage of released beads captured at Thunder Ranch (table 7); however, this was not the case at Stewart Lake. Beads entrained per cubic feet per second of water entrained were highest at Stewart Lake at the second highest flow and beads per cubic feet per second in the river reached a plateau at the second highest flow (table 8). Entrainment at Bonanza Bridge did not mirror either of these other flood plains and likely was influenced by river channel morphology.

Finally, we again observed an increase in beads captured within the breach relative to those captured in the near-shore nets at higher flows (table 10), particularly at Thunder Ranch and Stewart Lake. However, at Bonanza Bridge, percentages of beads captured within the breach were similar between the first two releases and then declined at the third release.

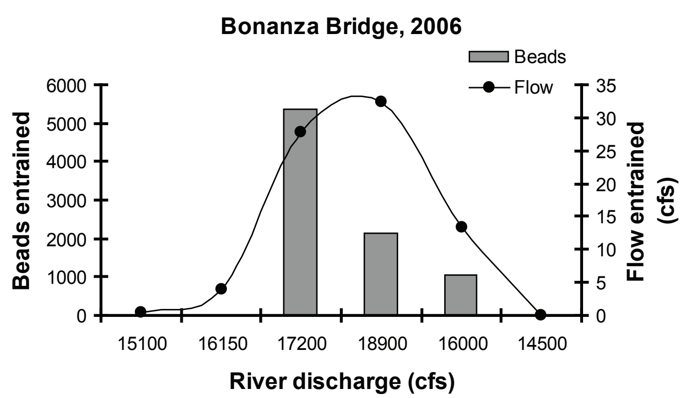

Figure 6. Bead and flow entrainment at Bonanza Bridge in 2006 at three flows sampled: two on the ascending limb/peak and the last on the descending limb of the spring hydrograph.

Table 7. Percentage of Green River flow and released beads entrained and rates of bead entrainment (beads per cubic feet per second (ft^3/s) entrained into flood plain and beads per ft^3/s in river) at various Jensen gage (station 09261000) measurements at Thunder Ranch in 2006.

	Thunder Ranch, 2006				
	River flow (ft^3/s)	Percent flow entrained	Percent beads entrained	Beads per ft^3/s entrained	Beads per ft^3/s in river
First release	15,200	0.30	0.70	88.7	0.3
Second release	16,800	0.80	4.40	173.8	1.4
Third release	18,600	1.50	14.50	294.8	4.2
Fourth release	14,500	0.40	0.90	93.7	0.3

Table 8. Percentage of Green River flow and released beads entrained and rates of bead entrainment (beads per cubic feet per second (ft^3/s) entrained into flood plain and beads per ft^3/s in river) at various Jensen gage (station 09261000) measurements at Stewart Lake in 2006.

	Stewart Lake, 2006				
	River flow (ft^3/s)	Percent flow entrained	Percent beads entrained	Beads per ft^3/s entrained	Beads per ft^3/s in river
First release	11,450	0.20	1.60	411.8	0.7
Second release	12,200	0.20	0.60	117.2	0.3
Third release	15,100	0.40	7.60	614.9	2.7
Fourth release	18,200	0.80	9.00	346.5	2.7

Table 9. Percentage of Green River flow and released beads entrained and rates of bead entrainment (beads per cubic feet per second (ft^3/s) entrained into flood plain and beads per ft^3/s in river) at various Jensen gage (station 09261000) measurements at Bonanza Bridge in 2006.

	Bonanza Bridge, 2006				
	River flow (ft^3/s)	Percent flow entrained	Percent beads entrained	Beads per ft^3/s entrained	Beads per ft^3/s in river
First release	16,700	0.16	1.08	210.2	0.3
Second release	17,400	0.17	0.40	66.5	0.1
Third release	15,900	0.08	0.19	77.1	0.1

Table 10. Percentages of beads captured within breaches and in near-shore nets for all flood plains in 2006.

[ft^3/s, cubic feet per second]

2006 river flow (ft^3/s)	Thunder Ranch		Stewart Lake		Bonanza Bridge	
	Breach	Near shore	Breach	Near shore	Breach	Near shore
11,450	-	-	431 (39%)	683 (61%)	-	-
12,200	-	-	657 (57%)	489 (43%)	-	-
15,100–15,200	249 (25%)	729 (75%)	678 (52%)	638 (48%)	-	-
16,800–17,200	959 (83%)	202 (17%)	-	-	1083 (91%)	108 (9%)
18,200–18,600	3205 (81%)	758 (19%)	814 (70%)	354 (30%)	-	-
18,900	-	-	-	-	693 (87%)	101 (13%)
16,000	-	-	-	-	301 (74%)	104 (26%)
14,500	639 (49%)	657 (51%)	-	-	-	-

Discussion

There are three main points to be learned from data collected over the course of this study. First, flood-plain sites nearest to spawning bars (i.e., less than 10 miles downstream) will not receive larvae produced on the opposite side of the river over most flow levels studied. However, while flood plains nearest to and on the same side of the river as spawning bars are likely most important for entraining greater numbers of fish, beads were captured entering all flood plains, even those 50 miles downstream from the release sites. Research on riverine drift of black fly (*Simulium nigricoxum*) larvae concluded that ability to predict larval drift distance was related to the rate of sinking and also flow velocity (Fonseca, 1999), suggesting that (1) larval fish may be transported downstream further distances than the beads studied (which tended to be captured toward the bottom of the water column) and (2) larval fish may be carried further downstream at higher flows.

The second main finding was that flowthrough sites entrain far more beads (and likely, larval fish) than sites with a single breach. This is because single-breach flood plains fill over the course of spring runoff and exchange little water with the river once full, whereas flowthrough sites entrain water for the entire connection period. In addition, results from 2006 support the conclusion that entrainment into most flowthrough flood plains increases as flows increase within the middle Green River. Highest entrainment of both beads and water occurred at Thunder Ranch in 2005 and Thunder Ranch and Stewart Lake breaches in 2006 and at the highest flows sampled. In fact, based on these results, we would expect entrainment of drift materials to continue to increase with increasing flows at these two sites. The ability of these two sites to entrain drift and their proximity to the known spawning bars within this reach highlight the potential importance of these two flood plains to recovery of the razorback sucker. However, flowthrough flood plains entrain all types of drift particles, including sediment, which may result in shallower flood plains over time. In fact, the breach at Stewart Lake is cleared of sediment annually to maintain its current riverine flow connection of 8,000 ft^3/s.

Entrainment results at Bonanza Bridge varied from what was seen in other flowthrough flood plains, which is likely a result of differing flood-plain breach and main channel morphology. There is a sandbar immediately adjacent to the breaches at the Bonanza Bridge wetland, and it is possible that as flows increased in the river, more drift was carried away from the flood-plain breaches with the thalweg, thus becoming unavailable for entrainment into the breach.

Finally, we conclude that flood-plain breaches of flowthrough sites entrain a greater overall percentage of drift at higher flows. At the highest flows sampled, we observed a greater percentage of released material recaptured in our breach nets and a smaller percentage of released beads bypassing the breach. We saw fewer beads bypassing the breach at higher flows not only in 2006 when beads were released immediately above the flood plains, but also in 2005 when beads and larvae were released halfway across the river channel and further upstream. We thus conclude that, at most flowthrough sites, fewer wild larvae will bypass the breach and more larvae will become entrained at higher flows.

Implications for Management

We can apply our findings to numerous aspects of flow and flood-plain management. Certainly, some flood plains were likely better at entraining larval fish than others, based on bead capture data. Entrainment at the Bonanza Bridge site may be improved by placing breaches further upstream, above the sand and sediment accumulation. Keeping breaches (and flood plains) free of sediment and sand accumulation may be difficult because of the formation of flood plains on inside river bends; however, entrainment will not occur as predicted for flowthrough sites if the thalweg carries drift material away from breaches during peak flows.

In addition, sedimentation must be managed within any and all upstream breaches, regardless of whether the flood plain is surrounded by a sandbar. Flowthrough flood plains that entrain more larval fish will also entrain more sediment. Sediment accumulates in breaches and flood plains over time, thus decreasing their likelihood of persistence and the likelihood that young-of-year fish will survive their first winter. It is especially important to maintain adequate breach and flood-plain depth in those flood plains expected to receive the most larval razorback sucker in order to ensure their persistence over time. In order to ensure maximum entrainment rates of larval fish, sediment removal must be actively undertaken in upstream breaches or additional breach morphologies must be researched to increase entrainment while minimizing sedimentation.

Our results show that higher flows entrain more larval fish at flowthrough sites. Not only does entrainment increase as flows increase, cross-channel mixing increases as well, meaning that more larvae produced at Escalante Bar will be available for entrainment at Stewart Lake (or vice versa for Razorback Bar and Thunder Ranch) in higher flows. Depending on the number of larval fish produced at each spawning bar, higher flows could substantially increase the number of larval fish available for entrainment at upstream flood plains in the middle Green River.

We also now have the ability to predict how many larvae are entrained at different flows and in different flood plains, depending on flood-plain type (flowthrough versus single breach) and distance from spawning bars. Survival rates of razorback sucker within flood-plain sites, even in the presence of nonnative fish, have been analyzed in previous studies (Brunson and Christopherson, 2005). A next logical step is to synthesize all available studies, to better link razorback sucker life-history information with flow and flood-plain entrainment data. One outcome may be evaluation of the ability of the flow recommendations currently in place (Muth and others, 2000) to provide the necessary levels of entrainment and recruitment in flood plains of interest and the potential of each to contribute to recovery of the species over a range of flows.

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How Has Over-Allocating the Colorado River Affected Species in the Gulf of California?

By Kirsten Rowell[1] and David L. Dettman[2]

Abstract

The Colorado River previously was a major influence on the upper Gulf of California. Today, virtually no river flow reaches the sea, resulting in the elimination of wetlands and estuarine habitat. While there is a great deal of focus on the ecological effects of dam operations along Colorado River corridor, surprisingly little research attention has been paid to the ecological impacts of diverting river flow from the Gulf of California. Here we take some first steps in addressing *How has the marine ecosystem responded to the cessation of the Colorado River?* We compare the chemistry and annual rings in fish otoliths (ear bones) from before the dams (≈5,000 years before present) and after dams (contemporary otoliths) to determine pre-dam conditions and fish response to damming. We focus on two endemic and economically important species: the endangered *Totoaba macdonaldi* and the threatened *Cynoscion othonopterus.* We found that Colorado River water was an important feature of these two species' nursery grounds. Growth increments document that totoaba grew twice as fast and matured in half the time before the dams; oxygen isotope ratios link this finding to the presence of Colorado River flows. In summary, the geochemistry embedded in otoliths provides the first layer of evidence that Colorado River flow is an important resource for fish in the Gulf of California, and the loss of flow impacts demographics and life history of these species.

Introduction

Before dams and diversions, the Colorado River had large and variable flows (fig. 1). These flows maintained an estuary comprising about 4,000 square kilometers (km^2) of the uppermost Gulf of California (Lavín and Sánchez, 1999) and a mixing zone of fresh and marine water (brackish water) extending about 65 kilometers (km) south of the mouth of the river (Carbajal and others 1997; Rodriguez and others, 2001).

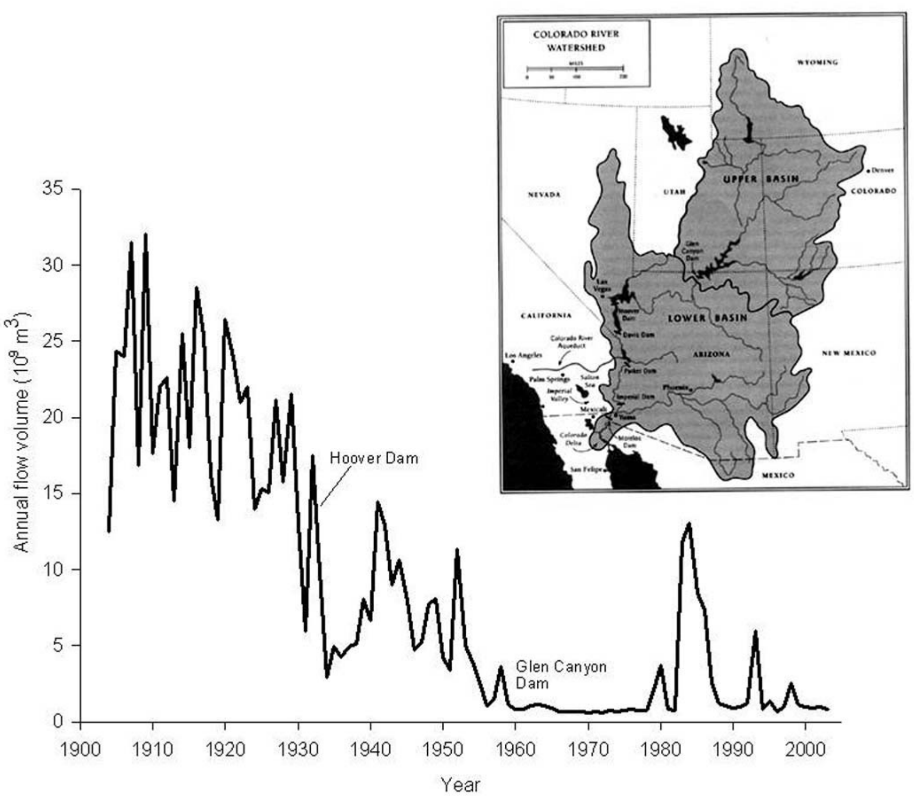

Figure 1. Colorado River annual flow volume (10^9 m^3) below Yuma Main Canal at Yuma, Arizona, (station 09521100), for years 1904–2003. Years where the hydrograph is flat depict the annual flow of 1.8 × 10^9 m^3 required by the 1944 Mexican water treaty.

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Today, flow rarely connects the Colorado River to the Gulf of California (fig. 2) (Glenn and others, 2007). The only water that crosses the U.S. border in normal years is the annual flow of 1.8 x 10^9 cubic meters (m^3) required by the 1944 Mexican water treaty. This water is almost entirely consumed by municipal and agricultural users in Mexico, though a fraction probably reaches the gulf by way of subsurface flow (Hernández-Ayón and others, 1993; Lavín and others, 1998; Glenn and others, 2007). Diminished flow has resulted in the shrinking of Colorado River estuarine habitat (Carbajal and others, 1997). Today, the northernmost portion of the Gulf of California is hypersaline compared to the adjacent open marine waters (Lavín and others, 1998). The combination of arid environment, high evaporation rates, and decreased river flow has resulted in salinities between 36 and 40 parts per thousand (‰) in the upper gulf (Hernández-Ayón and others, 1993; Lavín and Sánchez, 1999). We use prehistoric remains from fish (the endangered totoaba (*Totoaba macdonaldi*) and the economically important and threatened gulf corvina (*Cynoscion othonopterus*)) to investigate how fish in the upper Gulf of California lived in the past and how the cessation of the Colorado River flow has changed their ecology. The skeletal remains of these fish provide tools to look into the past and test the hypothesis that the Colorado River was an essential component to the natural history of these fish at risk.

One of the difficulties researchers face when investigating ecological impacts of diminished Colorado River flow into the gulf is the lack of empirical ecological information from before upstream river regulation. Knowledge of the marine environment before the 1960s is limited to ecological knowledge from fisherman (fig. 3; Sala and others, 2004; Sáenz-Arroyo and others, 2005) and recorded fisheries catch (Flanagan and Hendrickson, 1976). Fishermen commonly cite the lack of river flow into nursery and spawning habitat as a reason for decreased stock. In addition, declines in fisheries landings of the top-predator fish, totoaba, and shrimp in the upper gulf are correlated with reduced river flow into the gulf (Flanagan and Hendrickson, 1976; Galindo-Bect and others, 2000). These data point to a large ecological impact of upstream river regulation, but both of these data sources are notoriously difficult to interpret, and neither provides information on potential mechanisms that might link upstream

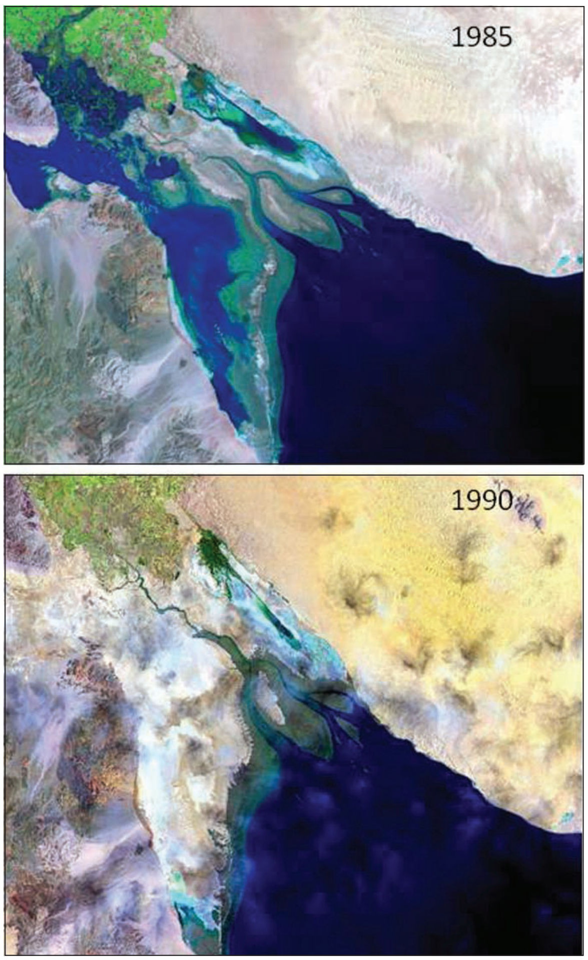

Figure 2. Landsat images of the Colorado River delta, illustrating the extensive wetland habitat created by the Colorado River. These photographs of the Colorado River mouth and the upper Gulf of California were taken during May of 1985 and 1990. In 1984, an abnormal snowmelt triggered a release of river water at the southern international boundary in excess of the 1944 Mexican water treaty. The inundation of extensive mudflats created kilometers of shallow wetlands and protected nursery habitat. The 1990 photograph depicts a typical year—habitats created by the Colorado River flow are absent and the river does not connect to the sea.

Figure 3. Totoaba fishermen in San Felipe, Baja de California Norte, Mexico, in the 1950s. Totoaba were fished while they spawned in the mouth of the Colorado during the spring high flows. Photographs like these are some of the only documentation of how productive the upper Gulf of California was before the Colorado River flows vanished. Today, a fisherman would be hard-pressed to catch an endangered totoaba of this size, even though totoaba have been internationally protected for almost 35 years.

management with changes in the Gulf of California. Here we use the chemistry and growth rates of skeletal fish remains to provide insights to how fish have responded to the wholesale removal of the Colorado River flow to the upper Gulf of California. We test the hypothesis that the Colorado River flow created critical nursery habitat for multiple species of marine fish and that by cutting off flow to the gulf these fish were negatively impacted.

Methods

Fish otoliths (ear bones; fig. 4) are calcium carbonate bone-like structures that are composed of daily additions of small amounts of aragonite. Because the fast-growth (summer) and slow-growth (winter) portions are visibly different, the otoliths have visible annual rings, creating a record of environmental conditions and rates of fish growth (Campana and Thorrold, 2001). The growth increments are essentially chapters in the life of a fish, recording growth, onset of sexual maturation, and water chemistry. The timekeeping property makes fish otoliths great candidates for the investigation of changes in habitat use and associated life-history parameters. Because otoliths are inert, their chemical composition is conserved over the course of the fish's life, and they are easily preserved as post-mortem remains for thousands of years. Otoliths can thus be thought of as time capsules, and the elements trapped in the aragonite can be used as environmental records; combining these two properties allows us to compare ancient and modern life history and habitat use for these fish by comparing otolith chemical records (Campana and Thorrold, 2001). In this way, otoliths can be interpreted to help establish ecological baselines for ecosystems and species of concern.

We use the chemistry of otoliths from fish in the upper Gulf of California to determine the presence of Colorado River flow, which will provide baseline information about the ecosystem and the species that lived there before river diversions. Pre-dam otoliths found in aboriginal shell middens (dated 1,000–5,000 years before present) along the coast provide a pre-dam record of environmental conditions. We compare the information from pre-dam otoliths to post-dam otoliths. By using environmental markers formed by changing isotopic ratios of oxygen embedded in otoliths, we are able to examine the impacts of diminished Colorado River flow on environmental conditions (salinity). (For a full explanation of methods please refer to Rowell and others, 2005; Rowell, 2006; Rowell and others, 2008a; Rowell and others, 2008b.) By comparing these data to changes in growth rates (also measured from the otolith), we can link these environmental changes to shifts in fish life history. This method allows us to address two ecological and economically important questions: (1) Do these fish use nursery habitat provided by the Colorado River and (2) How has altering this habitat, by diverting the Colorado River away from the Gulf of California, affected the endangered totoaba?

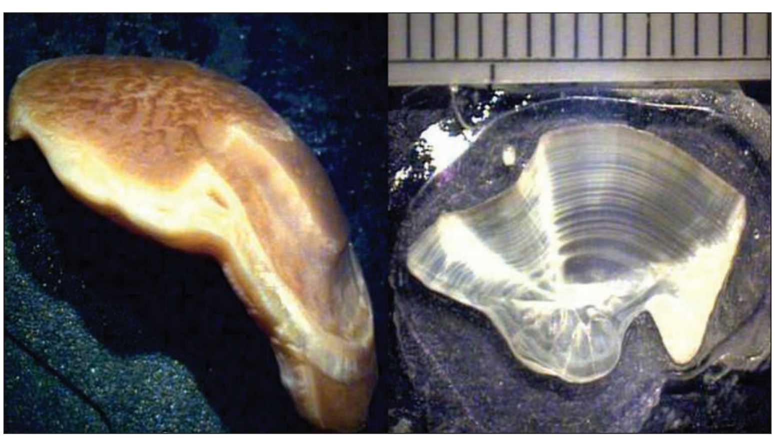

Figure 4. A totoaba otolith (≈5,000 years old) from an aboriginal shell midden along the coast of Sonora, Mexico. Annual rings can be seen in cross section. The oxygen isotopes embedded in the calcium carbonate of the growth increments document when these fish are residing in estuarine habitat created by Colorado River flows to the Gulf of California.

Results

Nursery Habitat Created by the Colorado River

In the Gulf of California, the isotopic ratio of $^{18}O/^{16}O$ ($\delta^{18}O$) of water tracks salinity (Dettman and others, 2004). Fish otoliths record the oxygen isotope ratio of the water they live in, and because freshwater has a more negative $\delta^{18}O$ value than marine water, we can determine when fish were living in the estuarine conditions provided by the Colorado River. Today's Colorado River water has a value of ≈ –12 ‰ Vienna standard mean ocean water (VSMOW), and upper Gulf water is ≈ +0.6 ‰ VSMOW (Dettman and others, 2004). We found that otolith $\delta^{18}O$ values for the summer growth are positively correlated with Colorado River flow (Rowell and others, 2005). Pre-dam otoliths from totoaba and gulf corvina have $\delta^{18}O$ values that are significantly (statistically) more negative than open-ocean otolith values in the juvenile portions of the otolith, indicating that these fish prefer the less saline nursery habitat provided by river flow (Rowell and others, 2005; Rowell and others, 2008b). Salinity estimates made from contemporary fish suggest these fish were seeking out habitats that were up to 11‰ less saline for early growth (Rowell and others, 2005). In fact, the oxygen isotope ratio of the prehistoric otoliths suggests that both totoaba and gulf corvina may spend their first 3 years in the Colorado River estuary habitat, when freshwater flowed to create a brackish estuarine habitat, before moving into the open marine waters (Rowell and others 2005; Rowell and others 2008a).

Life-History Shifts Associated with the Absence of the Colorado River

So why do these fish prefer to live in the Colorado River estuary when it is available? Using otolith growth increments, we can compare growth rates between pre-dam fish that used the Colorado River estuary habitat to growth rates of modern fish that do not have access to these nursery habitats. We found that pre-dam totoaba grew fast enough to reach sexual maturity in 2 years, in contrast to today's totoaba, which do not reach this size until 5 to 7 years of age (fig. 5; Rowell and others, 2008a). In other words, totoaba living today, without the Colorado River estuary as nursery habitat, grow much slower and reach maturity much later than pre-dam fish. Growth for both pre-dam and post-dam fish is strongly correlated with the Colorado River flow—especially in the first year of growth (Rowell and others, 2008a). Faster growth and lower age at maturation are both tightly linked with increased probability of survival to breeding age, increased number of breeding events an individual will have in their lifetime, and increased annual fecundity. By reducing growth and increasing age at maturity, the probability of totoaba recovering from over fishing is further compromised (Reynolds and others, 2005). Our results indicate that water diversion acts as a "bottom up" pressure, causing large reductions in the quality of nursery habitat and reducing population viability for this once economically important and now endangered fish (Rowell and others, 2008a).

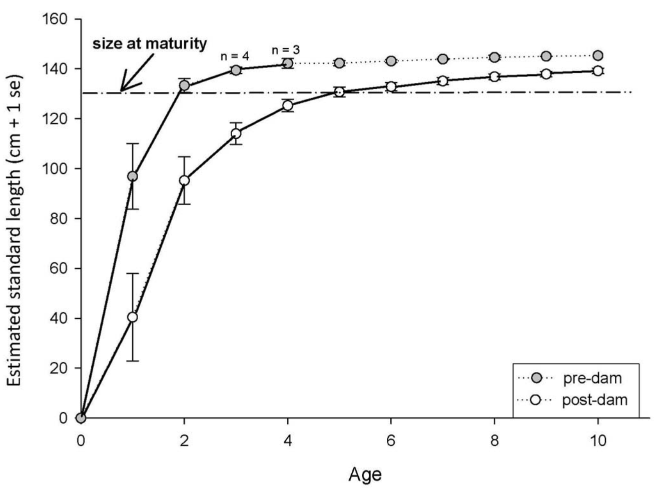

Figure 5. The growth curve and body size of pre-dam and post-dam totoaba, showing slowed growth and delayed maturation of totoaba that live in the absence of river water. The dotted line indicates where sample size equals two (from Rowell and others, 2008a).

Implications for Management

Our results indicate that upstream river diversions have had major impacts on two of the most important fin fisheries in the upper Gulf of California. The endangered totoaba, an apex predator in the system (Lercari and Chávez, 2007), once supported a thriving fishery, and gulf corvina is one of the most economically important fin fish in the region. The severe reduction of Colorado River flow to the gulf has reduced or eliminated the preferred estuarine nursery habitat for these fish. These alterations appear to have deleterious results to the population biology of the endangered totoaba.

The mechanisms for slowed growth in totoaba in association with reduced river inflow are not yet known, but two likely hypotheses include a decrease ecosystem productivity, similar to what is documented by Galindo-Bect and others (2000), or a decrease in optimal nursery habitat conditions. When Colorado River flow diminished in the 1960s, so did the brackish water habitat that functioned as spawning and nursery grounds for fish and invertebrates (Cisneros-Mata and others, 1995; Galindo-Bect and others, 2000; Rowell and others, 2005; Rowell and others, 2008b), and the riverine nutrients that fuel the high-productivity characteristic of coastal habitats were also reduced. Few watersheds in the Gulf of California deliver inland nutrients, but the few that do reach the gulf have profound impacts on regional productivity—increased productivity can be observed hundreds of kilometers from the mouth of a river (Beman and others, 2005). The Colorado River drains the Southwestern United States (fig. 1) and represents one of the largest abiotic influences on the upper gulf. The large pulses of snowmelt waters, sediments, and nutrients likely influenced the local oceanography (Carbajal and others, 1997; Lavín and others, 1998), built natural sediment levies (creating protected wetlands; fig. 2), and spurred pulses of higher productivity in the region (Rowell, 2006). Whatever the mechanism is between growth and river flow, it is clear that the Colorado River is an important component to the upper region of the Gulf of California.

There is no doubt that aggressive fishing practices, such as targeting the breeding aggregations, also had a strong negative impact on totoaba. Totoaba were fished heavily from the 1940s until they were listed as endangered in 1975 (Flanagan and Hendrickson, 1976; Cisneros-Mata and others, 1995). Since that time, totoaba have been protected by the Convention on International Trade in Endangered Species of Fauna and Flora (CITES), as well as legislation in the United States and Mexico to protect endangered species. Despite nearly 35 years of protection, totoaba populations have still not recovered (Cisneros-Mata and others, 1995; Lercari and Chávez, 2007), suggesting that something other than fishing is preventing the recovery of this species. Our results suggest the loss of Colorado River water as a large contributor.

The Colorado River provided habitat that increased diversity, benefited key fisheries, and increased the resiliency of the upper Gulf of California ecosystem (Levin and Lubchenco, 2008). Restoring seasonal (spring) flows may bring back keystone ecosystem functions by recreating wetland habitats and brackish estuarine inlets and by increasing the quality of nursery habitat and local productivity. Research supports a bottom-up approach that ripples through the ecosystem, benefiting the coastal fisheries and the environment. Galindo-Bect and others (2000) calculate an increase of only 30.8×10^7 m^3·year of Colorado River water could double shrimp production in the upper gulf (shrimp are also a part of the gulf corvina and juvenile totoaba diet). Glenn and others (1996) estimate that returning only 0.5 percent of mean annual flow could sustain the lush riparian and aquatic habitats in the terrestrial portion of the Colorado River delta, which is an important migratory bird habitat (Pitt and others, 2000; Glenn and others, 2001). In addition, the near extinction of the endemic clam, *Mulinia coloradoensis,* is attributed to the decline in river flow (Rodriguez and others, 2001), and the decline of this species may have led to the decline of its predators (Cintra-Buenrostro and others, 2005). The connection between declining diversity in marine ecosystems and alterations in estuaries has been observed in other systems (Kennish, 2002; Lotze and others, 2006). Recognizing and documenting the importance of rivers to the productivity of estuarine nursery habitat is critical for responsible management of the world's large rivers and adjacent coastal habitats (Drinkwater and Frank, 1994), which support the majority of economically important fisheries.

The importance of the Colorado River water to the marine ecosystem and Mexican fisheries adds complexity to managing a river that is already over allocated and is subject to increasing demand. Because approximately 90 percent of the river's annual flow is diverted for use in the United States, and the remaining 10 percent is used for urban and agricultural purposes in Mexico, allocation of restoration flows for the estuary will require bi-national efforts. While impacts of U.S. river regulation on the Colorado River delta traditionally have been ignored, the United States may have social incentives to address them. The downstream ecological effects may cascade through the marine ecosystem and into social and economic systems, spanning political boundaries.

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Salinity Tolerances for Egg and Larval Stages of Razorback Sucker

By James R. Stolberg[1] and Michael J. Horn[2]

Abstract

The success of numerous habitats currently being used to rear razorback suckers (*Xyrauchen texanus*) along the lower Colorado River has been somewhat hit or miss in terms of numbers of fish produced and overall survivorship. One of the key problems has been determining what factors result in a successful habitat. Both high salinity values and low dissolved-oxygen concentrations are thought to be problems in several of the current areas being used to rear these fish. To determine the effects of high salinity on in-pond recruitment and early life stages, razorback sucker eggs and larvae were exposed to a range of different salinities to determine critical lethal limits of hatching and survival. Egg and larval responses were measured as percent hatch and percent mortality at 72 hours, respectively. Larvae were also monitored for 45–60 days at all experimental salinities to determine long-term survivability. Long-term survivability refers only to the larval fish stage or until larvae reach approximately 25 millimeters total length and transition to juvenile fish. Successful hatching occurred at salinities up to 12,000 microsiemens per centimeter (µS/cm), while lethal salinity to 50 percent of the larvae occurred at >27,000 µS/cm. Larvae were also shown to be capable of long-term survival at 20,000–23,000 µS/cm.

Introduction

The Lower Colorado River Multi-Species Conservation Program (LCR MSCP) is developing 360 acres of backwater habitat for razorback sucker (*Xyrauchen texanus*) and bonytail (*Gila elegans*), two endangered fishes of the Colorado River Basin (fig. 1). Strategies for establishing suitable habitat include making small or large changes to existing backwaters as well as creating new backwaters through excavation of undeveloped land. Most of these created backwater habitats will be flood-plain ponds and sloughs isolated from the main river channel. Once created, these habitats will be managed and maintained as native fish refugia. Because of high air temperatures, low humidity, and limited hydrologic exchange with the adjacent river, salinity typically increases in isolated flood-plain ponds along the lower Colorado River. Freshening of these ponds will need to occur periodically to reset water-quality conditions. In this 2-year study, we evaluated salinity tolerances for egg and larval stages of razorback sucker in an effort to help fishery managers develop freshening schedules for these backwater habitats as well as aid in future site selection.

Figure 1. Ponds developed for native fish at the Imperial National Wildlife Refuge, Yuma, AZ (Bureau of Reclamation photograph by Andy Pernick).

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Study Objectives

An experimental study was designed to determine critical lethal limits, with respect to salinity, for razorback sucker eggs and larvae. This was accomplished by determining the maximum salinity at which eggs can successfully hatch, the maximum salinity levels at which larval fish can survive for 72 hours (h), and the long-term survivability of larval fish under different salinities for a period of 1–2 months. The long-term survival of larval fish was observed until their transition to juveniles as described by Snyder and Muth (2004). Field work associated with this study took place in the LCR MSCP's river reach 2, Lake Mohave, Arizona and Nevada.

Methods

Adult razorback suckers were collected by trammel net and electrofishing from shoreline areas of Lake Mohave in March 2007 and 2008 (fig. 2). Seven female and 9 male razorbacks were captured in 2007, and 12 females and 8 males were captured in 2008 for use as brood fish. Eleven hundred larval razorbacks also were captured during the 2007 study year to ensure their availability for trials if egg hatching was unsuccessful. Adult fish were separated by sex and held in separate live wells for a period of 18–24 h before being manually spawned. Fish were stripped by applying hand pressure to the ventral and lateral sides of ripe individuals in a head to tail direction. Eggs from females and sperm from males were captured simultaneously in the same 9.5 liter (L) container partially filled with one of the experimental salinities (2007: 1,000, 3,000, 6,000, 10,000, 15,000, and 20,000 µS/cm; 2008: 10,000, 12,000, 14,000, 16,000, and 18,000 µS/cm). Multiple females were used in each spawning when possible, and multiple males always were used.

Figure 2. Razorback spawning group, Tequila Cove, Lake Mohave, NV (Bureau of Reclamation photograph by Jon Nelson).

Salinities were prepared by mixing deionized (DI) water with measured amounts of Instant Ocean® synthetic sea salt. Salinity values were selected on the basis of tolerances of associated game fish as well as to provide us with a wide range of salinities for study. Striped bass (*Morone saxatilis*) eggs and larvae were able to develop and survive at salinities up to 14,000 µS/cm (Morgan and others, 1981). Lethal effects of salinity for Colorado pikeminnow (*Ptychocheilus lucius*) have been determined to be in the 18,000 µS/cm range (Nelson and Flickinger, 1992), and flathead catfish (*Pylodictis olivaris*) tolerances have been shown to be even higher, averaging over 20,000 µS/cm at 18 degrees Celsius (°C) (Bringolf and others, 2005). Other freshwater fish species that are found in isolated habitats also have considerable salt tolerances, often in excess of 20,000 µS/cm (Ostrand and Wilde, 2001).

During fertilization, gametes were gently mixed together, and calcium bentonite was added to prevent fertilized eggs from clumping together or from adhering to the side of the container. Fertilized eggs were then transferred to floating Nitex® cloth hatching trays. Before transfer, hatching trays were placed in large containers of corresponding salinities in preparation for the fertilized eggs. Eggs were allowed to water harden overnight and were then removed from hatching trays by using a small dip net. Eggs were placed into 3.8-L aquaria bags with sufficient amounts of corresponding saline water and arranged in a small cooler for transport to the laboratory.

The laboratory portion of this study was conducted at the Bureau of Reclamation fisheries office in Boulder City, NV, from March to early May of both years. The laboratory was outfitted with twenty 38-L aquaria before spawning the fish. Egg tanks were set up in triplicate: three tanks for each of the experimental salinities, and each tank was filled with approximately 8 L of water at the required salinity. A single 25-centimeter (cm) x 40-cm floating hatching tray was placed in each tank, and tanks were numbered for individual identification. Egg densities for all spawning salinities were estimated volumetrically on the basis of measurements of eggs per milliliter. Eggs from individual spawning salinities were divided equally between hatching trays in the three tanks (fig. 3). With the exception of the 15,000 µS/cm spawning in 2007, multiple females were used in each spawning. Eggs from each spawning were mixed together for transport, and assuming each adult fish supplied viable gametes, all tanks received fertilized eggs of mixed parentage. Total egg volumes varied between salinities as a result of the individual fecundity of the female or females used.

For the duration of this experiment a 12-h light, 12-h dark photoperiod was maintained to mimic vernal conditions. Daytime hours were sustained using both natural and overhead artificial light. Water temperatures for egg tanks were maintained between 18 and 20 °C, and water exchanges were performed daily to prevent fouling during incubation. Researchers took great care to disturb eggs as little as possible. Fungal growth was also a concern at this stage, so each hatching tray was dipped in a 1:150 formalin solution. In

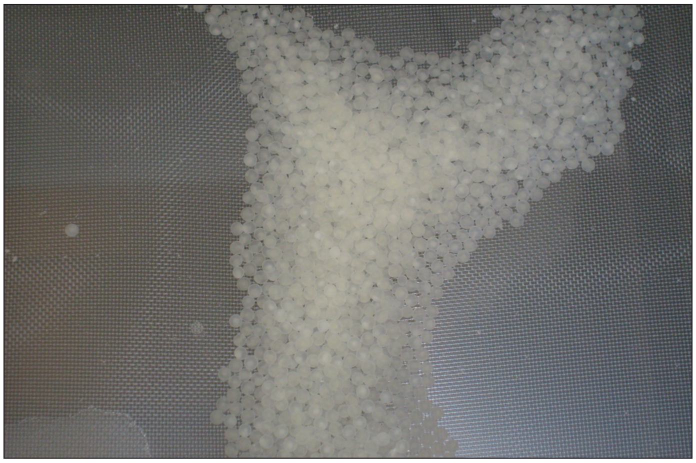

Figure 3. Floating Nitex® cloth hatching tray with razorback eggs.

addition, egg tanks were examined routinely for fungus, and dead (white/opaque) eggs were removed.

Once hatch larvae were swimming, hatching trays were removed and tanks were thoroughly cleaned. At this time all fish were counted and combined into single tanks of their respective salinities. One hundred and fifty larvae from each of the combined tanks were separated and placed in individual tanks—one tank for each cohort of 150 at the salinity in which they were spawned. These larvae acted as the control group for the duration of the experiment.

Throughout the larval portion of the experiment, we performed water changes on all tanks every 1 to 2 days. As larval yolk sacs were absorbed, we began feeding twice daily using brine shrimp. Tanks were cleaned before each feeding, and brine shrimp were siphoned into small dip nets and rinsed with DI water before being introduced into the tanks. Salinity readings from each tank were taken routinely using a Hydrolab Quanta® meter. Water temperatures for each tank were also recorded during salinity readings. Temperatures averaged between 18.5 and 19.75 °C for individual tanks.

2007 Larvae Trials

Salinity toxicity tests were begun by observing larvae in the salinity in which they were spawned for 168 h. During this period, salinity and temperature measurements were taken, and mortalities were recorded as they occurred. Also during this period, an additional six aquaria were set up for use as long-term holding tanks. These tanks were used to determine long-term survival as well as provide space for larvae not being used immediately during the experimental trials that followed. After 7 days, no significant mortality was observed in spawning salinities that successfully produced larvae. Significant mortality was defined as mortality of 10 percent or greater.

LC_{50} trials (72-h durations) began with larvae from all salinities being exposed to each of the higher experimental salinities (fig. 4). As was done with the control group, cohorts of 150 larvae were used in this trial. Tanks were observed routinely each day, and mortalities were counted and removed as they occurred. Percent mortality was recorded at 72 h.

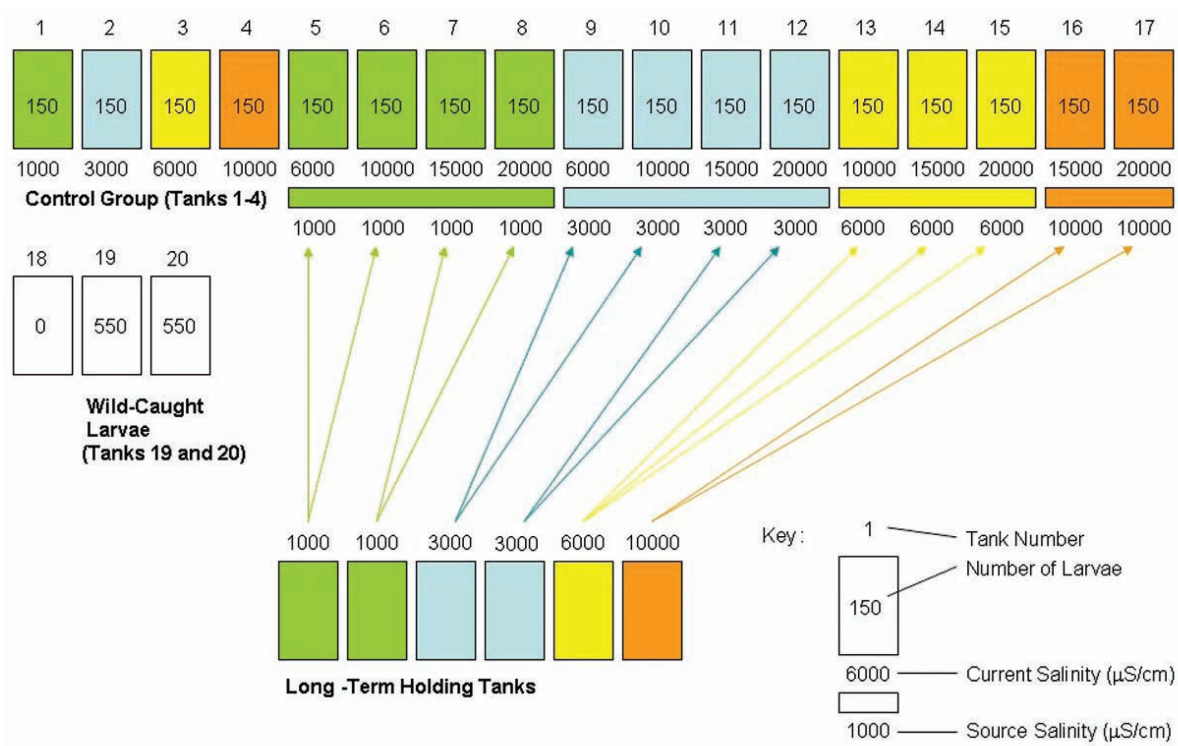

Figure 4. First trial tank setup (prepared by Dr. Mike Horn with modifications by James Stolberg).

Information gathered from the first trial indicated that additional experimental salinities would be required to determine the salinity tolerance of razorback larvae. Tanks from the first trial were emptied, and surviving larvae were transported to Willow Beach National Fish Hatchery to be reared in captivity and eventually released into Lake Mohave. Some larvae from the first trial were retained to ensure adequate numbers from each of the salinities would be available for the second experimental trial. Salinities of 23,000, 26,000, and 29,000 µS/cm were prepared for the second 72-h LC_{50} trial, and larvae from 1,000, 6,000, 10,000, and 20,000 µS/cm were exposed in the same manner as previously described. Less than 450 larvae were available from the 20,000 µS/cm source following the previous trial. Cohorts of 104, 141, and 126 larvae were used in this instance. Percent mortality was again recorded at 72 h.

2008 Larvae Trial

For the 2008 study year, two changes were made with respect to the larval trial. First, cohort sizes were doubled from the previous year to 300 larvae per tank. This was done to provide more flexibility with larvae from all salinities if the need to examine other effects, such as relative growth or condition factor, arose. Second, larvae were exposed to higher salinities incrementally, as opposed to moving them directly from low to high salinities. Findings from the first study year indicated that survival of larvae may be improved when they are tempered from lower to higher salinities. This period of acclimation more closely mimics natural salinity increases and gives us a better idea of how this species may react in ponds along the lower Colorado River. In addition, tempering razorback larvae may reduce any "shock" response associated with moving them from relatively low salinities directly into higher salinities.

Control tanks for larvae spawned in 10,000 and 12,000 µS/cm were set up in triplicate. An additional four tanks housed 14,000 µS/cm larvae for long-term observation. For tempering trials, six 38-L aquaria were used. Three of these aquaria contained larvae from the 10,000 µS/cm spawn, and three had larvae from the 12,000 µS/cm spawn. Tempering was accomplished by increasing salinities in each tank at a rate of 500 µS/cm per day. Each increase was followed by a 24-h acclimation period before salinities were increased again. Tempering continued until significant mortality was observed at which time salinities were held at their current values and larvae were monitored for 72 h.

Results

2007 Hatch

Four to 10 days were required for the complete hatching of eggs at all salinities. Eggs fertilized in 1,000, 3,000, 6,000, and 10,000 µS/cm developed normally, and larvae began swimming by 24-h post-hatch. Eggs in 15,000 µS/cm tanks were observed to be of comparatively reduced size. This was likely because of the osmotic effects of this higher salinity. Larvae from these tanks hatched early, were small, and were few in number. None survived past 16 h. No development was observed in 20,000 µS/cm tanks.

Hatch rates were variable among salinities with a successful brood (table 1). It may be possible to increase hatching percentages for the salinities that successfully produced larvae with improvements to the methods used. It is unlikely that hatch for salinities ≥15,000 µS/cm can be successful. Experimental salinities for the second study year were based on these findings and chosen to more accurately define the upper salinity tolerance for successful egg development.

Table 1. 2007-Mean (±SD) percent hatch of razorback sucker eggs subjected to experimental salinities. Number of eggs and larvae from eggs are combined totals for the three replicate treatments.

[µS/cm, microsiemens per centimeter; mL, milliliter]

Spawning salinity (µS/cm)	Number of eggs *	Larvae from eggs	% Hatch
1,000	7,500	4,115	54.8 ± 4.7
3,000	7,800	4,421	56.6 ± 1.5
6,000	3,750	1,125	30.0 ± 2.3
10,000	7,200	1,579	21.9 ± 9.4
15,000	2,400	0	0
20,000	8,750	0	0

* Number of eggs estimated on the basis of 50 eggs/mL measurement.

2007 Larvae Trials

The first larval trial resulted in limited mortality. Mortality at 72 h ranged from 0 to 26 percent, which was insufficient for LC_{50} criteria. Larvae were kept in these tanks an additional 240 h for observation. This additional period resulted in minimal change to mortality percentages (table 2).

For the second trial, larvae were initially observed for the predetermined 72-h period. Observations, however, were extended to 312 h after improved survival was noted in the 20,000 µS/cm source tanks (table 3). Larvae from 1,000, 6,000, 10,000, and 20,000 µS/cm exposed to 23,000 µS/cm during the second trial did well. Percentage of mortality was low and ranged from 0 to 18 percent over 72 h. Greater than

Table 2. 2007-First larval trial percent mortality at 72 h and 240 h.

[h, hour; µS/cm, microsiemens per centimeter]

Tank #	Source (µS/cm)	Current (µS/cm)	72 h		240 h	
			Mortalities	%	Mortalities	%
5	1,000	6,000	15	10.0	16	10.6
6	1,000	10,000	21	14.0	23	15.3
7	1,000	15,000	38	25.3	39	26.0
8	1,000	20,000	12	8.0	16	10.6
9	3,000	6,000	2	1.3	4	2.6
10	3,000	10,000	4	2.6	9	6.0
11	3,000	15,000	10	6.6	15	10.0
12	3,000	20,000	40	26.6	46	30.6
13	6,000	10,000	0	0	1	0.6
14	6,000	15,000	1	0.6	2	1.3
15	6,000	20,000	4	2.6	9	6.0
16	10,000	15,000	1	0.6	2	1.3
17	10,000	20,000	23	15.3	23	15.3

Table 3. 2007-Second larval trial percent mortality at 72 h, 312 h, and time to ≥ 50 percent.

[h, hour; µS/cm, microsiemens per centimeter; N/A indicates < 50% mortality for trial duration]

Tank #	Source (µS/cm)	Current (µS/cm)	72 h		312 h		≥ 50% Mortality		
			Mortalities	%	Mortalities	%	Time(h)	Mortalities	%
7	1,000	23,000	27	18.0	29	19.3	N/A	N/A	N/A
8	1,000	26,000	77	51.3	139	92.6	72	77	51.3
9	1,000	29,000	150	100.0	150	100.0	24	147	98.0
10	6,000	23,000	3	2.0	5	3.3	N/A	N/A	N/A
11	6,000	26,000	29	19.3	121	80.6	96	75	50.0
12	6,000	29,000	150	100.0	150	100.0	24	110	73.3
13	10,000	23,000	4	2.6	11	7.3	N/A	N/A	N/A
14	10,000	26,000	129	86.0	143	95.3	72	129	86.0
15	10,000	29,000	132	88.0	141	94.0	24	75	50.0
16	20,000	23,000	0	0	0	0	N/A	N/A	N/A
17	20,000	26,000	5	3.5	68	48.2	N/A	N/A	N/A
18	20,000	29,000	2	1.6	108	85.7	216	68	53.9

50 percent of larvae from 1,000 and 10,000 µS/cm exposed to 26,000 µS/cm died within 72 h, and larvae from 6,000 and 20,000 µS/cm died at 96 h and 312 h, respectively. This range, 72–312 h, suggests that fish from the 20,000 µS/cm source may have survived longer because they had been acclimated to a significantly higher salinity for a longer period of time. Acclimation can be accomplished through either behavioral or physiological responses to changes in salinity. The period of time required for acclimation is species dependent (Parry, 1966). Larval fish from 1,000 and 6,000 µS/cm exposed to 29,000 µS/cm had 100 percent mortality at 72 h. Eighty-eight percent of the 10,000 µS/cm source larvae exposed to 29,000 µS/cm also died within 72 h. Delayed mortality was observed once again with larvae from the 20,000 µS/cm source. At 72 h, only 1.6 percent mortality had occurred. Mortality of greater than 50 percent required a total of 216 h. Again, this suggests some degree of acclimation occurred and that incremental exposure to higher salinities may improve survival.

Mortality rates for control and long-term holding tanks were examined to determine the difference in long-term survival between salinity levels. Control and long-term holding tanks for 1,000, 3,000, 6,000, and 10,000 µS/cm were monitored over a 60-day period. Control tanks for 15,000 and 20,000 µS/cm were obtained by retaining 1,000 µS/cm source fish that had been exposed to these salinities during the first trial. These tanks were monitored for a total of 50 days, which includes their participation in the first trial.

Within the first 5 days, mortality for 1,000, 3,000, 6,000, 10,000, 15,000, and 20,000 µS/cm control tanks totaled 9.3, 8.6, 37, 23, 26, and 11 percent, respectively. Most of the mortality for the control group occurred within these first few days, after which rates slowed to approximately 0.34 mortalities per day. Over the 50–60 day period, control tanks for 1,000, 3,000, and 20,000 µS/cm showed minimal mortality, totaling 11.3, 8.6, and 12 percent, respectively. Control tanks for 10,000 and 15,000 µS/cm had slightly higher percentages of 24 and 26.7 percent, and the 6,000 µS/cm control had the highest mortality rate at 42.7 percent.

Mortality rates for long-term holding tanks ranged from 6.5 to 100 percent. Again, most mortality occurred in the first few days. The exception in this case was the 1,000 µS/cm tank, which experienced considerable mortality over the first 2 weeks. Dead larvae were comparatively smaller and showed high incidence of crooked backs. Crooked backed larvae were also observed swimming and often had small amounts of fungus growing on them. Larval densities and fungus or infection resulting from handling are likely factors contributing to this mortality. Densities present in 3,000 µS/cm holding tanks were similar, but mortality rates were lowest overall. This indicates a possible therapeutic effect at this salinity that may have prevented mortality because of fungus or infection. Piper and others (1982) suggest a similar salt concentration for extended treatments of bacterial disease and external parasites on hatchery raised fish species. Congruent with our control group findings, 6,000 µS/cm larvae had the highest mortality during long-term observation. One hundred percent mortality occurred for this treatment; however, this is partly because of the low starting numbers.

2008 Hatch

As was the case during the 2007 study year, hatch rates for successful salinities varied (table 4). Eggs fertilized in 10,000, 12,000, and 14,000 µS/cm developed successfully and hatched in 5 to 9 days. Resultant larvae began swimming within 30 h of hatching. Similar to our findings from comparative salinities during the first study year, eggs fertilized in 16,000 and 18,000 µS/cm salinities were of reduced size and unsuccessful. These eggs were examined after 5 days and discarded when no further development was observed.

Table 4. 2008-Mean (±SD) percent hatch of razorback sucker eggs subjected to experimental salinities. Number of eggs and larvae from eggs are combined totals for the three replicate treatments.

[µS/cm, microsiemens per centimeter; mL, milliliter]

Spawning salinity (µS/cm)	Number of eggs *	Larvae from eggs	% Hatch
10,000	7,350	3,089	42.0 ± 5.2
12,000	7,350	2,533	34.5 ± 6.9
14,000	7,350	366	5.0 ± 0.01
16,000	2,300	0	0
18,000	2,300	0	0

* Number of eggs for 10,000, 12,000, and 14,000 µS/cm estimated on the basis of 49 eggs/mL measurement; 16,000 and 18,000 µS/cm on the basis of 92 eggs/mL measurement.

2008 Larvae Trial

Because of an insufficient number of larvae available from the 14,000 µS/cm hatch, larvae for the 2008 trial came only from the 10,000 and 12,000 µS/cm spawning salinities. Larvae from the 14,000 µS/cm hatch, however, were kept in their respective tanks and grouped with the control tanks for long-term observation. Our results indicate that tempering these fish did have a positive effect on survival when compared to our findings from the previous year. Significant mortality did not occur until salinities approached 27,500 µS/cm. Four of the six trial tanks had greater than 50 percent mortality at 72 h with salinities ranging from 27,300 to 27,500 µS/cm. The remaining two tanks took 96 h to achieve greater than 50 percent mortality and had salinities of 27,500 and 27,750 µS/cm (table 5).

Long-term survival was monitored in control tanks over 45 days. Mortality for 10,000 and 12,000 µS/cm control tanks remained low overall and ranged from 1.6 to 7 percent. Mortality for 14,000 µS/cm was considerably higher, ranging from 50 to 58 percent between the four tanks. The high mortality rate for these tanks is likely an effect of being spawned in this salinity. We observed larvae enduring much higher salinities with less mortality both in the first study year and in the tempering trial. The key factor separating these groups is that larvae able to survive at higher salinities were all spawned at salinities below 14,000 µS/cm.

Table 5. 2008-Salinity tempering trial. Concentration and time to ≥ 50% mortality.

[µS/cm, microsiemens per centimeter; h, hour]

Tank #	Source (µS/cm)	Current (µS/cm)	Time (h)	% Mortality
11	10,000	27,500	96	55.3
12	10,000	27,450	72	52.6
13	10,000	27,750	96	52.4
14	12,000	27,500	72	68.9
15	12,000	27,300	72	78.1
16	12,000	27,450	72	64.2

Discussion

Razorback sucker eggs fertilized in experimental salinities between 1,000 and 12,000 µS/cm developed normally and produced larvae within 10 days. For these salinities, 22 to 56 percent of the fertilized eggs were successful. We were most successful with our 1,000 and 3,000 µS/cm groups, which had hatch rates of 55 and 56 percent, respectively. These findings suggest that even though eggs are able to develop and hatch at higher salinity levels, moderate to low salinities promote greater egg success. Our 14,000 µS/cm group also produced larvae with a 5 percent hatch overall. This low rate of success indicates this value is very near the maximum that these eggs can tolerate.

Razorback larvae were exposed to a wide range of salinities (1,000–29,000 µS/cm) over the course of this study. Results from the first year showed that of the chosen experimental salinities, 26,000 µS/cm was the minimum value lethal to 50 percent of larvae at 72 h. Observations also showed that survival at 23,000 µS/cm was possible, as relatively low mortality rates of 0, 3, 7, and 19 percent were observed in these four trial tanks over 312 h. Further observations made during the second trial led to the hypothesis that acclimating larvae to increasing levels of salinity would improve survival at higher salinities. This hypothesis was tested during the second study year, and the minimum lethal salinity increased to 27,300 µS/cm. Parity was observed in all tanks used in this trial with lethal salinity ranging from 27,300 to 27,750 µS/cm. Depending on the method by which larvae are exposed to extreme salinities, the maximum salinity tolerance will range from >23,000 to 27,750 µS/cm.

Long-term survival of larval razorbacks can be expected in salinities up to and including 20,000 µS/cm. Results from the 2007 study showed only 12 percent mortality for 20,000 µS/cm larvae after 50 days of exposure. Although it appears larvae may also be able to survive at salinities as high as 23,000 µS/cm, our experiment did not allow for a long enough period of observation to make this determination. In general, larvae handled our low and mid-range experimental salinities well. This is of significant importance as the majority of managed habitats for these fish fall within this range. The large reservoirs of the lower Colorado River, including Lakes Mead, Mohave, and Havasu, all have salinities that range between 800 and 1,100 µS/cm, depending on flow. Backwater habitats on Lake Mohave, currently used as grow-out ponds for razorback suckers, tend to have slightly higher salinities ranging from 1,000 to 3,500 µS/cm. The Davis Cove native fish sanctuary pond, also found on Lake Mohave, has had salinities recorded in excess of 5,000 µS/cm, while supporting small populations of razorbacks (Mueller, 2007). It should be noted that although this species shows the ability to tolerate relatively high salinities, preferred salinities may be found in the low- to mid-range values. Meador and Kelso (1989) investigated behavioral responses of largemouth bass (*Micropterus salmoides*) to various salinities (0–17,000 µS/cm) and found that young largemouth preferred the lowest available salinity, and adult largemouth preferred the 4,000 µS/cm salinity. In managing razorback sucker habitat, identifying tolerances as well as preferences is important for creating a successful environment for these fish.

Implications for Management

The goal of this study, as well as our future research into salinity tolerances for juvenile razorbacks; dissolved oxygen tolerances for egg, larvae, and juvenile razorbacks; and the repetition of these studies using bonytail, is to provide managers with an effective set of guidelines to aid in management, operation, and development of natural and manmade backwater habitats. Understanding the tolerances of these species with respect to various water-quality parameters will allow managers to assess habitat conditions through low-cost water-quality monitoring. Implementation of appropriate strategies to maintain optimal water quality will depend on individual site conditions and available resources.

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Effectiveness of the Barrier-and-Renovate Approach to Recovery of Warmwater Native Fishes in the Gila River Basin

By Robert W. Clarkson[1] and Paul C. Marsh[2]

Abstract

Segregating native from nonnative species is the primary tactic in recent efforts to conserve and recover imperiled warmwater native fishes in the Gila River Basin of Arizona, New Mexico, and Sonora. Isolation of the two types of species has been achieved primarily through barrier construction followed by chemical eradication of the nonnative fauna and repatriation of native fishes. A similar approach has assisted with conservation of federally listed trouts across the West, but application to lower elevation, arid-land streams can be more difficult because of the larger watersheds involved and related hydrological differences. These latter distinctions often include: (1) a need for more massive (and hence expensive) fish barriers, in part as protection against flood damage; (2) greater geomorphological impacts to the streambed from barrier emplacement; (3) consideration of upland stock tanks that may harbor nonnative fishes; and (4) diverse land ownership patterns that complicate right-of-entry and environmental compliance. Here we assess examples of barrier-and-renovate projects that have been applied to warmwater streams in the Gila River Basin. We conclude such projects represent the only viable solution currently available to conserve and recover native fishes, but these projects must be carefully selected and comprehensively implemented to achieve maximum conservation benefit with limited funding.

Introduction

Segregating native from nonnative species, or isolation management (Novinger and Rahel, 2003), is the primary tactic in recent efforts to conserve and recover imperiled warmwater native fishes in the Gila River Basin of Arizona, New Mexico, and Sonora. Establishment of nonnative fishes in waters of the American Southwest is now considered the primary cause of the deteriorating status of native fishes in the region and prevents their recovery (Minckley, 1991; Clarkson and others, 2005; Light and Marchetti, 2007; Minckley and Marsh, 2009; Stefferud and others, 2009). Segregation of the native and nonnative faunas has been achieved primarily through chemical eradication of nonnative fishes and repatriation of native fishes following barrier construction to preclude re-contamination of the upstream, treated reach. A similar approach has been incorporated into recovery planning for many federally listed trouts across the West, which has improved or minimally halted further deterioration of their conservation status (Young, 1995; Thompson and Rahel, 1998; Avenetti and others, 2006; Pritchard and Cowley, 2006; but see Hilderbrand and Kershner, 2000; Novinger and Rahel, 2003). However, application of the barrier-and-renovate approach to lower elevation, arid-land streams can be more difficult because of the larger watersheds involved and more complex hydrological and land-use differences.

The purpose of our paper is to describe the characteristics of warmwater streams in the Gila River Basin as they relate to fish barrier construction and chemical renovations, and to assess the successes and failures of barrier-and-renovate projects that have been applied toward native fish recovery in these stream types. We conclude with a discussion of potential future directions of recovery efforts for the warmwater native fauna in the region.

Methods

We have participated at various levels in the planning and implementation of most barrier-and-renovate projects for warmwater native fishes in the Gila River Basin during the past decade, and our experiences form the basis of this paper. Robert W. Clarkson leads a Bureau of Reclamation (Reclamation) program mandated by the U.S. Fish and Wildlife Service to construct fish barriers on a dozen streams to assist with recovery of federally listed warmwater native fishes in the

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basin. Clarkson and Paul C. Marsh developed criteria for the basic designs of the newly constructed fish barriers discussed below (with the exception of Arnett Creek), and both have reviewed and modified construction specifications. In addition, Clarkson co-authored and Marsh commented on environmental planning documents (National Environmental Policy Act and Endangered Species Act) for all aspects of the barrier-and-renovate projects (barrier construction, fish salvage, chemical renovation, native fish repatriation, post-project monitoring) and participated with much of the on-the-ground implementation of the projects.

Marsh also assisted with project planning and most aspects of project implementation, and his consulting company has been a primary contractor for post-project fish monitoring and reporting relative to the success of the barriers and species repatriations. This collective involvement provided us with documentation and first-hand experience with such projects from concept to conclusion. Locations of the various fish barrier projects discussed here are shown in figure 1.

Results and Discussion

Features of Low-Elevation Watersheds

The most obvious features of watersheds that distinguish low-elevation streams from high-elevation streams in the southwestern desert region are the larger watersheds involved and the increased frequency and magnitude of flood events. Lower parts of watersheds accumulate flood impacts from disparate subbasins upstream, resulting in highly variable and more-elevated hydrographs. Low-order, headwater reaches exhibit more stable flow regimes. Relative to streams in more mesic areas, design specifications for barriers across this continuum must accommodate variable flooding impacts that result from differences in drainage size, precipitation patterns, and other factors.

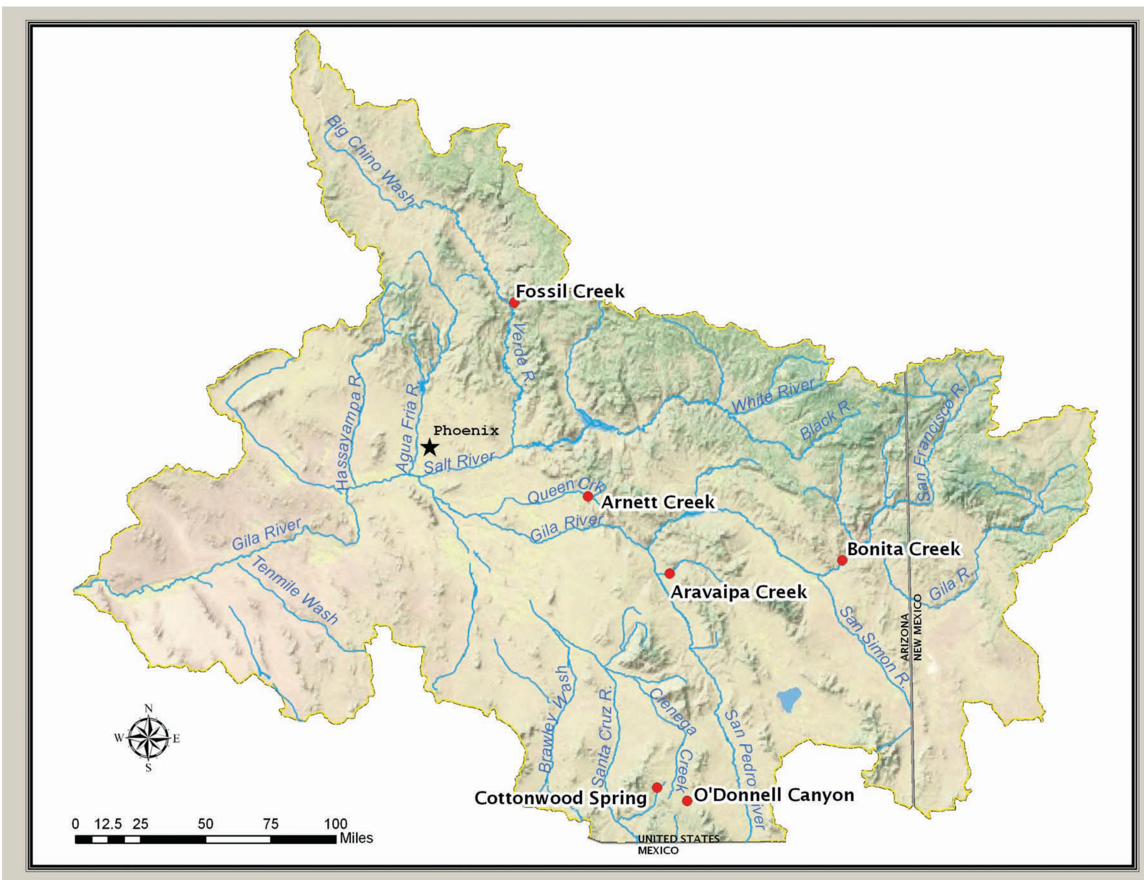

Figure 1. Boundaries of the Gila River Basin, Arizona and New Mexico (exclusive of Sonora), showing major streams and locations of fish barrier projects discussed in the text.

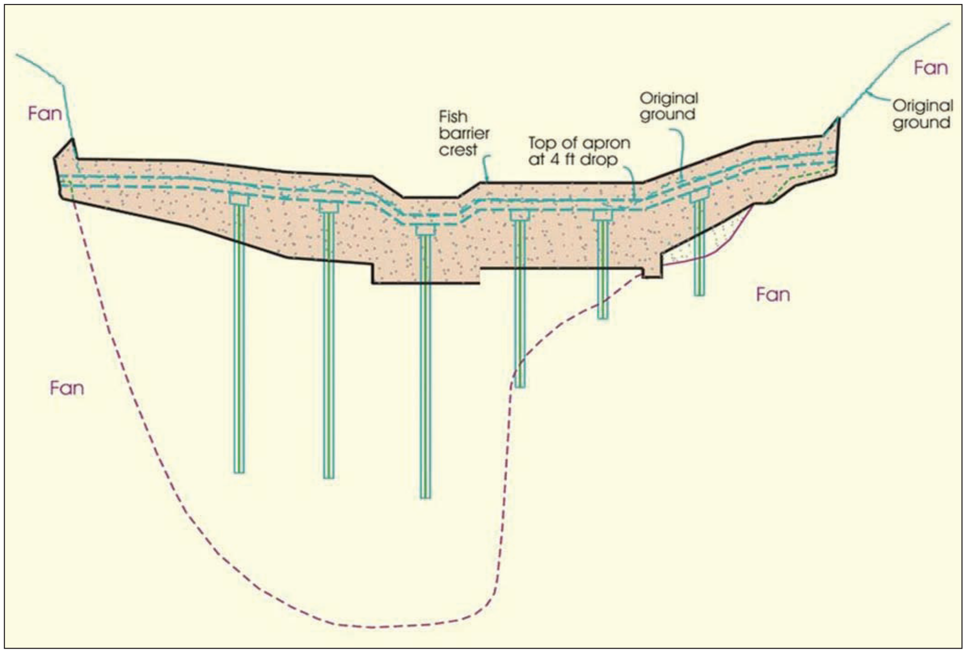

Figure 2. Plan view of the lower fish barrier on Aravaipa Creek, Pinal County, Arizona, showing the placement of concrete piles to stabilize the structure. "Fan" refers to fanglomerate, an accumulation of cemented coarse materials in an alluvial fan.

Unless tied into bedrock at all points across the stream channel, high-magnitude floods dictate that fish barrier designs accommodate increased scour effects to protect them from flood damage. Most barrier-and-renovate projects we describe below are of steel-reinforced concrete design and incorporated upstream and downstream keys (scour walls) to protect against natural bed scour, bridge pier-type scour, and scour induced by the structure. In one case, reinforced concrete piles were sunk up to 60 feet below grade to help protect the structure against sliding forces (fig. 2). In addition, riprap placement along the downstream key is a common design element in an attempt to prevent a scour hole from developing downstream from the structures (fig. 3). Prevention of scour holes also minimizes the attraction of recreationists to the site, which should reduce the potential for humans to move nonnatives above the barrier.

Although design engineers can protect barriers against most flood damage, emplacement of a hard structure within an alluvial stream channel has strong potential to alter channel geomorphology. In addition to alteration of channel slopes upstream from barriers as a result of aggradation, scour effects downstream from barriers have potential to remove sediment from stream terraces and the thalweg, despite emplacement of riprap armoring. If riprap materials are not of sufficient diameter and their placement does not extend below scour depth, scour during high-magnitude floods can erode underneath and sink or transport materials downstream. In addition, riprap must be emplaced with these considerations in mind across the width of the entire channel to prevent erosion from progressing from channel margins toward the thalweg. Prevention of these types of scour appears to be the most daunting challenge to fish barrier designs in low-elevation streams in the basin.

Figure 3. Riprap placement downstream from the apron of the Bonita Creek fish barrier, Graham County, Arizona.

Because of the aridity of lower elevations of the Gila River Basin, uncounted stock ponds have been constructed to facilitate better use of uplands by domestic livestock. Many of these artificial impoundments hold water year round, and they often harbor populations of nonnative fishes that have potential to contaminate downstream waters during spill events. Because fish barriers only prevent upstream invasion of nonnative fishes, stream restoration projects must also eliminate nonnative fishes from upstream sources to secure the drainage.

The larger drainage areas typically associated with low-elevation perennial streams in arid environments often mean that there is greater variability in land ownership. For example, application of piscicide to private property parcels that typically fall along stream corridors requires that each property owner must approve the project or the project cannot be completed successfully. Often, different Federal or State land managers have differing management priorities that can conflict with project goals. In general, as watershed size increases, so does project complexity and potential for controversy.

Successes and Failures of Barrier-and-Renovate Projects

Table 1 summarizes results of representative case histories of barrier-and-renovate projects, each treated in detail below. Data are mostly from unpublished reports that are available from the respective agencies. We include dates of repatriations and numbers of individuals stocked to provide the reader an opportunity to independently assess the actions. Additional information on fish barrier specifications can be found at *http://www.usbr.gov/lc/phoenix/biology/azfish/dropbarriers.html*.

O'Donnell Canyon

One of the first stream restoration projects attempted for warmwater native fishes in the Gila River Basin was undertaken in 2001 at O'Donnell Canyon, a tributary to Babocomari River in the San Pedro River drainage (fig. 1). Historically an important locality for endangered Gila chub (*Gila intermedia*), the population was increasingly depleted over time because of infestation by invasive green sunfish (*Lepomis cyanellus*). The stream already was protected against upstream fish invasions by two small dams constructed in the 1950s, and so the project consisted primarily of chemical renovation of the stream above the dams using the piscicide antimycin-A. The renovation was successful in removing the sunfish population, and the small number of salvaged Gila chub and Sonora sucker (*Catostomus insignis*) were repatriated (table 1).

In this case, although repatriated Gila chub reproduced and population numbers expanded, natural dispersal to previously occupied reaches of the stream has not yet occurred (repatriation was only to the upper of two reaches). Augmentation of the population appears necessary, as well as human-assisted releases of fish (including Sonora sucker) to unoccupied areas. O'Donnell Canyon is one of the few remaining ciénegas (marshes) formerly common in the southern Gila River Basin (Hendrickson and Minckley, 1985) and is characterized by mostly deep, narrow pools connected by low base-flow channels. A major drought occurred in the region around the time of the renovation, and most surface flows between pools have since been intermittent. Perhaps this flow reduction explains the lack of dispersal of Gila chub downstream.

Historical collection records indicated native longfin dace (*Agosia chrysogaster*) and Gila topminnow (*Poeciliopsis occidentalis*) were present in O'Donnell Canyon, but neither had been detected for many years before the renovation. Remarkably, both appeared post-project, and both have since been observed throughout most of the creek upstream from the barriers. In 2008, two stock tanks in the drainage above the ciénega were found to harbor nonnative fishes, and one (mosquitofish [*Gambusia affinis*]) was found in the ciénega headwaters in 2009. These sources, and possibly the entire stream system, will have to be re-renovated to re-establish an intact native fish assemblage.

Both of the 1950s-era dams that have functioned as fish barriers are currently in danger of failure, and planning is underway to stabilize one or both or to construct an additional barrier further downstream that would protect additional subdrainages. The decision to build a new barrier hinges on whether renovations of the new subdrainages are politically feasible because of the considerable number of private property owners along one of the streams.

Fossil Creek

The 2004 Fossil Creek Native Fish Restoration Project has thus far been the most complex, comprehensive, and successful attempt at securing a stream for warmwater native fish recovery purposes in the basin (fig. 1). Nearly all of the 43 cubic-feet-per-second base flow of this stream had been diverted for hydropower purposes for the past century, and nonnatives had also invaded or been stocked, drastically suppressing the remnant native fish community (headwater chub [*Gila nigra*], roundtail chub [*Gila robusta*], longfin dace, speckled dace [*Rhinichthys osculus*], Sonora sucker, and desert sucker [*Pantosteus clarki*]). In a remarkable and historic occasion, the hydropower company, Arizona Public Service, agreed to return full flows to the channel and decommission the project in concert with native fish restoration efforts. First, a fish barrier was constructed across an existing bedrock outcrop (fig. 4; cost $275 thousand), followed by native fish salvage and chemical renovation of the stream (antimycin-A) and upland stock tanks (rotenone). Full flows were returned to the stream, and salvaged fishes were repatriated.

Figure 4. The fish barrier on Fossil Creek, Gila and Yavapai Counties, Arizona.

Table 1. Species assemblages before and after native fish restorations of selected warmwater streams in the Gila River Basin, Arizona and New Mexico. Data sources are provided in the first column following stream name. Indications of reproduction are based on presence of young-of-year captured during routine post-project monitoring. Asterisks denote nonnative species. In most cases, augmentations of repatriations are ongoing.

Stream/data source	Pre-restoration			Post-restoration			
	Species assemblage[a]	Date of barrier construction	Date of renovation	Date(s) of repatriation	Numbers repatriated	Species assemblage	Reproduction
Aravaipa Creek[b]	Desert sucker	2001	Not applicable	Not applicable	Not applicable	Desert sucker	Yes
Reinthal, P., University of Arizona, unpub. data, 2009	Sonora sucker					Sonora sucker	Yes
	Roundtail chub					Roundtail chub	Yes
	Speckled dace					Speckled dace	Yes
	Longfin dace					Longfin dace	Yes
	Loach minnow					Loach minnow	Yes
	Spikedace					Spikedace	Yes
	Green sunfish*					Green sunfish*	Yes
	Yellow bullhead*					Yellow bullhead*	Yes
	Red shiner*					Red shiner*	Yes
Arnett Creek	Green sunfish*	1997	1997	1999	13	Desert sucker	No
Robinson, A.T., unpub. report, 2008	Mosquitofish*			1999	1	Sonora sucker	-
				1999	23	Longfin dace	No
				2007	100	Longfin dace	Yes
Bonita Creek	Desert sucker	2008	2008	2008	1	Desert sucker	-
Robinson and others, unpub. report, 2009	Sonora sucker			2008	201	Sonora sucker	Undetermined
	Gila chub			2008	230	Gila chub	Undetermined
	Speckled dace			2008	25	Speckled dace	Undetermined
	Longfin dace			2008	107	Longfin dace	Undetermined
	Green sunfish*			2008	678	Loach minnow	Undetermined
	Smallmouth bass*			2008	448	Spikedace	Undetermined
	Fathead minnow*			2008	147	Desert pupfish	Undetermined
	Common carp*			2008	975	Gila topminnow	Undetermined
	Yellow bullhead*			-	0	Mosquitofish*	Undetermined
	Black bullhead*						
	Channel catfish*						
	Flathead catfish*						
	Mosquitofish*						
Cottonwood Spring[b]	Gila topminnow	2003	Not applicable	Not applicable	Not applicable	Gila topminnow	Yes
Stefferud, S. (retired), U.S. Fish and Wildlife Service, oral comm., 2009	Desert sucker					Desert sucker	Yes
	Longfin dace					Longfin dace	Yes
Fossil Creek	Desert sucker	2004	2004	2004	354	Desert sucker	Yes
Weedman and others, unpub. report, 2005	Sonora sucker			2004	204	Sonora sucker	Yes
	Roundtail chub			2004	250	Chub spp.	Yes
Marsh and others, unpub. report, 2009	Headwater chub			2004	906	Speckled dace	Yes
	Speckled dace			2004, 2007	318	Longfin dace	Undetermined
Robinson, A.T., unpub. report, 2009	Longfin dace			2007–2008	2128	Loach minnow	Undetermined
	Green sunfish*			2007–2008	725	Spikedace	Undetermined
	Smallmouth bass*			2007–2008	5000	Gila topminnow	Yes
	Yellow bullhead*			2007–2008	579	Razorback sucker	No
O'Donnell Canyon[c]	Sonora sucker	1950s	2001	2001	~30	Sonora sucker	Yes
Blasius, H., Bureau of Land Management, oral comm., 2009	Gila chub			2001	~20	Gila chub	Yes
	Gila topminnow			-	0	Gila topminnow	Yes
	Longfin dace			-	0	Longfin dace	Yes
	Green sunfish*						

[a] Does not necessarily reflect the historical (pre-settlement) assemblage of native species.

[b] Barrier construction only; project intended to prevent invasions of new nonnatives.

[c] Renovation and repatriations upstream of pre-existing fish barriers.

Native fishes reproduced (table 1) and have recolonized most of the stream. Five additional federally listed species (Gila topminnow, desert pupfish [*Cyprinodon macularius*], loach minnow [*Tiaroga cobitis*], spikedace [*Meda fulgida*], and razorback sucker [*Xyrauchen texanus*]) were also released to the stream in 2008, but it is too early to determine if they have persisted and established self-reproducing populations. Monitoring has found that two stock tanks were stocked with nonnatives and both were successfully re-renovated. No non-natives have reinvaded the stream to date (table 1), and only relatively minor barrier maintenance has yet been required.

Figure 5. The fish barrier on Bonita Creek, Graham County, Arizona.

Bonita Creek

In 2008, a 160-foot wide reinforced-concrete fish barrier was constructed on Bonita Creek (cost $2.01 million), a tributary to Gila River in eastern Arizona (figs. 1 and 5), and a contaminated portion of the stream was chemically renovated with rotenone (CFT Legumide). Salvaged native fishes (Gila chub, speckled dace, longfin dace, Sonora sucker, desert sucker) plus four federally listed taxa native to the area but not known to be from the stream (spikedace, loach minnow, Gila topminnow, desert pupfish) were repatriated in furtherance of the species' recovery goals. Newly added species will be augmented in spring 2009 to increase founding population size and enhance genetic variability. Future monitoring will determine success of this restoration attempt. This project required a compromise that provided water rights to a municipal water user that allowed the project to proceed.

Figure 6. The lower fish barrier on Aravaipa Creek, Pinal County, Arizona.

Other Streams

Other situations exist in the basin where fish barriers have been constructed to protect either intact native assemblages from potential nonnative fish contamination in the future or where contaminated streams yet hold valuable native fish communities worth protecting against contamination by additional invasive species. In the case of Aravaipa Creek, tributary to San Pedro River (figs. 1 and 6) and one of Arizona's most valued native fish communities (seven extant species), two barriers were built in 2001 (total cost $3.1 million) to protect against invasion by species such as red shiner (*Cyprinella lutrensis*) and in the hope that extant nonnative species (primarily green sunfish and yellow bullhead [*Ameiurus natalis*]) could be washed from the system during flood events and prevented from reinvading. Unfortunately red shiner accessed the stream before the barriers could be completed, and a 50-year flood event that occurred after construction failed to remove any unwanted species.

Aravaipa Creek was the first barrier project completed under Reclamation's barrier construction program in the basin, and this project consisted of paired barriers. The rationale was that if nonnative fishes passed the lower barrier, they could be removed from between the barriers before they could invade further upstream. Private landowners opposed the project. As a result, the barriers were constructed downstream on a parcel that was too small, and design miscalculations resulted in the upper barrier being buried by aggradation behind the lower barrier. The paired barrier concept has since been abandoned primarily because of cost:benefit concerns; construction of single barriers on twice as many streams in theory could achieve greater conservation benefit for native fishes.

Channel degradation downstream from the lower Aravaipa Creek barrier also has been significant (fig. 7), but the lower barrier has been successful in preventing invasion by Northern crayfish (*Orconectes virilis*) and possibly other species, and natives continue to vastly outnumber nonnatives upstream. Should natives begin to decline significantly relative to nonnatives, the stream could become a candidate for chemical renovation, but the size of the watershed and the significant number of private properties along the stream would make such a renovation a challenge.

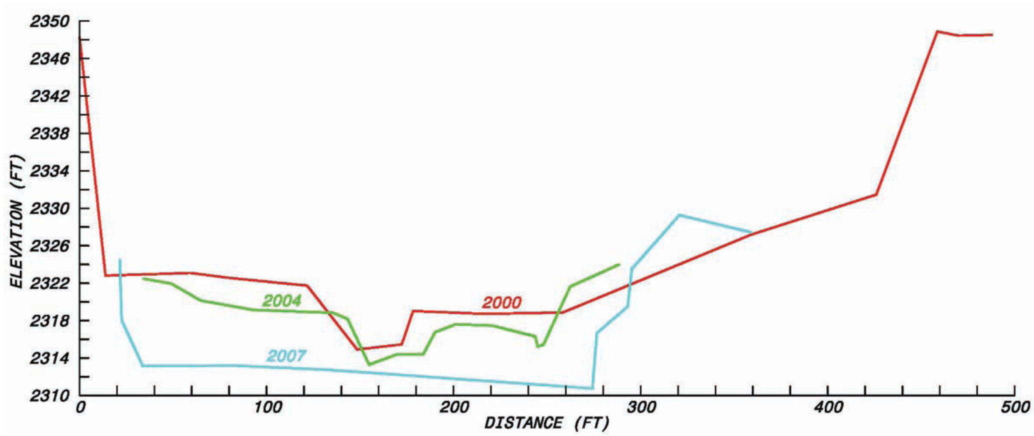

Figure 7. Surveyed cross sections of the stream channel on Aravaipa Creek immediately downstream from the lower fish barrier, showing channel configurations immediately pre-construction (2000), 3 years post-construction (2004), and 1 year following passage of an estimated 50-year flood event (2007).

A fish barrier constructed downstream from Cottonwood Spring on Sonoita Creek in the Santa Cruz River subbasin (figs. 1 and 8; cost $115 thousand) protects a population of endangered Gila topminnow and a couple of other unlisted native species (table 1). Only the reach downstream from the barrier is contaminated by nonnatives. The purpose of this barrier, to prevent an important native fish population from nonnative invasion, has thus far been successful. A willing private landowner in this instance greatly facilitated the implementation of the project.

The Native Fish Restoration Project on Arnett Creek, tributary to Queen Creek in the lower Salt River drainage (figs. 1 and 9), is an example of what can go wrong with a barrier-and-renovate project. The stream historically harbored at least three native fishes (Gila chub, longfin dace, desert sucker), but green sunfish and mosquitofish had invaded and decimated the native fishes. A poorly designed rock gabion fish barrier was constructed and later nearly destroyed by flood. The barrier was rebuilt and reinforced with concrete. The stream was successfully chemically renovated to remove all fishes. Following stocking of very small numbers of two unlisted fishes (table 1), the stream desiccated in 2002 and remained fishless until longfin dace was repatriated in 2007. Plans to stock the stream with additional listed species have not yet been implemented because of various concerns, including drought and habitat changes, that have occurred since the barrier construction. We remain optimistic that Arnett Creek can eventually contribute to recovery of native fishes.

Figure 8. The fish barrier on Sonoita Creek, Santa Cruz County, Arizona, downstream from Cottonwood Spring.

Figure 9. The reconstructed fish barrier on Arnett Creek, Pinal County, Arizona.

Implications for Management

We acknowledge that artificial barriers fragment populations already partitioned by groundwater pumping, diversions, dams, and occupation of mainstem habitats by nonnative fishes. Such fragmentation depletes population genetic variability, and populations may become less adaptable to change and more susceptible to extirpation. However, the continued declining trend of native fishes in the region, including losses of populations in mainstem and tributary habitats, dictates that protection of remaining populations is a higher priority than meeting longer term evolutionary needs. Human intervention will be necessary to ensure that genetic variability of populations above barriers is maintained until adjacent stream reaches can be cleansed of nonnatives. Once a drainage network is protected, upstream barriers could then be breached to once again restore natural connectivity among populations. At present, sociopolitical circumstances prevent decontamination of nonnative fishes from larger drainage networks that also support nonnative sport fisheries (Clarkson and others, 2005).

Recently implemented barrier-and-renovate projects have demonstrated the viability of the approach in conserving native fishes in arid region streams typified by the Gila River Basin. However, success is contingent upon several important factors. First, streams must be carefully selected to ensure they meet physical, biological, and sociopolitical criteria necessary for successful barrier installation, renovation, and restoration of native communities. All aspects must be comprehensively performed, as failure of any ensures failure of the whole project. Comprehensive performance of a restoration means that all potential sources of contamination—from upstream, downstream, or by human transport—are identified and eliminated or minimized. Barrier construction and stream renovations are costly endeavors, and each demands detailed planning, substantial time, and a large workforce to successfully complete. The politics of federally listed species repatriations also must be carefully worked through to complete a project. The end result can be a substantial enhancement of the conservation status of native fish communities.

We stress that the only viable direction for recovery of native fishes in the region is segregation of native from nonnative fishes, and that in the Gila River Basin the barrier-and-renovate strategy appears to be the only currently available option that can effectively achieve such segregation. Potential alternatives, such as application of taxon-specific piscicides to remove target species without the need for complete assemblage renovation or genetic bioengineering that has similar potential to remove targeted populations while leaving others intact, have been identified but their implementation is far in the future. Both of these options likely would yet require use of fish barriers to maintain segregation. Although the handful of barrier-and-renovate projects described here appears to be mostly successful in establishing and preserving viable native fish communities in lower elevations of the Gila River Basin, dozens of additional streams must be dedicated toward these efforts, and tens of millions of dollars for barrier constructions and renovations will be required if biologically significant conservation of native species is to occur. Virtually all viable streams that could be devoted for native fish use without significant impact to existing sport fisheries already have been identified, and restoration projects are complete or in planning stages. Any further commitment to native fish conservation thus will require compromise on the behalf of sport fish and other interests. Without such compromise, we cannot envision a future where Gila River Basin native fishes are recovered and Endangered Species Act (Public Law 93–205) protections are eliminated.

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Evaluating Effects of a High-Flow Event on Rainbow Trout Movement in Glen and Marble Canyons, Arizona, by Using Acoustic Telemetry and Relative Abundance Measures

By Kara D. Hilwig[1] and Andy S. Makinster[2]

Abstract

In March 2008, the Department of the Interior conducted a high-flow event (HFE; 1,175 cubic meters per second for 60 hours) through Glen Canyon Dam and Grand Canyon. This study evaluated the impact of the HFE on movement of adult and juvenile rainbow trout (*Oncorhynchus mykiss*) in Lees Ferry. Downstream displacement of rainbow trout could impact the endangered humpback chub (*Gila cypha*) in downstream areas and recreational angling in Lees Ferry. We evaluated rainbow trout movement by comparing relative abundance indices from electrofishing surveys and acoustic telemetry techniques before and after the HFE. We determined that rainbow trout relative abundance indices were similar before and after the HFE. Acoustic tagged rainbow trout did not appear to displace downstream, and relative movement was similar before and after the HFE. Movement of tagged rainbow trout also did not correlate with length class or sex. Abundance indices in combination with acoustic telemetry results indicate that the March 2008 HFE did not appear to cause significant downstream displacement of adult and juvenile rainbow trout in Lees Ferry. Other evidence suggests that populations of young rainbow trout (age-0 and age-1 less than 100 millimeters) were not impacted by the March 2008 HFE. However, a threefold decrease in population size of young rainbow trout was observed during the November 2004 HFE. These data suggest the need for further studies to track the fate of young rainbow trout and other environmental and temporal factors that may cause movement during future HFEs.

Introduction

High-flow events (HFE) were conducted in 1996, 2004, and 2008 by the Department of the Interior to investigate their utility in restoring natural, cultural, and recreational resources within Grand Canyon National Park. A high-flow experiment was conducted March 4–6, 2008, with flows reaching a maximum of 1,175 cubic meters per second (m^3/s) for about 60 hours. These flows were approximately three times greater than the peak flows released by Glen Canyon Dam immediately preceding the HFE.

The HFE was conducted in an attempt to move sand in the Colorado River system and conserve beach habitats. Other important resources for conservation include the Lees Ferry recreational rainbow trout (*Oncorhynchus mykiss*) fishery in the tailwaters of Glen Canyon Dam and the federally endangered humpback chub (*Gila cypha*), which is found further downstream in Grand Canyon. Lees Ferry is located approximately 15 river miles[3] downstream from Glen Canyon Dam near Page, AZ (fig. 1). Two concerns were raised regarding potential rainbow trout movement as a result of the HFE. Recreational anglers were concerned that adult rainbow trout may be displaced downstream from Lees Ferry into areas inaccessible to the majority of the angling community. Conservationists were concerned that the HFE could cause downstream displacement of adult rainbow trout into the Little Colorado River inflow reach of the Colorado River where they could prey on humpback chub. To address these concerns, we developed this investigation to evaluate the impact of the HFE on rainbow trout movement in the Lees Ferry area.

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[3] By convention, river mile is used to describe distance along the Colorado River in Grand Canyon.

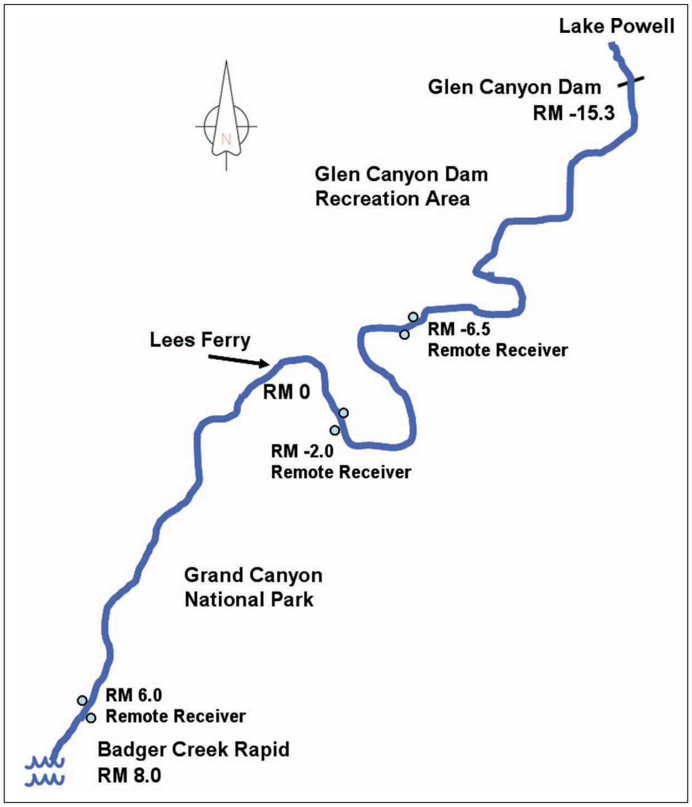

Figure 1. The study area in the Lees Ferry area from Glen Canyon Dam to Badger Creek Rapid in Glen Canyon Dam National Recreation Area and Grand Canyon National Park near Page, AZ. Dots indicate the placement of remote receivers to detect passing acoustic tagged rainbow trout. River mile (RM) is used to describe distance along the Colorado River in Grand Canyon. Lees Ferry is the starting point, RM 0, with mileage measured for both upstream (–) and downstream directions.

Inferences on fish movement can be made by comparing relative abundance indices before and after a flood disturbance (Meffe, 1984; Matthews, 1986; Meffe and Minckley, 1987), but they are limited without considering ancillary information. During a previous HFE in Grand Canyon in March 1996, an increase was observed in relative abundance of rainbow trout (<152 millimeter (mm) total length) in the Little Colorado River inflow reach of the Colorado River (Valdez and Cowdell, unpub. report, 1996). The authors hypothesized that downstream displacement of fish from Lees Ferry and Glen Canyon by the HFE was likely responsible for increased relative abundance; however, no direct linkage to the source of the displaced fish could be made. Korman (2009) observed a threefold decrease in the population size of young rainbow trout (age-0 and age-1; <100 mm) in Lees Ferry after the November 2004 HFE and hypothesized downstream displacement or mortality of these fish. In both cases, however, direct observation of displacement or the fate of displaced fish could not be made using relative abundance indices.

Determining the fate of fish displaced by flood disturbance can be difficult (Chapman and Kramer, 1991). Often researchers individually mark fish to track movement, however, marked fish must be recaptured. Few recaptures of these marked fish often limit the utility of the information in evaluating population level movement (Halls and others, 1998). Use of radio or acoustic telemetry has been useful in evaluating environmental effects, including disturbance, on fish movement in other systems (Harvey and others, 1999; Valdez and others, 2001). Given the concern for displacement of adult rainbow trout and suggested displacement of juvenile rainbow trout associated with the HFE, we developed this study to compare relative abundance indices with acoustic telemetry to evaluate movement of adult and juvenile rainbow trout before and after the HFE. The goals of this experimental study were to (1) determine if the HFE causes displacement of acoustic tagged rainbow trout downstream from Lees Ferry, (2) determine if such displacement occurs differentially among different size classes of acoustic tagged rainbow trout, and (3) compare rainbow trout relative abundance estimates in Lees Ferry before and after the HFE with acoustic tagged rainbow trout movement.

Methods

Study Area

This study was conducted in the Lees Ferry area of Glen Canyon Dam Recreation Area downstream from Glen Canyon Dam near Page, AZ (fig. 1). The study area encompassed the 15-mile reach from Lees Ferry upstream to Glen Canyon Dam and also included an 8-mile reach downstream from Lees Ferry to Badger Rapid. Discharge from Glen Canyon Dam in the year preceding the HFE typically ranged from approximately 227 to 481 m^3/s, and water temperature ranged from approximately 12.5 to 8 degrees Celsius (°C). In the month preceding the HFE, discharge fluctuated daily from approximately 227 to 396 m^3/s, and water temperature was 8 °C.

Electrofishing Surveys

We sampled the tailwater upstream from Lees Ferry on February 28–March 1, 2008 (pre-HFE), and March 18–20, 2008 (post-HFE). As part of standardized monitoring, we sampled the same 34 sites during both sampling events once per sampling event using a raft mounted electrofishing rig. Sampling was conducted with an Achilles inflatable raft equipped with Coffelt CPS output regulators. We applied approximately 350–400 volts and 12–15 amps to a 35-centimeter (cm) stainless steel anode while two crew-members netted stunned fish from the bow of the boat. These surveys were conducted to determine relative abundance (catch-per-unit-effort, CPUE) of adult and juvenile rainbow trout before and after the HFE. Electrofishing was also used to capture rainbow trout for surgical implantation of acoustic tags.

Analysis of Electrofishing Captures

Size stratified rainbow trout relative abundances (number captured per minute of electrofishing effort) were compared before and after the HFE by using a one-way analysis of variance. All statistical tests were considered significant at the $\alpha = 0.05$ level. Size classes analyzed were fish <152 mm, 152–304 mm, 304–405 mm, and >405 mm total length (TL). These length categories approximate age-1, age-2, age-3, and age-4+ rainbow trout, respectively.

Surgical Implantation and Tagged Fish Locations

The surgery protocol used to implant acoustic tags was developed by the U.S. Geological Survey (USGS) Columbia River Research Laboratory in Cook, WA. Carbon dioxide was used to anesthetize fish. Following surgical and anesthetic protocols, 19 rainbow trout were implanted with dummy tags and held for 60 days in a hatchery to evaluate long-term post-surgery survivorship. Following this same protocol for the field experiment, Sonotronics acoustic tags (thirty-two IBT-96-1 and sixty-two IBT-96-2; configured for minimum 60-day ping duration) and passive integrated transponders (PIT) tags were surgically implanted in 94 rainbow trout. Implanted rainbow trout ranged in size from 157 mm to 409 mm TL and were released at six locations above Lees Ferry ramp (February 14–23, 2008). Implanted fish were held in a perforated plastic can for a minimum of 24 hours post-surgery. Additionally, six test fish were implanted with dummy tags following the same procedures and held in the pens for 72 hours post-surgery. Remote receivers were placed at three locations to detect acoustic tagged rainbow trout between manual tracking events (fig. 1). We selected remote receiver locations that encompassed the Lees Ferry boat ramp where anchoring options were adequate and river channel was deep and flat. Four manual tracking events were conducted from Glen Canyon Dam to Badger Rapid to locate tagged fish and monitor movement; two events each were conducted pre- (pre-HFE1 February 23–24, pre-HFE2 March 2–4) and post- (post-HFE1 March 10–11, post-HFE2 March 27–28) HFE.

Acoustic tagged rainbow trout positions were recorded on a touch screen computer with ArcGIS ArcMap Version 9.2. Point locations of each fish were located on orthorectified digital images of the river corridor. Each tagged rainbow trout position was then assigned to the nearest tenth of a river mile.

Analysis of Tagged Fish Movement

Individual fish movement was calculated as change in river miles for four periods: (1) from the point of release to pre-HFE1, (2) from pre-HFE1 to pre-HFE2, (3) pre-HFE2 to post-HFE1, and (4) post-HFE1 to post-HFE2. Relative upstream and downstream movement is represented by positive and negative values, respectively. Relative average movement was calculated by averaging change in individual fish positions before the HFE (point of release to pre-HFE2) and after the HFE (pre-HFE2 to post-HFE2). The analysis period after the HFE encompassed movement that occurred during the HFE. Average fish movement of tagged trout before and after the HFE was compared using one-way analysis of variance. Analysis was also stratified by size class and sex of tagged rainbow trout. All statistical tests were considered significant at the $\alpha = 0.05$ level. Size classes analyzed were consistent with length categories used for electrofishing surveys (see above).

Results

Electrofishing

During the pre-HFE sampling event, we captured a total of 412 rainbow trout ranging in size from 48 mm to 439 mm TL. During the post-HFE sampling event, we captured a total of 352 rainbow trout ranging in size between 62 and 435 mm TL. The length frequency distribution of all rainbow trout captured during the pre- and post-HFE sampling events showed a bimodal distribution dominated by fish <200 mm TL (fig. 2).

Preliminary data indicate mean CPUE (fish caught per minute of electrofishing) of all rainbow trout did not differ significantly between pre- and post-HFE sampling events (1.40 ± 0.44 and 1.34 ± 0.51, respectively; mean ± 2 standard errors; fig. 3). Analysis showed that mean size-specific rainbow trout CPUE also did not differ between pre- and post-HFE sampling events including the youngest rainbow trout size class (<152 mm; fig. 4).

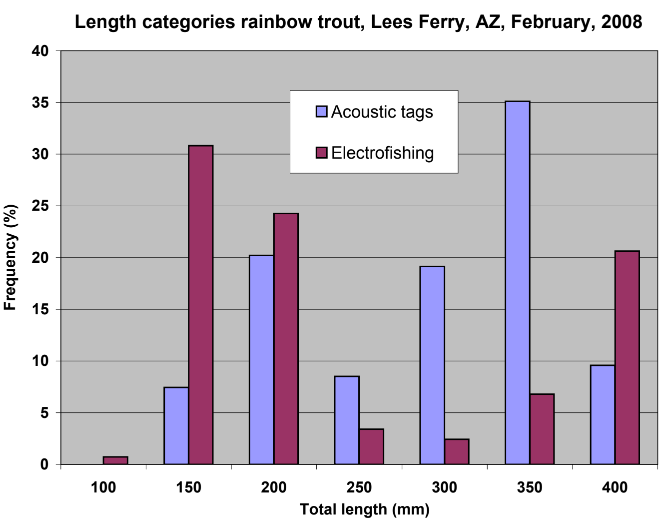

Figure 2. Length frequency of rainbow trout sampled with electrofishing and those that were implanted with acoustic tags in the Lees Ferry area during the March 2008 high-flow experiment. Fish less than 157 mm were too small to carry the acoustic tag, and fish larger than 400 mm were not susceptible to deep anesthesia required for surgery using carbon dioxide.

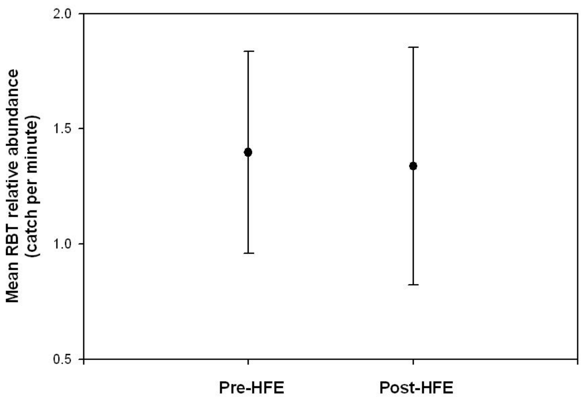

Figure 3. Mean relative abundance (catch per minute of electrofishing) of all size classes of rainbow trout (RBT) captured with electrofishing during pre- (February 28–March 1, 2008) and post-high flow experiment (HFE; March 18–20, 2008) sampling in the Lees Ferry area of the Colorado River, AZ. Bars represent ±2 standard errors of the mean.

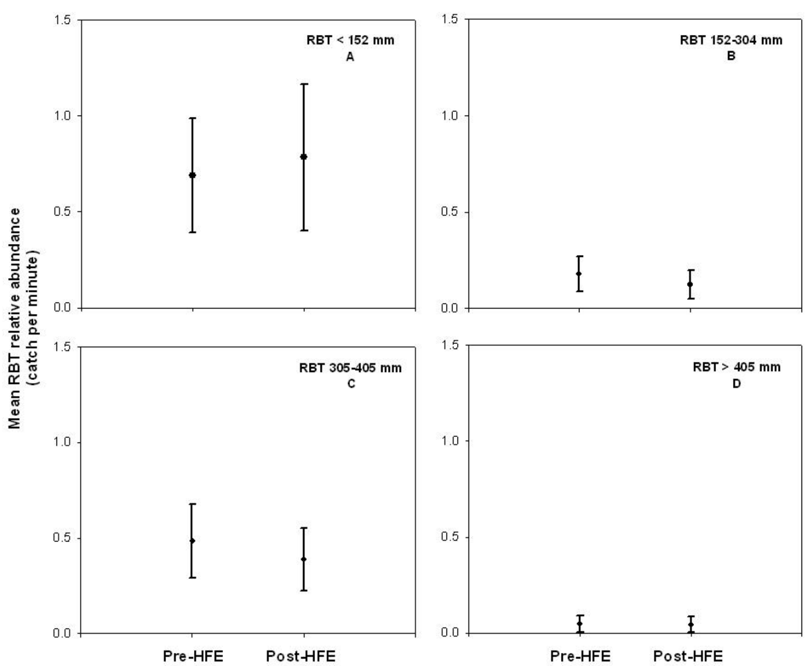

Figure 4. Size-stratified mean relative abundance (catch per minute of electrofishing) of rainbow trout (A) <152 mm total length (TL), (B) 152–304 mm TL, (C) 305–405 mm TL, and (D) >405 mm TL captured with electrofishing during pre- (February 28–March 1, 2008) and post-high flow experiment (HFE; March 18–20, 2008) sampling in the Lees Ferry area of the Colorado River, AZ. Bars represent ±2 standard errors of the mean.

Surgical Implantation

No mortality was observed in rainbow trout held for 60 days post-surgery or in dummy tagged rainbow trout held in Lees Ferry 72 hours post-surgery. Two study fish with active tags exhibited abnormal behavior 24 hours post-surgery and were replaced with two healthy fish. One acoustic tagged fish was captured by electrofishing crews 7 days post-surgery. The crew commented that the sutures had dissolved and the incision was healing well.

The length frequency of acoustic tagged fish did not exactly overlap that of fish captured during electrofishing surveys (fig. 2). Fish less than 157 mm were too small to carry the acoustic tag, and fish larger than 409 mm were not susceptible to deep anesthesia required for surgery using carbon dioxide. Therefore, movement analysis for acoustic tagged rainbow trout was limited to adult fish 152–304 mm and 305–405 mm. Thus, the population of rainbow trout that we were able to implant with tags did not proportionally represent the size classes of rainbow trout present in Lees Ferry.

Acoustic Tag Detection and Movement

Fifty-seven of 94 tagged fish were detected during pre-HFE manual tracking events. Of these 57 fish located before the HFE, 50 were also located after the HFE positioned in the Lees Ferry reach (88 percent of tags known to be present in Lees Ferry before the HFE). Six additional tagged fish were located upstream from Lees Ferry after the HFE that had not been located before the HFE, indicating significant tag detection problems. No fish were positioned at the exact same location throughout the duration of the study, indicating survivorship of tagged fish. No significant differences were determined in mean relative movement before and after the HFE among sexes (P = 0.69) and length classes (P = 0.36; table 1). Three tagged rainbow trout were detected by a remote receiver located 6 miles downstream from Lees Ferry 3–6 days before the HFE. The greatest documented movement of a tagged trout was more than 15.5 miles downstream and occurred before the HFE. The greatest upstream movement of a tagged trout was 11.2 miles and also occurred before the HFE. Individual fish movement was highly variable and did not relate to the occurrence of the HFE (fig. 5), length class, or sex (table 1). Average relative movement of tagged rainbow trout 305–405 mm tended to be less variable after the HFE.

Table 1. Average movement of acoustic tagged rainbow trout in Lees Ferry by size class and sex before and after the March 2008 high-flow experiment (HFE; mean ± 2 standard errors). Positive and negative values represent relative upstream and downstream movement, respectively. No significant differences were detected in movement before and after the HFE among sexes (P = 0.69) and length classes (P = 0.36).

[N, number; mm, millimeter]

Rainbow trout	Pre-HFE (miles)	N	Post-HFE (miles)	N	P-value
152–304 mm	0.3 ± 1.4	22	–0.9 ± 1.8	14	0.29
305–405 mm	0.1 ± 0.6	79	–0.1 ± 0.2	76	0.55
Female	0.3 ± 1.3	25	–0.6 ± 1.3	21	0.34
Male	–0.2 ± 1.0	33	–0.1 ± 0.2	34	0.75

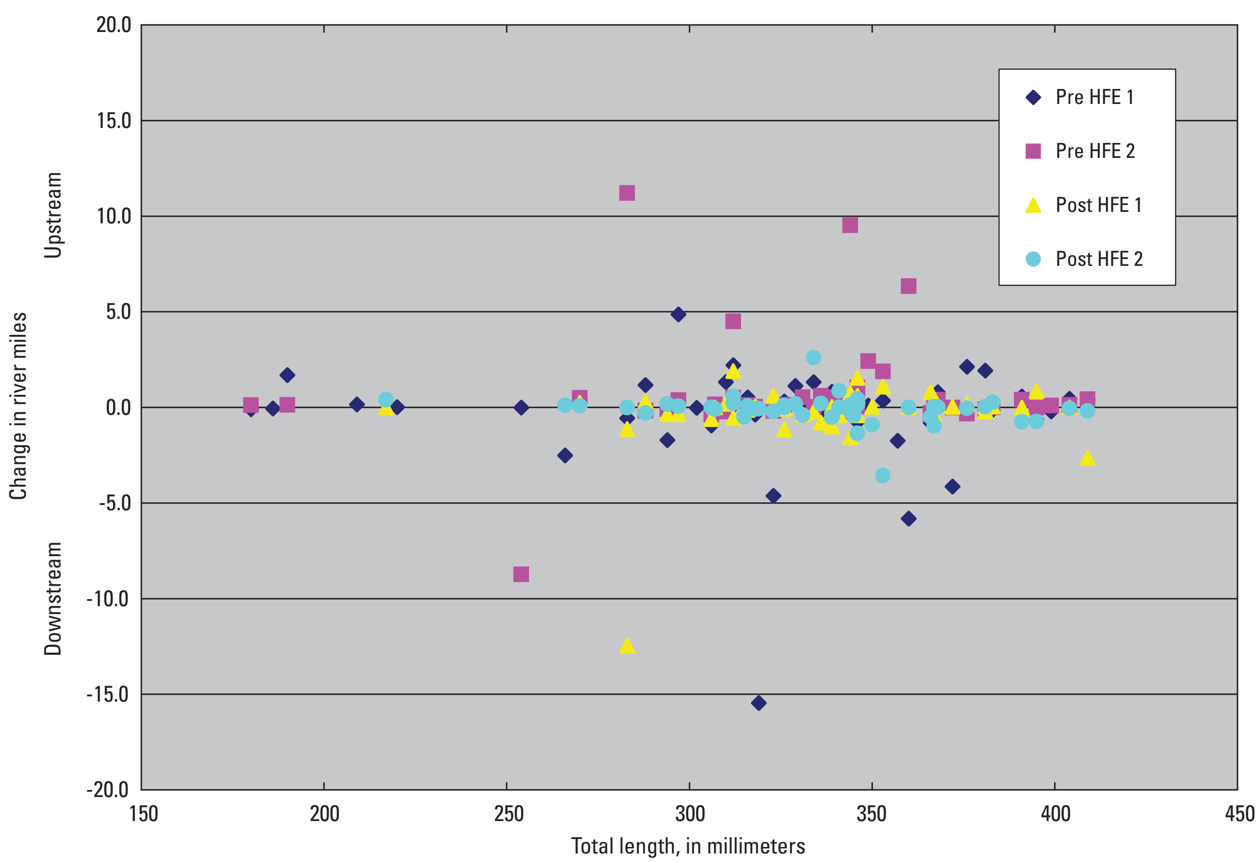

Figure 5. Scatter plot showing individual acoustic tagged rainbow trout movement in the Lees Ferry reach during the two tracking events before (Pre-HFE 1 and 2) and two tracking events after (Post-HFE 1 and 2) the March 2008 high-flow experiment (HFE). Individual tagged fish movement was highly variable and did not correlate to length or the occurrence of the HFE.

Discussion

Preliminary data from relative abundance indices and acoustic telemetry indicate the HFE conducted during March 2008 did not cause significant downstream movement of juvenile and adult rainbow trout below Lees Ferry. Relative abundance was similar before and after the experiment, which suggests that 41,500 ft^3/s did not cause significant displacement of rainbow trout downstream from the Lees Ferry reach for any size class fish (48 – 439 mm). The size structure of the rainbow trout sampled with electrofishing was similar before and after the March 2008 HFE, indicating no size-specific impacts. This assessment is supported by acoustic telemetry data, indicating 88 percent of tags located before the HFE were relocated after the HFE in Lees Ferry. Further, no significant difference in movement of tagged fish between 157– 404 mm occurred after the HFE. Telemetry data also indicate that movement did not relate to sex. The combined results indicate that no significant rainbow trout displacement occurred from the Lees Ferry trout fishery in association with the HFE.

Movement of rainbow trout in Lees Ferry was also investigated by using radio telemetry (Angradi and others, unpub. report, 1992). Eight tagged rainbow trout were located throughout a 1-year period in November 1990–1991 associated with various flow operations. Three tagged trout demonstrated substantial up and downstream movement of several miles (5+ miles) throughout the study. One tagged rainbow trout traveled 2 miles downstream from Lees Ferry and was not relocated during the duration of the study. Daily movement ranged from 0.02 to 0.08 miles during various flow regimes, and fish demonstrated considerable site fidelity. Methods for locating radio-tagged fish included triangulation to approximate location within a few feet, whereas methods used during this study were to locate tags to the nearest tenth of a mile (to accommodate locating 50 or more tags per day). Long-range movement observed during this study was consistent with long-range movement observed in radio-tagged rainbow trout. During both of these telemetry studies, tagged rainbow trout were observed dispersing downstream from Lees Ferry. This observed dispersal, though only four observations, indicates that rainbow trout from Lees Ferry can disperse into areas where angler access is limited and potentially have impacts on humpback chub in downstream reaches.

The March 2008 HFE appeared not to impact trout movement; however, study results from previous HFEs indicate a negative impact of large flows on young trout populations. Analysis of relative abundance data showed young rainbow trout (<152 mm) were not subjected to downstream displacement during the March 2008 HFE. This observation is supported by independent data (U.S. Geological Survey, unpub. data, 2008) in Lees Ferry, which indicate no change in absolute abundance for young trout (40–140 mm) immediately before and after the HFE. However, during the November 2004 HFE, a threefold decrease in abundance of young trout in Lees Ferry was observed (Korman, 2009). Temperatures of water released from Glen Canyon Dam during the November 2004 and March 2008 HFEs were approximately 15 °C and 8 °C, respectively. These data suggest the need for further studies to track the fate of young rainbow trout and other environmental and temporal factors that may increase young rainbow trout displacement risk during future HFEs. These factors may include water temperature, food availability, rainbow trout density, timing of the HFE, differences in ramp rates, diurnal timing of initial ramping, and other factors.

Implications for Management

Downstream movement of rainbow trout from Lees Ferry is a concern for managers of the Lees Ferry rainbow trout fishery and the endangered humpback chub population. The results of this experiment indicate that there was no significant impact of the March 2008 HFE on rainbow trout movement. However, during this study and a previous study (Angradi and others, unpub. report, 1992), tagged adult rainbow trout were observed dispersing downstream from Lees Ferry. In addition, Korman (2009) observed a threefold decrease in population size of age-0 trout in Lees Ferry during the November 2004 HFE. The fate of these age-0 fish was not directly measured; however, it was assumed that these fish likely displaced downstream or did not survive. These results suggest the need for further studies to track the fate of rainbow trout <150 mm and other factors that may cause adult fish movement downstream from Lees Ferry. This effort would require continuation of robust long-term monitoring protocols for all life-history stages of rainbow trout, development of more suitable individual fish tracking methods for fish <150 mm, and continued commitment to conducting experimental high flows in Grand Canyon.

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Mechanical Removal of Nonnative Fish in the Colorado River Within Grand Canyon

By Lewis G. Coggins, Jr.,[1,2] and Michael D. Yard[1]

Abstract

During 2003–2006, 23,266 nonnative fish were mechanically removed from critical humpback chub (*Gila cypha*) habitat in the Colorado River near the confluence of the Little Colorado River. This effort was conducted to evaluate the feasibility of nonnative control in the Colorado River and to document subsequent changes in the fish community within this river reach. While the fish community composition rapidly shifted from one dominated numerically by introduced rainbow (*Oncorhynchus mykiss*) and brown trout (*Salmo trutta*) to one primarily composed of native fishes and nonnative fathead minnows (*Pimephales promelas*) during mechanical removal efforts, the abundance of rainbow trout simultaneously declined throughout the Grand Canyon stretch of the Colorado River. As such, while mechanical removal efforts certainly impacted the fish community in this reach, the shift in fish community composition was also aided by environmental factors unassociated with nonnative control efforts.

Introduction

Native fish conservation is a key goal of the Glen Canyon Dam Adaptive Management Program primarily because humpback chub (*Gila cypha*), a native fish endemic to the Colorado River Basin, are protected under the U.S. Endangered Species Act (Public Law 93–205). Current knowledge suggests that factors influencing the humpback chub in Grand Canyon include: (1) nonnative fish (Gorman and others, 2005; Olden and Poff, 2005), (2) water temperature (Robinson and Childs, 2001), (3) flow regulation (Osmundson and others, 2002), (4) tributary rearing habitat (Stone and Gorman, 2006), and (5) parasites and disease (Choudhury and others, 2004). Of these factors, previous work has shown that factors 1–3 are likely dominant drivers of native fish population dynamics in this system (Walters and others, 2000) and suggests that improving rearing conditions in the mainstem Colorado River will likely provide the most significant benefit to native fish. Additionally, of the factors possibly influencing native fish population dynamics, controlled manipulation of factors 1–3 in an experimental framework is most tenable and, in recent years, has been the focus of efforts in adaptive management for native fish conservation (Grand Canyon Monitoring and Research Center, 2008).

Over the last several decades, the fish community in the Grand Canyon stretch of the Colorado River has consisted primarily of the nonnative salmonids rainbow trout and brown trout (Gloss and Coggins, 2005). Introductions of nonnative salmonids have been shown to adversely impact invertebrate (Parker and others, 2001), amphibian (Knapp and Matthews, 2000), and fish (McDowall, 2003) communities. These two species of fish have also been identified as particularly damaging invasive species (Lowe and others, 2000) mainly because of the global scope of introductions—rainbow trout have been successfully established on every continent with the exception of Antarctica. Although it is unclear how detrimental these fish are to native fish in the Colorado River, interactions with various nonnative fish have been widely implicated in the decline of Southwestern native fishes (Minckley, 1991; Tyus and Saunders, 2000). Nonnative salmonids, particularly brown trout, have been shown to be predators of native fish (Valdez and Ryel, 1995; Marsh and Douglas, 1997) in Grand Canyon, and rainbow trout predation on native fish has also been documented in other Southwestern United States systems (Blinn and others, 1993). Besides direct mortality through predation, both rainbow trout and brown trout have demonstrated other negative interactions with native fish in Western U.S. river systems, including interference competition, habitat displacement, and agonistic behavior (Blinn and others, 1993; Taniguchi and others, 1998; Robinson and others, 2003; Olsen and Belk, 2005). These lethal and sub-lethal effects of interactions with native fish have also been widely documented in New Zealand, Australia, Patagonia, and South Africa (McDowall, 2006).

While control of nonnative species is widely considered as a management option, it is less often implemented and evaluated (Lessard and others, 2005; Pine and others, 2007),

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particularly for fish in large river systems. Removal of nonnative organisms to potentially benefit native species is more frequently conducted in small streams (Meyer and others, 2006), lakes and reservoirs (Hoffman and others, 2004; Vrendenburg, 2004; Lepak and others, 2006), and terrestrial environments (Erskine-Ogden and Rejmanek, 2005; Donlan and others, 2007). However, recently much effort has been expended to remove or reduce nonnative fish in the Colorado River (Tyus and Saunders, 2000). Unfortunately, little documentation is available to evaluate the efficacy of these efforts (Mueller, 2005). This study describes one such effort and evaluates the efficacy of a program to reduce nonnative fish within humpback chub critical habitat in the Colorado River. Specifically, the objectives of this study were to evaluate the effectiveness of nonnative control efforts in the mainstem Colorado River and characterize changes in the fish community.

Nonnative Fish Control in Grand Canyon

The Little Colorado River (LCR) inflow reach of the Colorado River extends from 56.3 river mile[3] (RM) to 65.7 RM, as measured downstream from 0 RM at Lees Ferry, and is recognized as having the highest abundance of adult and juvenile humpback chub in the Colorado River (Valdez and Ryel, 1995). This reach also has a relatively high abundance of flannelmouth sucker (*Catostomus latipinnis*), bluehead sucker (*Catostomus discobolus*), and speckled dace (*Rhinichthys osculus*), owing to the availability of spawning and rearing habitat in the LCR. From January 2003 through August 2006, a total of 23 field trips were conducted to mechanically remove nonnative fish with serial depletion passes by using boat-mounted electrofishing within the LCR inflow reach. Following capture, nonnative fish were euthanized, and native fish were released alive. Rainbow trout abundance was estimated using depletion methods as described by Coggins (2008).

To determine if changes in the fish community in the LCR inflow reach were related to environmental factors and not the mechanical removal, a control reach was established upstream from the LCR inflow reach in an area of high rainbow trout density (44 RM–52.1 RM). During each trip, the control reach was sampled using methods similar to those described for the LCR inflow reach above. All captured fish were released alive, and nonnative fish larger than 200 millimeters (mm) total length were implanted with a uniquely numbered external tag to estimate abundance within the control reach.

Results of Mechanical Removal of Nonnative Fish in the LCR Inflow Reach

More than 36,500 fish from 15 species were captured in the LCR inflow reach during 2003–2006 (fig. 1; Coggins, 2008). The majority of these fish (23,266; 64 percent) were nonnatives and were dominated by rainbow trout (19,020; 52 percent), fathead minnow (2,569; 7 percent), common carp (*Cyprinus carpio*) (802; 2 percent), and brown trout (479; 1 percent). Catches of native fish amounted to 13,268 (36 percent) and included flannelmouth sucker (7,347; 20 percent), humpback chub (2,606; 7 percent), bluehead sucker (2,243; 6 percent), and speckled dace (1,072; 3 percent). The contribution of rainbow trout to the overall species catch composition fell steadily through the course of the study from a high of approximately 90 percent in January 2003 to less than 10 percent in August 2006. Overall, nonnative fish accounted for more than 95 percent of the catch in 2003 but following July 2005 generally contributed less than 50 percent. Owing to particularly large catches of flannelmouth sucker and humpback chub in September 2005, the nonnative contribution to the catch in that month was less than 20 percent. While the catch of nonnative fish generally decreased throughout the course of the study, catches of nonnative cyprinids (dominated by fathead minnows) increased in 2006.

The estimated abundance of rainbow trout in the LCR inflow reach ranged from a high of 6,446 (95-percent credible interval (CI) 5,819–7,392) in January 2003 to a low of 617 (95-percent CI 371–1,034) in February 2006; a 90-percent reduction over this time period (fig. 2; Coggins, 2008). Between February 2006 and the final removal effort in August 2006, the estimated abundance increased by approximately 700 fish to 1,297 (95-percent CI 481–2,825).

Control Reach Results

A total of 11,221 fish representing seven species were captured during control reach sampling (Coggins, 2008). The majority of fish captured were rainbow trout (10,648; 95 percent), followed by flannelmouth sucker (378; 3 percent) and brown trout (134; 1 percent). A general pattern of decreasing rainbow trout abundance was observed throughout the study, particularly following spring of 2005 (fig. 3). Rainbow trout abundance within the control reach was estimated at between 7,000 and 10,000 fish during 2003–2004 and between 2,000 and 5,000 during 2004–2005, suggesting that rainbow trout abundance likely declined by one-half or more between the first and last 2 years of the study.

[3] By convention, river mile is used to describe distance along the Colorado River in Grand Canyon. Lees Ferry is the starting point, RM 0, with mileage measured for both upstream (–) and downstream directions.

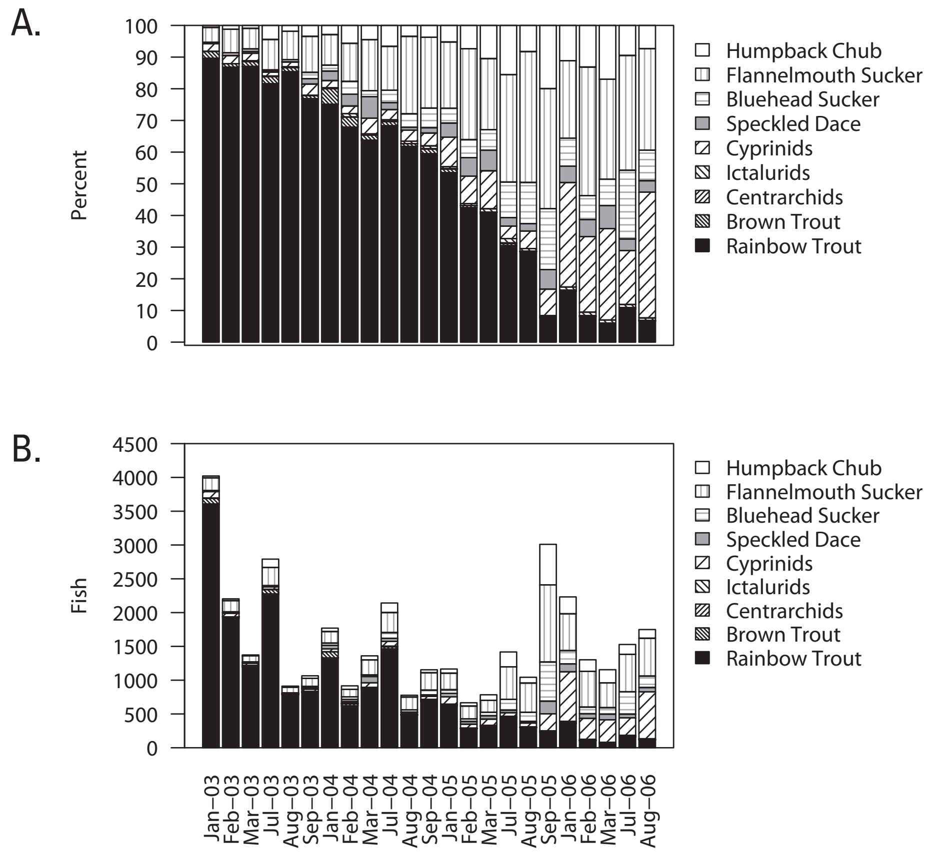

Figure 1. (A) Percent composition and (B) number of fish by species captured with electrofishing in the Little Colorado River inflow reach among months, 2003–2006.

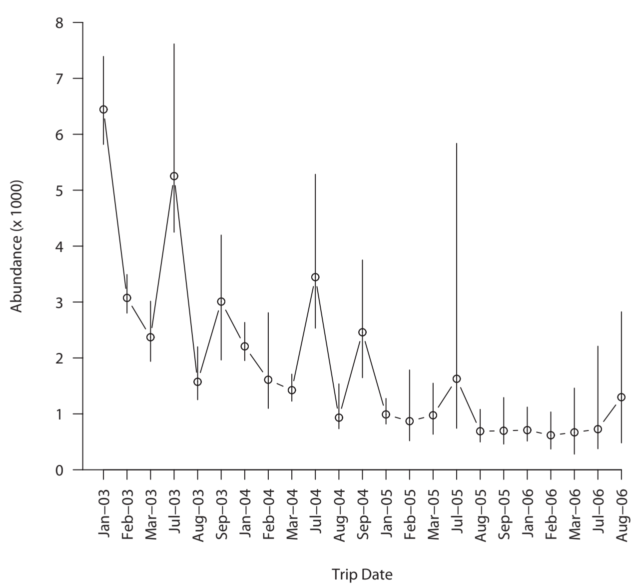

Figure 2. Estimated abundance of rainbow trout in Little Colorado River inflow reach among months, 2003–2006. Error bars represent 95-percent Bayesian credible intervals.

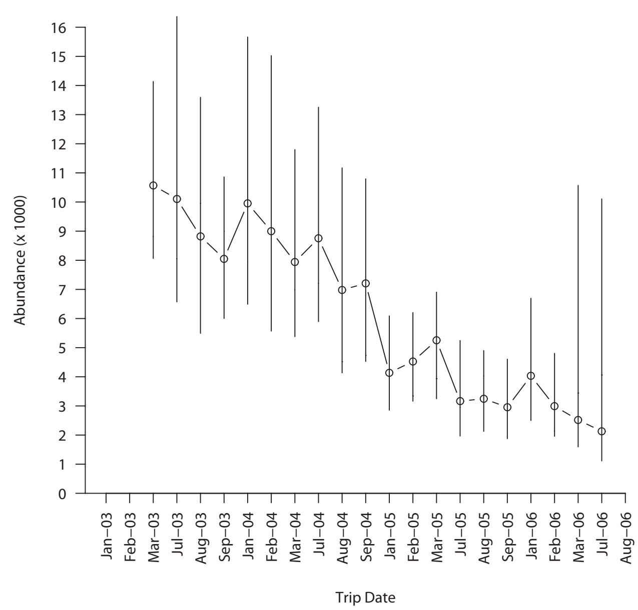

Figure 3. Estimated abundance of rainbow trout in the control reach among months, 2003–2006. Error bars represent 95-percent profile likelihood confidence intervals.

Comparison of Results from the LCR Inflow and Control Reaches

The abundance of rainbow trout declined throughout the study both in the LCR inflow reach and in the control reach; however, the pattern of decline was dissimilar between reaches (fig. 4). In the LCR inflow reach, the largest decline (62 percent) occurred between January 2003 and September 2004. Rainbow trout abundance in this reach declined much less rapidly from January 2005 to August 2006. In contrast, rainbow trout abundance in the control reach was constant to slightly declining from March 2003 through September 2004, but displayed a strong negative trend subsequently. These patterns suggest that removal efforts likely affected abundance in the LCR inflow reach predominantly during 2003 and 2004.

Another difference between the LCR inflow and control reaches was the seasonal patterns in rainbow trout abundance. In the LCR inflow reach, a pattern of declining abundance during each 3-month removal effort (for example, January–March) was followed by an increase in abundance at the beginning of the next series of removal efforts (for example, July–September), particularly during 2003–2004 (fig. 2). This pattern would be expected if the removal rate was greater than the immigration rate only during each removal series. This pattern was not evident in the control reach, suggesting that mechanical removal was influencing the abundance of rainbow trout in the LCR inflow reach.

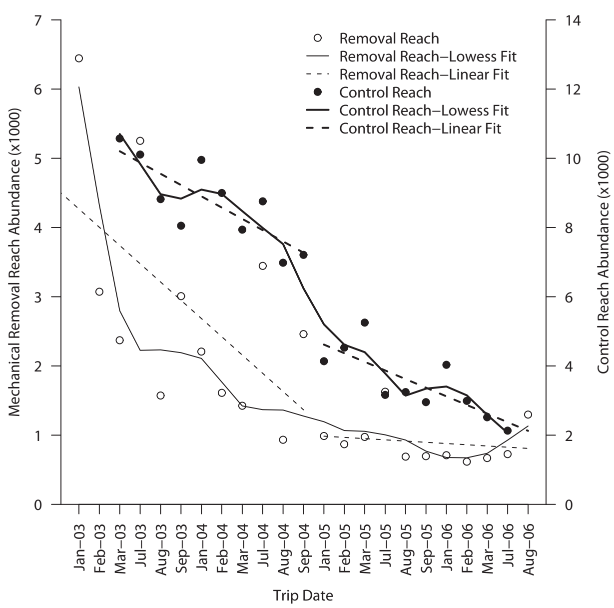

Figure 4. Estimated rainbow trout abundance in both the Little Colorado River inflow and control reaches at the beginning of each trip during 2003–2006. The solid lines represent the locally weighted polynomial regressions (Lowess) fit to each time series. The dashed lines represent linear regressions fit to either the 2003–2004 or 2005–2006 portions of the time series.

Implications for Management

Results suggest that the mechanical removal program was successful in reducing the abundance of nonnative fishes, primarily rainbow trout, in a large segment of the Colorado River in Grand Canyon. However, maintenance of low rainbow trout abundance in the LCR inflow reach was also facilitated by reduced immigration rates during 2005–2006 (Coggins, 2008) and a river-wide decline in abundance. The decline of rainbow trout abundance observed in the control reach was likely precipitated by at least two factors. First, rainbow trout abundance in the Lees Ferry reach (–15 RM to RM 0) of the Colorado River increased during approximately 1992–2001, and abundance in this reach steadily fell during 2002–2006 (Makinster and others, 2007). With the exception of limited spawning activity in select tributaries of the Colorado River in Grand Canyon, rainbow trout reproductive activity appears to be limited mainly to the Lees Ferry reach (Korman and others, 2005). Examination of length frequency distributions of rainbow trout captured using electrofishing from Glen Canyon Dam to RM 56 during 1991 through 2004 also supports the idea that Lees Ferry is the primary spawning site, as the juvenile size class of rainbow trout is largely absent from collections downstream from RM 10 (fig. 5). Thus, it is reasonable to conclude that at least for the last 10–15 years, the natal source of most rainbow trout in this system is the Lees Ferry reach. This conclusion is significant for management as it implies that abundance of rainbow trout in Grand Canyon is partially influenced by trends in rainbow trout abundance and reproduction in the Lees Ferry reach.

Second, it has been widely demonstrated that the density of rainbow trout is not uniform in the Colorado River below Glen Canyon Dam, and distribution patterns likely are influenced by food resources and foraging efficiency (Gloss and Coggins, 2005). Rainbow trout density generally declines with downstream distance from Glen Canyon Dam but exhibits punctuated declines below the confluences of the Paria River and the LCR. The density of algae and invertebrates in the Colorado River also decline along this gradient (Kennedy and Gloss, 2005), suggesting a possible link between distance from the dam and primary production. A major factor influencing these distributional patterns is sediment delivery from tributaries and the subsequent effects of elevated turbidity in the Colorado River in downstream sections. Yard (2003) demonstrated that these tributary inputs of sediment contribute to high turbidity and limit aquatic primary production. Trout are predominantly sight feeders—thus, high turbidity limits foraging efficiency and possibly survival by decreasing encounter rate and reactive distance to prey items (Barrett and others, 1992). Estimated rainbow trout survival rates in the control reach generally support the notion that rainbow trout experienced diminished survival rates during late 2004 and early 2005 (Coggins, 2008). This was a period of high turbidity owing to significant sediment inputs from the Paria River that also triggered an experimental high flow from Glen Canyon Dam in November 2004.

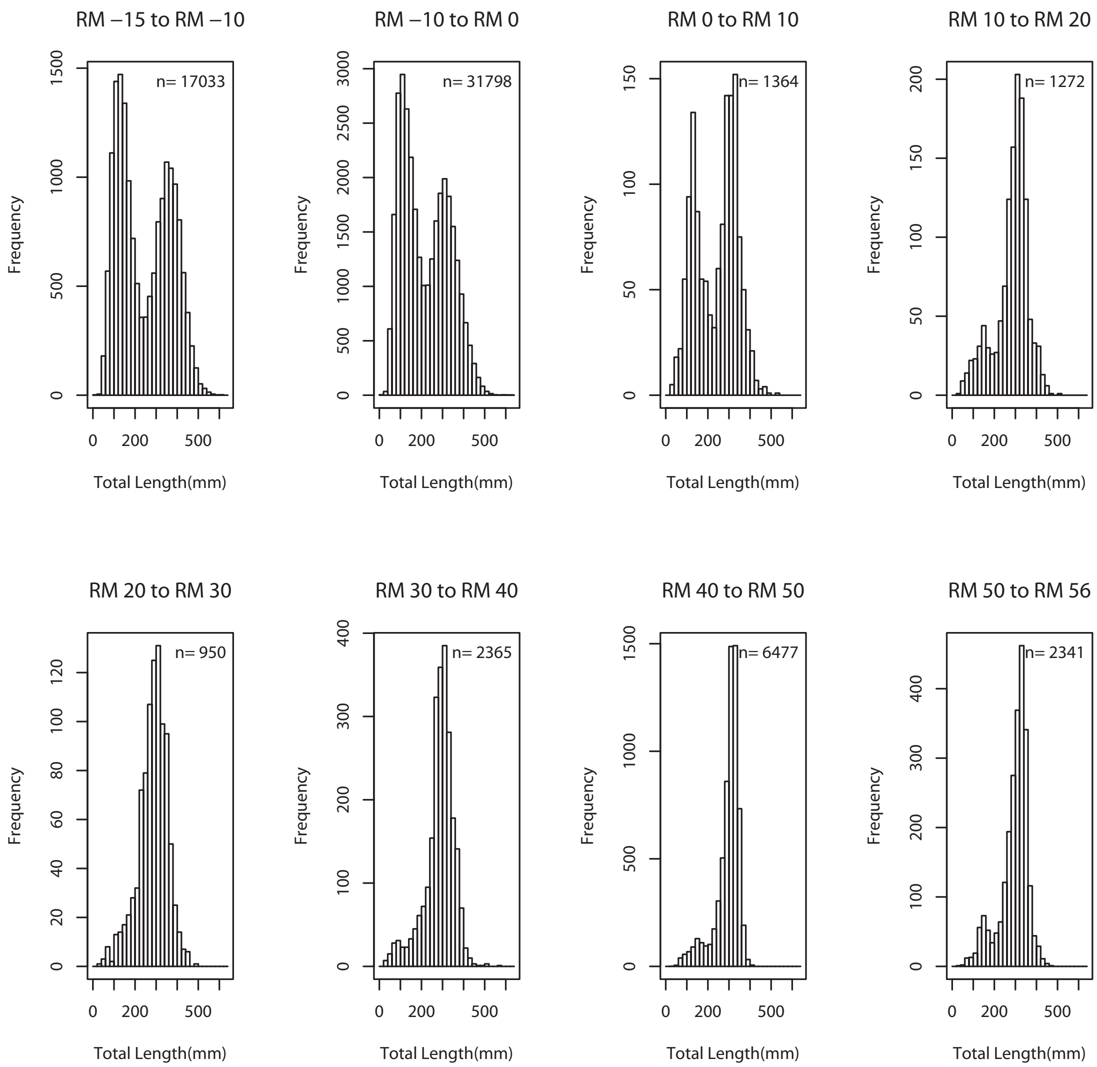

Figure 5. Length frequency distributions of rainbow trout captured during 1991–2004 using electrofishing in the Colorado River from river mile –15 to river mile 56. Each panel represents captures of fish within the identified river segment.

Other Species

Beginning in September 2005, large increases in the catch of nonnative fathead minnow and black bullhead (*Ameiurus melas*) were observed compared to the previous 17 trips, suggesting either increased immigration and (or) survival of these fish in the LCR inflow reach. Since these fish are not captured with any regularity in the control reach or in other sampling upstream from RM 44 (U.S. Geological Survey, unpub. data, 2008), it is reasonable to conclude that their source is not upstream. Stone and others (2007) documented the presence of these species and other warmwater nonnatives in the LCR ≈132 kilometers upstream from the confluence and suggested this tributary as the likely source of fathead minnow, black bullhead, and six other nonnative fish frequently encountered in the lower LCR and the LCR inflow reach. Thus, one possibility for the elevated catch of fathead minnow and black bullhead in the LCR inflow reach during this latter timeframe is an elevated emigration rate of these fish from the LCR. Alternatively, increasing water temperature, particularly in 2005 (Voichick and Wright, 2007), and the concurrent reductions in rainbow trout biomass may have influenced the survival and activity of these fish causing them to be both more abundant and more susceptible to capture. If warmer water discharges from Glen Canyon Dam continue into the future, it is likely that the nonnative fish community in the LCR inflow reach may shift to an assemblage dominated by fish less tolerant of cold water releases. Because many of these species are potentially both more difficult to control and more detrimental to native fish (Johnson and others, 2008), managers can usefully continue to support research aimed at developing better control methods for warmwater fish.

Recommendations for Future Mechanical Removal Operations

A recent biological opinion prepared by the U.S. Fish and Wildlife Service calls for continuation of mechanical removal of nonnative fish in critical humpback chub habitat (U.S. Fish and Wildlife Service, 2008). To more efficiently target nonnative species, further research is needed to better describe nonnative habitat selection. As an example, Royle and Dorazio (2006) present a technique to predict the density of organisms as a function of habitat characteristics that could be incorporated into future mechanical removal efforts with minimal modifications to current field procedures.

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Fish Management in National Park Units Along the Colorado River

By Melissa Trammell[1]

Abstract

The National Park Service (NPS) has a long cultural legacy of fishing. The National Park System was created for the enjoyment of the people as well as protection of natural beauty and resources—often referred to as a dual mandate. Fishing has been seen since the beginning of the National Park System as an important part of the enjoyment of the people—so much so that the NPS began stocking both native and non-native fish almost as soon as the first park, Yellowstone, was established in 1872. There are eight major national park units along the Colorado River from Colorado to Arizona covering 941 miles of river and including three national recreation areas. Fisheries management in parks is guided by law and policy that emphasizes native fish and ecosystem restoration; however, fisheries management in the recreation areas is primarily for recreational sportfishing, while the riverine parks support more native fish communities. In reservoirs, there has been a nearly complete displacement of all native fish species. Conditions in river reaches below dams favor nonnative and sportfishes over the native fish community. Much effort has been expended in removing or reducing the nonnative fishes in the Colorado River system. However, nonnative species remain abundant in many parks and support recreational fishing in many areas, and conflicts between management of native and nonnative species continue. To improve effective fish management in the Colorado River parks and help resolve conflicts, additional fisheries staff could be deployed, and fish management plans could be developed and implemented for each park in consultation and cooperation with States, anglers, and other affected parties. The NPS Fisheries Database could be kept current, and a Colorado River network within the NPS Inventory and Monitoring Program could be developed.

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Introduction

History and Policy

The National Park Service (NPS) has a long cultural legacy of fishing. The National Park System was created for the enjoyment of the people as well as protection of natural beauty and resources—often referred to as a dual mandate. Fishing has been seen since the beginning of the National Park System as an important part of the enjoyment of the people—so much so that the NPS began stocking both native and nonnative fish almost as soon as the first park, Yellowstone, was established in 1872. As time went by, park managers came to realize the damage that was being done to the native species, and now most stocking of nonnative species is prohibited (Sellars, 1997). However, nonnative species remain abundant in many parks and support recreational fishing in many areas, and conflicts between management of native and nonnative species continue.

The cultural legacy of fishing continues with the unique status of fish in the NPS; while removal or harvesting of all other natural resources in the parks generally is prohibited unless otherwise allowed, fishing is allowed unless otherwise prohibited. Fishing is further defined as one person fishing with hook and line. All other methods are prohibited unless specifically allowed by a park unit. Although fish are the only resource generally allowed to be harvested, the NPS still strives to manage the aquatic resources, including fish, according to guiding authorities and policies of conservation, which emphasize native species and ecosystem restoration. Significant habitat alterations as a result of dams and diversions, conflicting Federal and State policies, and invasive species and nonnative fish interactions all contribute to the difficulties managers face in achieving native and ecosystem restoration. Despite these challenges, few Colorado River parks have a designated fish biologist on staff. However, technical assistance on fisheries management is available from the national office of the Water Resources Division in Fort Collins, CO, and the regional fishery biologist.

The NPS has many guiding authorities, beginning with the National Park Service Organic Act of 1916, which famously directs the Park Service “to conserve the scenery and the natural and historic objects and the wildlife therein and to provide for the enjoyment of the same in such manner and by such means as will leave them unimpaired for the enjoyment of future generations.” The aquatic resources in national parks are protected and managed in a manner according to the mandates established by the following authorities among others:

- Wilderness Act of 1964 (Public Law 88–577)
- Wild and Scenic Rivers Act of 1968 (Public Law 90–542)
- National Environmental Policy Act of 1969 (Public Law 91–190)
- Clean Water Act of 1972 (Public Law 92–500)
- Endangered Species Act of 1973 (Public Law 93–205)
- Redwoods Act of 1978 (Public Law 95–250)
- Pollution Prevention Act of 1990 (Public Law 101–508)
- Executive Order 11987, Exotic Organisms, 1977
- Executive Order 11990, Protection of Wetlands, 1977
- Executive Order 12088, Federal Compliance with Pollution Control Standards, 1978
- Executive Order 13112, Invasive Species, 1999
- NPS Management Policies 2006

The NPS Management Policies (National Park Service, 2006) contain more specific guidance on how to go about conserving our resources unimpaired. Excerpts from the policies direct the NPS to maintain as parts of the natural ecosystems of parks all native plants and animals through

- Preserving and restoring the natural abundance, diversities, dynamics, distributions, genetic and ecological integrity, and behaviors of native species and the communities and ecosystems in which they occur;
- Restoring native species in parks when they have been extirpated by past human-caused actions;
- Initiating the return of human-disturbed areas to natural conditions (or the natural trajectory), including the processes characteristic of the ecology zone;
- Minimizing human impacts on native species, communities, and ecosystems, and the processes that sustain them;
- Preventing the introduction of exotic species and removing established populations;
- Monitoring natural systems and human influences upon them to detect change and developing appropriate management actions; and
- Protecting watersheds, as complete hydrologic systems, primarily by avoiding impacts to watershed and riparian vegetation, and by allowing natural fluvial processes to proceed unimpeded.

Regulations in the Code of Federal Regulations (CFR) and the NPS Management Policies specific to fishing emphasize the importance of working in consultation with the States where the parks are situated and with the State laws and regulations; however, the National Park Service Organic Act of 1916 grants the Secretary of the Interior the authority to implement rules and regulations deemed necessary or proper for the use and management of lands and waters under the jurisdiction of the NPS. The CFR section on fishing states the following:

> (a) Except in designated areas or as provided in this section, fishing shall be in accordance with the laws and regulations of the State within whose exterior boundaries a park area or portion thereof is located.

Non-conflicting State laws are adopted as a part of these regulations (36 CFR Chap. I § 2.3 Fishing). Further, the NPS Management Policies Section 8.2.2.5 on fishing states the following:

> Recreational fishing will be allowed in parks when it is authorized or not specifically prohibited by federal law provided that it has been determined to be an appropriate use per Section 8.1 of these policies. When fishing is allowed, it will be conducted in accordance with applicable federal laws and treaty rights, and non-conflicting state laws and regulations… representatives of appropriate tribes and state and federal agencies will be consulted to ensure that all available scientific data are considered in the decision-making process.

The relation of the NPS with the States is further defined by three levels of regulatory jurisdiction: exclusive, concurrent, and proprietary. In parks with exclusive jurisdiction, the NPS has primary regulatory authority, though State regulations are usually adopted. In concurrent jurisdictions, regulatory authority is shared with the State, and State regulations are usually adopted unless there is a conflict with management objectives of the park. The NPS has less authority to impose restriction in parks with proprietary jurisdiction, where regulatory authority rests primarily with the State. In practice, the NPS almost always adopts the rules, regulations, and management of the adjacent State, unless there is a major management conflict.

A Heritage of Fishing—the NPS Recreational Fisheries Program

In 1992, the NPS adopted its recreational fisheries program, "A Heritage of Fishing." Program purposes are to improve the management of fishery resources, improve public understanding of aquatic ecology and angler ethics, promote research into management of quality fisheries and the contribution of fish to ecosystem processes, and increase the number and quality of recreational opportunities available to the public both inside and outside of the National Park System. This program established the framework for the NPS to continue to provide unique fishing opportunities while restoring and protecting native fishes and their associated ecosystems. The NPS also developed a national fisheries database containing important information on species, management objectives, existing plans and projects, and management concerns for each park. The Water Resources Division administers this program from their national office in Fort Collins, CO.

Colorado River Parks

The NPS manages a substantial portion of public lands along the Colorado River. There are eight large parks along the Colorado River and four of its major tributaries, including Rocky Mountain National Park (NP) at the headwaters of the Colorado River. Numerous other parks are on smaller tributaries such as Capitol Reef NP on the Fremont River and Zion NP on the Virgin River. The NPS manages about one-third of the total river miles, including reservoirs in the Colorado River system, and almost half of the Colorado River itself (table 1). NPS influence over fisheries management in the parks has been limited except in the case of endangered fish; however, the NPS participates in conservation agreements for several native species, including roundtail chub (*Gila robusta*), flannelmouth (*Catostomus latipinnis*) and bluehead sucker (*Catostomus discobolus*), and several subspecies of cutthroat trout (*Oncorhynchus clarkii* spp.) (table 2). We support conservation actions that improve and stabilize fish habitat and native fish populations.

Fisheries Resources in Colorado River Parks

The Colorado River parks encompass a wide variety of fishery resources including cold, cool, and warmwater species, and reservoirs, rivers, and streams (table 2). The three national recreation areas (NRA) were created around large reservoirs and were intended to provide recreational fishing opportunities. Blue Mesa Reservoir in Curecanti provides excellent fishing for lake trout (*Salvelinus namaycush*) and kokanee salmon (*Oncorhynchus nerka*), while Lake Powell and Lake Mead are warmwater fisheries with striped bass (*Morone saxatilis*), black bass (*Micropterus* spp. Lacepède, 1802), and panfish. The rivers below dams are cooler than the natural rivers and often support world-class trout fisheries. The warmer sections of the rivers often harbor catfish or bass

Table 1. Total river miles and miles managed by the National Park Service. Total river miles includes river and reservoir miles in the Colorado River up to Grand Lake below Rocky Mountain National Park, the Green River up to Fontanelle Dam, the Gunnison River to the upper end of Blue Mesa Reservoir, the Yampa River below Catamount Reservoir, and the San Juan River below Navajo Dam.

[NRA, National Recreation Area; NP, National Park; NM, National Monument]

Park unit	Colorado	Green	Gunnison	Yampa	San Juan	Total
	National Park Service river miles including reservoirs					
Lake Mead NRA	139					138
Grand Canyon NP	275					275
Glen Canyon NP	216				100	316
Canyonlands NP	49	46				95
Dinosaur NM		45		50		95
Curecanti NRA			9			9
Black Canyon NP			12			12
Total park miles	679	91	21	50	100	941
Total river	1,450	730	200	200	224	2,804
Percent park miles	46.8%	12.5%	10.5%	25.0%	46.6%	33.6%

Table 2. Native fishes of the Colorado River, Federal listing status, and National Park Service occurrence.

[C, candidate; E, endangered; T, threatened]

Family	Species	Common name	Federal listing status	Occurs in NPS
Catostomidae	*Catostomus clarkii*	desert sucker	None	Y
	Catostomus discobolus	bluehead sucker	None	Y
	Catostomus insignis	Sonora sucker	C	Y
	Catostomus latipinnis	flannelmouth sucker	None	Y
	Catostomus platyrhynchus	mountain sucker	None	Y
	Catostomus latipinnis spp	Little Colorado sucker	None	N
	Xyrauchen texanus	razorback sucker	E	Y
Cottidae	*Cottus bairdii*	mottled sculpin	None	Y
	Cottus beldingii	Paiute sculpin	None	N
Cyprinidae	*Agosia chrysogaster*	longfin dace	C	Y
	Gila cypha	humpback chub	E	Y
	Gila elegans	bonytail	E	Y
	Gila intermedia	Gila chub	E	?
	Gila jordoni	Pahranagat roundtail chub	E	N
	Gila nigra	headwater chub	C	Y
	Gila robusta	roundtail chub	C	Y
	Gila seminuda	Virgin River chub	E	N
	Lepidomeda albivallis	White River Spinedace	E	N
	Lepidomeda altivelis	Pahranagat spinedace	Extinct	N
	Lepidomeda mollispinis mollispinus	Virgin River spinedace	C	Y
	Lepidomeda vittata	Little Colorado River spinedace	T	N
	Meda fulgida	spikedace	E	Y
	Moapa coriacea	Moapa dace	E	N
	Plagopterus argentissimus	woundfin	E	N
	Ptychocheilus lucius	Colorado pikeminnow	E	Y
	Rhinichthys cobitis	loach minnow	E	Y
	Rhinichthys deaconi	Las Vegas dace	Extinct	N
	Rhinichthys osculus	speckled dace	None	Y
	Rhinichthys osculus moapae	Moapa speckled dace	None	N
	Rhinichthys osculus thermalis	Kendall warm springs dace	E	N
Cyprinodontidae	*Crenichthys baileyi baileyi*	Moapa White River springfish	T	N
	Cyprinodon macularius spp.	Monkey Spring pupfish	Extinct	N
	Cyprinodon macularius	desert pupfish	E	Y
Poeciliidae	*Poeciliopsis occidentalis*	Gila topminnow	E	Y
Salmonidae	*Oncorhynchus apache*	Apache trout	T	N
	Oncorhynchus clarkii	cutthroat trout	None	Y
	Oncorhynchus gilae	Gila Trout	T	Y
	Prosopium williamsoni	mountain whitefish	None	Y
Elops affinis	*Elops affinis*	machete	None	N
Mugilidae	*Mugil cephalus*	striped mullet	None	N

species such as channel catfish (*Ictalurus punctatus*) and largemouth (*Micropterus salmoides*) and smallmouth bass (*Micropterus dolomieu*). Major fisheries resources in Colorado River parks include the following:

Reservoirs
- Blue Mesa in Curecanti NRA
- Lake Powell in Glen Canyon NRA
- Lake Mead and Lake Mohave in Lake Mead NRA

Rivers
- Gunnison River in Black Canyon of the Gunnison NP
- Green and Yampa Rivers in Dinosaur National Monument (NM)
- Green and Colorado Rivers in Canyonlands NP
- Colorado River in Glen Canyon NRA, Grand Canyon NP, Lake Mead NRA
- San Juan River in Glen Canyon NRA

Tributaries
- Most parks
- Both warm and cool water tributaries

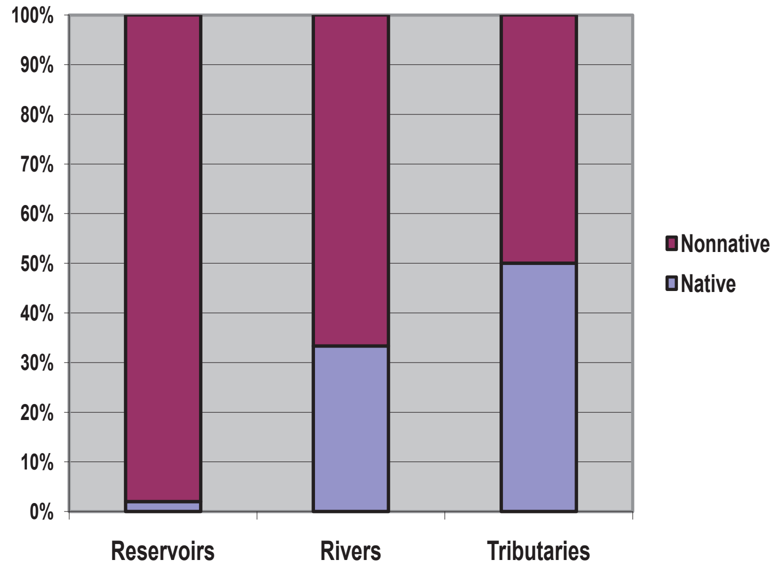


Figure 1. Ratio of number of native and nonnative fish species in Colorado River park reservoirs, rivers, and tributaries.

Nonnative species dominate all waters in each of the parks. In the reservoirs formed by the major dams, nearly 100 percent of the species are nonnatives, while in the rivers, the ratio is closer to 35 percent native. Smaller tributaries sometimes fare better, with 50 percent native species composition (fig. 1). Species data were derived from NPSpecies, an NPS database documenting species occurrence and status in each park. At least 62 nonnative species have been introduced into the Colorado River system (Olden and others, 2008), but "only" 25–30 have become well established in any one park. The nonnatives usually comprise a larger proportion of biomass and total numbers of fish, as there are fewer individuals of the native species. Native species are severely compromised throughout the Colorado River, and the parks are no exception.

Nearly all of the sportfish in the Colorado River are introduced species, with the exception of salmonid species native to some parks. Although some of the native fish grow quite large and could provide sportfishing opportunities, many are not well valued by anglers. Fifteen native species are listed as threatened or endangered under the Endangered Species Act (table 2), and several more have been proposed for listing and are listed by the States as sensitive.

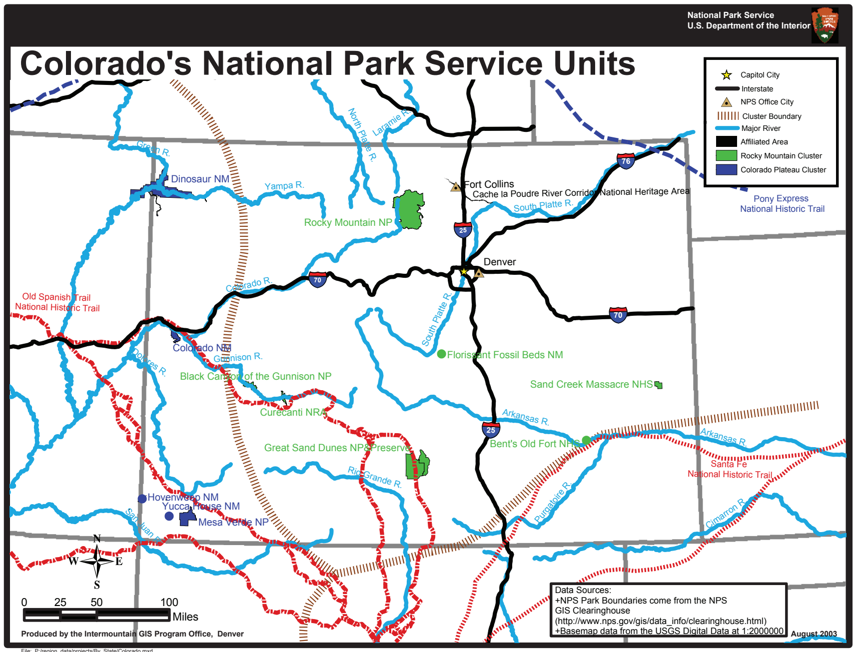
Curecanti National Recreation Area

Curecanti NRA has concurrent jurisdiction with the State of Colorado (fig. 2a). Curecanti NRA is composed of three reservoirs along the Gunnison River—Blue Mesa, Morrow Point, and Crystal. Blue Mesa is the largest and receives the majority of use and management. All three reservoirs are managed for salmonid sportfishing for kokanee and lake trout. Kokanee is the preferred species by the Colorado Division of Wildlife (CDOW) and most anglers, but lake trout have support from trophy anglers (Patrick Martinez, CDOW, oral commun., June 2009). Kokanee are allowed to be stocked into the reservoir. Eggs are harvested from spawning kokanee that migrate and are captured just upstream from the park and are stocked in many other State waters in addition to Blue Mesa. Kokanee are a major economic force in the area and support the tourist economy statewide through the egg harvest and stocking program. Other salmonids reproduce naturally.

Recent illicit introductions of northern pike (*Esox lucius*) and yellow perch (*Perca flavescens*) threaten the fishery, as does the possibility of the introduction of quagga (*Dreissena bugensis*) or zebra mussels (*Dreissena polymorpha*). The State and the park are taking measures to prevent the introduction of these extremely invasive species. All boats that launch are required to complete a self-certification form and display it on the vehicle. If there is a risk of contamination, hot water decontamination systems are required and are located at the main boat ramps. These stations are manned during high-use hours. However, the back country boat ramps are not patrolled, and in 2009, only about one-half of trailered vehicles showed the self certification as required.

The only native species restoration is confined to tributaries. Many small streams enter the reservoir, and some restoration of Colorado River cutthroat trout is being implemented. Trout are the only native species widely considered to be game fish and thus contributing to recreation. Consequently, restoration of trout species is more widely supported by the public than other lesser-known native fishes; thus, there are more funding opportunities for game fish restoration.

(a)



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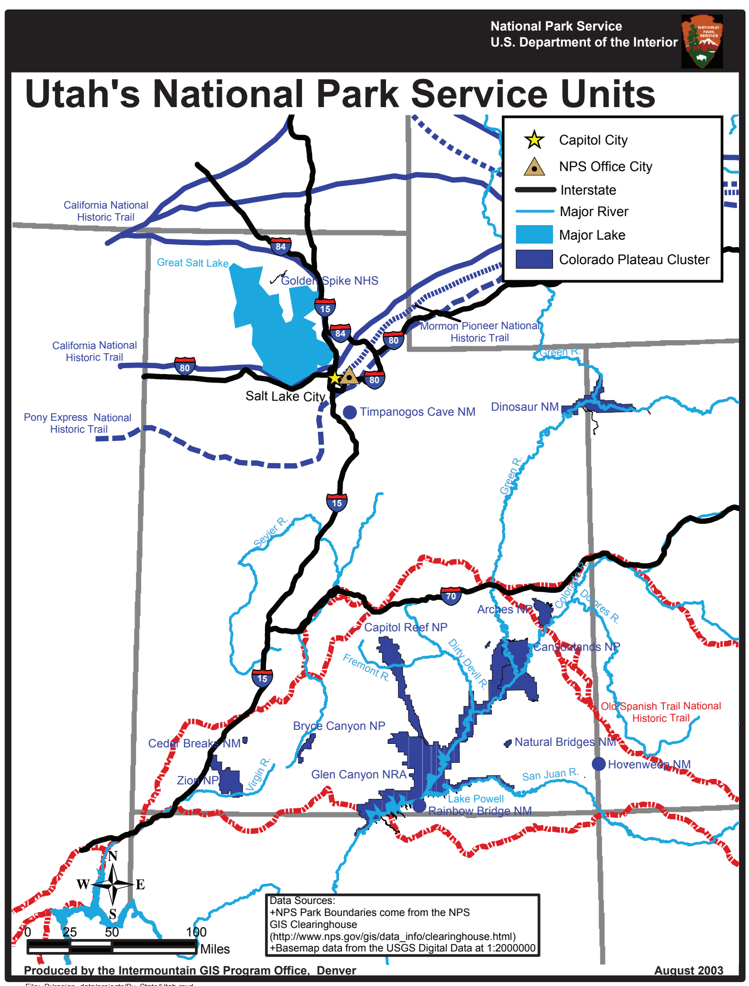

Figure 2. National Park units along the Colorado River system in (a) Colorado, (b) Utah, and (c) Arizona/Nevada.

(c)

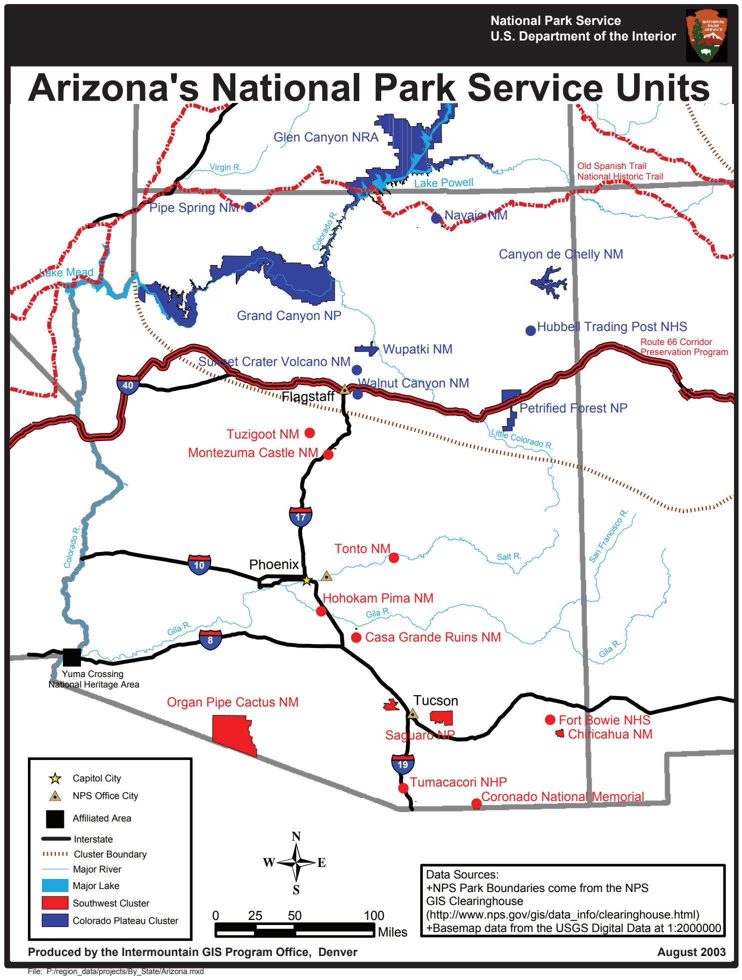

Figure 2. (continued) National Park Units along the Colorado River system in (a) Colorado, (b) Utah, and (c) Arizona/Nevada.

Black Canyon of the Gunnison National Park

Black Canyon of the Gunnison NP also has concurrent jurisdiction with the State of Colorado (fig. 2a). This park is managed for sportfishing, primarily for the nonnative rainbow (*Oncorhynchus mykiss*) and brown trout (*Salmo trutta*). The park is directly upstream and contiguous with a highly valued fishery maintained by CDOW on Bureau of Land Management lands. Whirling disease has severely impacted this fishery in recent years, and the CDOW is actively working to restore the rainbow trout fishery by stocking whirling disease-resistant Harrison-Hofer rainbow trout. However, stocking of this nonnative strain does not occur in the park.

The restoration of the native Colorado River cutthroat trout is desirable, but not considered realistic in the Gunnison River in the park because of the adjacent rainbow trout fishery. A few native warmwater species remain despite the cooler water released from the upstream dams. Flannelmouth sucker, bluehead sucker, speckled dace (*Rhinichthys osculus*), and mottled sculpin (*Cottus bairdii*) can still be found in small numbers in the park (table 2).

Dinosaur National Monument

Dinosaur NM (fig. 2a) is the only park with a full complement of native fish species, although some are very rare (cutthroat, humpback chub (*Gila cypha*)) and some are only present because they are stocked in the park or in contiguous rivers (bonytail (*Gila elegans*), razorback sucker (*Xyrauchen texanus*)) (Bestgen and others, 2007). Two rivers flow through Dinosaur NM and reach their confluence in the center of the park. The Green and Yampa Rivers are similar in size, but the Yampa River is largely free flowing and undeveloped, while the Green River is highly regulated by the upstream Flaming Gorge Dam. Dinosaur NM spans two States and has proprietary jurisdiction with both Utah and Colorado. Dinosaur is primarily managed for native species by the States and the park. However, anglers can fish for brown and rainbow trout, which are abundant in the Green River particularly above its confluence with the Yampa River, and smallmouth bass and channel catfish are found throughout both rivers. Although some fishing does take place, recreational rafting is the primary attraction by river users, and visitation to both rivers is limited by the number of boating permits issued by the park. Access is extremely limited other than by boat. Jones Hole Creek flows into the Green River in Utah and is a popular fishing area for rainbow trout.

The Upper Colorado River Basin Endangered Fish Recovery Program (Program) was created to recover four endangered fishes of the upper Colorado River: Colorado pikeminnow (*Ptychocheilus lucius*), razorback sucker, humpback chub, and bonytail (table 2). One of the recovery elements is the control and management of nonnative species that negatively impact the endangered fish. The Program has implemented a large-scale nonnative removal program in the Green and Yampa Rivers focused on removing northern pike and smallmouth bass, which are considered to have the most impact on the native fish community through predation and competition (Valdez and others, 2008). Channel catfish are also targeted for removal in some areas. Removal efforts in the park are conducted by the States, the U.S. Fish and Wildlife Service (FWS), and Colorado State University and are funded by the Program. Although brown and rainbow trout are known to prey upon native and endangered fishes (Coggins, 2008), these species are not part of the removal efforts.

Canyonlands National Park

Canyonlands NP also has proprietary jurisdiction with the State of Utah (fig. 2b). The Green River joins the Colorado River within the park, and the Colorado River continues to flow through Cataract Canyon into Glen Canyon NRA and Lake Powell. Canyonlands NP has a healthy native fish community, but there has been little management activity. The Utah Division of Wildlife Resources conducts monitoring of endangered and native fishes in the park for the Program. Channel catfish are plentiful but little sportfishing occurs, although it is not prohibited. Some removal of small-bodied nonnative cyprinids was attempted on an experimental basis in the past (Trammell and others, 2004), but no current removal efforts are underway.

Glen Canyon National Recreation Area

Glen Canyon NRA was established by Public Law 92–593 "to provide for public outdoor recreation use and enjoyment of Lake Powell and lands adjacent thereto… and to preserve the scenic, scientific, and historic features contributing to public enjoyment of the area." This legislation specifically mandates recreational fishing. Glen Canyon NRA is a large, complex area that includes Lake Powell, parts of the Colorado and San Juan Rivers, and the smaller tributary Escalante and Dirty Devil Rivers (fig. 2b). Encompassing nearly 2,000 miles of shoreline and over 180 miles in length, Lake Powell is the second largest reservoir of the Colorado River parks in water volume after Lake Mead. Most of the park is within the State of Utah although Glen Canyon Dam, a portion of the reservoir, and the 15-mile Lees Ferry reach of the Colorado River are in Arizona. Glen Canyon has proprietary jurisdiction with both Arizona and Utah.

Providing a quality recreational fishery is congruous with the NPS recreational fishing program, "A Heritage of Fishing." This program encourages all park units with fishery resources to develop fish management plans in consultation with the States; however, Glen Canyon NRA is the only park along the Colorado River that has an established plan (National Park Service, 2002). The plan was developed in consultation with Arizona and Utah to resolve fisheries management issues and provide for both an outstanding recreational sport fishery as well as preservation of native fish species. Although the plan covers both Lake Powell and Lees Ferry, the Lees Ferry section is a brief one page. Both areas are managed primarily for sportfishing. This 5-year plan is due for revision, and discussions among the participating agencies are ongoing.

In Lake Powell, native fish are now limited to the tributary arms of the Colorado and San Juan Rivers, although they can still be found upstream in the Escalante and Dirty Devil arms as well. The NPS supports native endangered fish through the Programs. Endangered fishes occur only near the inflow areas and consist of fish stocked in the rivers above the park. A program to reintroduce bonytail into a naturally dammed pond on Iceberg Canyon will begin in 2010. The NPS most recently has concentrated on the prevention of quagga/zebra mussels and other water-quality issues, while Utah manages the superb recreational fishery.

Lees Ferry is managed for sportfishing by Arizona Game and Fish Department (AGFD). The cold water released from Glen Canyon Dam supports a large population of rainbow trout. This spectacular fishing destination supports several fishing guides and the local economy. However, there are ongoing concerns about the contribution of this population of nonnative fish to downstream populations in Grand Canyon NP and their interaction with the endangered humpback chub. Native fishes are still present in the Lees Ferry reach, and flannelmouth sucker are found near the mouth of the Paria and near some warm springs a few miles upstream.

Grand Canyon National Park

Grand Canyon NP has exclusive jurisdiction (fig. 2c). There is no fish management plan, but fish and aquatic resources were considered as part of the Colorado River Management Plan (CRMP) and Environmental Impact Statement (EIS) (National Park Service, 2005). The CRMP primarily deals with recreational rafting impacts, but also addresses angling and native fish restoration. The management objective for aquatic resources is to manage river recreation use in a manner that protects native aquatic organisms, reduces aquatic habitat alteration, and minimizes the spread of exotic species. Specific management actions (contingent on funding) include a fishing ban within 1 mile of the Little Colorado River to protect the endangered humpback chub, a survey of streams and tributaries for native fishes, and, with Lake Mead NRA, a survey of the lower Grand Canyon and interface area for spawning razorback sucker. On the mainstem Colorado River, actions will continue to support adaptive management program activities within the park such as research, control of trout near the lower Colorado River, development of a nonnative fish control strategy, and implementation of conservation measures from several recent compliance documents. In the tributaries, actions will include translocation of humpback chub into Shinumo Creek and potentially other tributaries and removal of nonnative species.

Lake Mead National Recreation Area

The enabling legislation for Lake Mead NRA (Public Law 88–639) established the recreation area "for the general purposes of public recreation, benefit, and use…" Lake Mead NRA is composed of three reservoirs linked by short stretches of the Colorado River: Lake Mead, Lake Mohave, and Lake Havasu (fig. 2c). Lake Mead is the largest of the three reservoirs and rivals Lake Powell in size and complexity. In addition to the Colorado River inflow, the Virgin River is another large tributary. Lake Mead NRA shares proprietary jurisdiction with Nevada and Arizona. The Nevada Department of Wildlife is the primary wildlife management agency. The Lake Mead Lake Management Plan (National Park Service, 2003) was developed in cooperation with several agencies including FWS, Bureau of Reclamation, U.S. Geological Survey, as well as the States of Nevada and Arizona. This park is managed for sportfishing to provide public recreation. Similar to Lake Powell, the sportfish are nonnative and include striped and black bass and catfish, as well as stocked trout below the dams. The plan includes protection for native species. Most of the native fishes have been extirpated; however, endangered razorback sucker and bonytail still exist. The razorback sucker has been the subject of a long-term effort to augment the population by harvesting larvae from spawning adults, raising them in hatcheries or other predator-free environments, and repatriating them to the reservoirs (Albrecht and others, 2008). Bonytail are rare but are stocked annually in Lake Mohave and Lake Havasu. The plan includes closing of known spawning sites during spawning, monitoring of other sites during marina expansion to detect spawning, surveying for new spawning sites, with closures if necessary, and continuing repatriation and creation of new isolated cove-based refugia.

Implications for Management

The NPS policy is to manage all park units on the same principle: "to leave unimpaired for the enjoyment of future generations." However, national parks and monuments often differ in management practice from the NRAs—a consequence of the NPS "dual mandate." The enabling legislation for a NRA is often very clear about including recreational fishing as a park purpose, but fishing is often not mentioned in national park legislation—fish are usually considered one of the "natural resources" a given park was created to protect (Sellars, 1997).

In practice, the States generally do a good job of managing fishery resources within the parks—particularly sportfishing resources—and the NPS generally accepts their management direction. The NPS and the States sometimes differ when it comes to native fish management, and conflicts can arise between native fish conservation and nonnative sportfishing. Although the NPS endeavors to support native fish management where practicable, existing nonnative sportfisheries are often allowed to continue even where there is a conflict with native fish, and stocking of nonnatives continues in some areas closely adjacent to parks. As a part of the "A Heritage of Fishing" Program, each unit of the NPS with fishery resources is expected to develop a management plan and agreement with the States. However, Glen Canyon NRA is the only park along the Colorado River with a management plan, and it is overdue for revision and renewal.

Suggestions for Fish Management

Develop a fish management plan for each park. The NPS policies emphasize the need to work in consultation with the States and other interested parties when developing fish management plans. The process of developing a plan allows prioritization of species management and would help resolve conflicts in management objectives. Since fish management needs often transcend park boundaries, a multiparty plan could incorporate ecosystem restoration principles on a larger scale than possible within one park.

Fish management plans provide:

- An identification of the species that will be managed within the park,
- The desired conditions to be achieved,
- How the resources will be monitored to determine if the desired conditions are being achieved,
- Locations of fishermen access and other physical facilities to be maintained,
- Process by which regulations will be set,
- Protocols and working relations among the agencies involved,
- Monitoring activities to be conducted,
- Research and information needs.

Revise and maintain current information in NPS fisheries database. The NPS developed a fisheries database that contains important information on species, management objectives, existing plans and projects, and management concerns. The database serves as a reference tool for the storage and retrieval of information that is necessary for the management and protection of fishery and aquatic resources, which are extremely diverse and geographically dispersed. The database could also provide an institutional record in the event of staff turnover. However, to perform this function the database needs to be continually updated and revised as plans are completed and new information becomes available.

Develop Colorado River network within NPS Inventory and Monitoring Program. The NPS Inventory and Monitoring Program facilitates greater understanding and promotes science-driven management of natural resources. The program is divided into networks that cover geographically and biologically similar areas. The Colorado River parks are linked by the most important water resource in the region, and building a network around these parks would allow more comprehensive understanding and management of aquatic resources in these disparate parks.

Increase fish biologist staff in Colorado River parks. While existing park resource staff are concerned about fish management in the parks, direct management of fisheries is often deferred to the States because of park workload. Increasing the number of trained fish biologists available to parks would allow enhanced understanding and management guided by NPS policy.

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Bat Monitoring at Habitat Creation Areas as Part of the Lower Colorado River Multi-Species Conservation Program

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Abstract

The Lower Colorado River Multi-Species Conservation Program includes conservation measures for four bat species: the western red bat (*Lasiurus blossevillii*), the western yellow bat (*Lasiurus xanthinus*), the California leaf-nosed bat (*Macrotus californicus*), and the Townsend's big-eared bat (*Corynorhinus townsendii*). These measures include creating suitable habitat for each species. Monitoring existing habitat creation areas is required to aid in the adaptive management process by identifying what types of habitat will be designated for each species in the future. Monitoring of current habitat creation areas includes both acoustic and capture survey methods. Acoustic surveys are conducted using Anabat™ bat detectors, which are used to create an index of bat activity for each habitat type being monitored. Capture methods include the use of mist nets and harp traps. A total of 16 species have been recorded acoustically, and 9 species have been captured. Together, the two survey methods provide a good picture of bat use for each habitat creation area. These preliminary data will be used during the adaptive management process to further direct restoration and management of existing and future habitat creation areas.

Introduction

The Bureau of Reclamation (Reclamation) is the lead implementing agency for the Lower Colorado River Multi-Species Conservation Program (LCR MSCP). The LCR MSCP is a 50-year cooperative Federal-State-Tribal-County-Private endeavor that will manage the natural resources of the LCR watershed, provide regulatory relief for the use of water resources of the river, and create native habitat types along the LCR. Implementation of the LCR MSCP began in October 2005. In order to restore native habitats, the LCR MSCP will create the following cover types: (1) 5,940 acres (2,404 ha) of cottonwood-willow (*Populus fremontii*-*Salix* spp.); (2) 1,320 acres (534 ha) of honey mesquite (*Prosopis glandulosa*); (3) 512 acres (207 ha) of marsh; and (4) 360 acres (146 ha) of backwaters. A total of 26 covered and 5 evaluation species are included within the LCR MSCP Habitat Conservation Plan (HCP). The LCR MSCP Steering Committee developed, adopted, and applied criteria for selecting covered species from among 149 special status species that were considered. These criteria included those that were either listed under the Endangered Species Act (ESA; Public Law 93–205), were candidates for listing under the ESA, or were State listed by California, Nevada, or Arizona. Evaluation species were those that could not be added to the covered species list during program implementation because sufficient information was not available at the time to determine their status in the program area (Lower Colorado River Multi-Species Conservation Program, 2004).

Covered Bat Species

Four bat species are included in the LCR MSCP. The western red bat (*Lasiurus blossevillii*) and western yellow bat (*Lasiurus xanthinus*) are listed as covered species. The California leaf-nosed bat (*Macrotus californicus*) and Townsend's big-eared bat (*Corynorhinus townsendii*) are listed as evaluation species. Each species has conservation measures required by the HCP. Below is a brief account of each species.

Western Red Bat

The western red bat (fig. 1) is found primarily in riparian habitats throughout the West (Kays and Wilson, 2002). These bats prefer to roost in the foliage of large deciduous trees within riparian areas (Shump and Shump, 1982; Cryan, 2003). Western red bats are declining primarily because of the loss of habitat (Bolster, 2005). Moths are the preferred food of the western red bat, although they will also feed on beetles and other flying insects. Western red bats are thought to migrate long distances between summer and winter areas (Shump and Shump, 1982).

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Figure 1. The western red bat.

Western Yellow Bat

The western yellow bat (fig. 2) is found in riparian habitats throughout the Southwest (Kays and Wilson, 2002). These bats prefer to roost in fan palm trees (*Washingtonia* spp.) within the "skirt" of dead fronds (fig. 3), but will also roost in the foliage of deciduous trees (Cockrum, 1961; and Kurta and Lehr, 1995). Because of the introduction of ornamental palm trees, some researchers believe the range of the western yellow bat is expanding, though ornamental palms are only used if the trees have intact skirts. Like the western red bat, western yellow bats prefer to feed on moths, though they will take other prey. These bats also are thought to migrate long distances (Kurta and Lehr, 1995).

Figure 2. The western yellow bat.

Figure 3. A fan palm grove where yellow bats are known to roost north of Parker, AZ.

HCP Conservation Measures for the Western Red Bat and Western Yellow Bat

- Conduct surveys to determine the distribution of the western red bat and western yellow bat
- Create 765 acres of roosting habitat

California Leaf-Nosed Bat

The California leaf-nosed bat (fig. 4) roosts in mines and caves in southern Nevada, California, western and southern Arizona, and northwestern Mexico (Kays and Wilson, 2002). These bats forage in riparian areas and desert washes where they glean large beetles and other insects from vegetation (Brown, 2005). They are known to migrate locally to different roosts in the summer and winter, and they are active year round (Anderson, 1969; Brown, 2005).

Figure 4. The California leaf-nosed bat.

Townsend's Big-Eared Bat

The Townsend's big-eared bat (fig. 5) is found in appropriate roosts throughout the West (Kays and Wilson, 2002). Appropriate roosts include mines, caves, and buildings. These bats are known to be highly susceptible to disturbance and are known to abandon roosts. Thus, they are a species of concern throughout their range (Pearson and others, 1952; Pierson and Rainey, 1998). Townsend's big-eared bats forage in a variety of habitats and are known to prefer riparian areas when available (Pierson and others, 1999). They primarily feed on moths and spend the winter in hibernacula with very limited activity (Sample and Whitmore, 1993; Burford and Lacki, 1995).

Figure 5. Townsend's big-eared bat.

HCP Conservation Measures for the California Leaf-Nosed Bat and Townsend's Big-Eared Bat

- Conduct surveys to locate roost sites
- Create covered species habitat near roost sites

Additional Monitoring and Research Measures from the HCP

- Conduct surveys and research to better identify covered and evaluation species habitat requirements
- Monitor and adaptively manage created covered and evaluation species habitats

Bat Monitoring

The LCR MSCP has created over 500 acres of riparian habitat. Monitoring of these created habitats is essential to accomplish the measures listed above. Bat species are currently being monitored using two different methods: acoustic and capture surveys. The first method uses acoustic bat detectors that record bat echolocation calls. The Anabat™ bat detector stores these calls on a compact flash card, which can be downloaded and viewed on software (fig. 6). This software

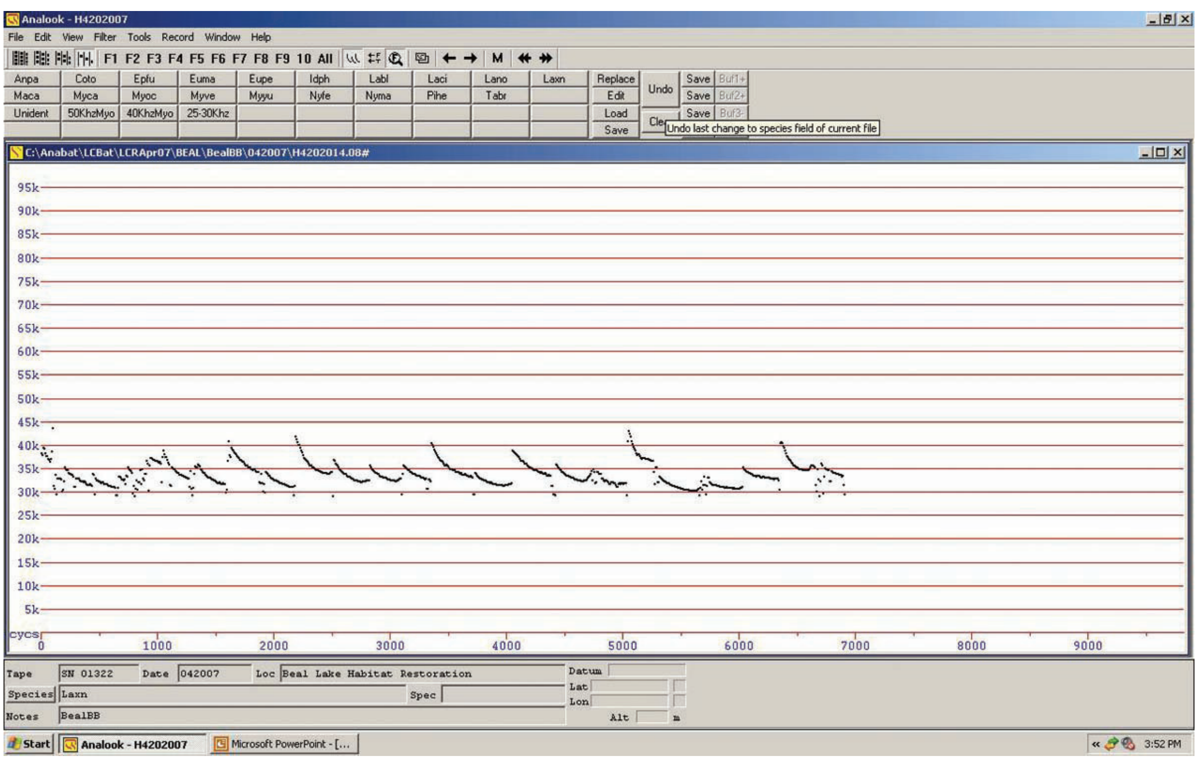

Figure 6. A screen shot of an Anabat™ call file displayed on the analyzing software.

is also used to identify species by using call parameters from known species reference calls. There are three methods for surveying bats using bat detectors. The first is known as active monitoring. This method allows a person to either walk or drive with the bat detector. Usually a small hand-held computer is attached to the bat detector for real-time observation of bat calls during the survey. The second method is known as short-term passive monitoring. This method enables one or more bat detectors to be deployed in one area over a short period of time (usually 2–3 days), after which the detectors are collected, and the data are analyzed. This process can then be repeated at regular intervals. The third method is known as long-term passive monitoring. This method involves attaching the bat detector to an external battery and solar panel, which allows data to be collected on a nightly basis as long as data need to be collected. Some researchers have had these "long-term stations" up and running for multiple years. Capture surveys are conducted using mist nets and harp traps in areas where bats are likely to be concentrated within a site.

Acoustic Methods and Preliminary Results

A pilot study began in the fall of 2006 to determine the effectiveness of short-term passive monitoring using bat detectors to monitor habitat creation areas. We placed multiple detectors across each site for 2 nights per season for a year-round picture of bat use at each site. Detectors were placed nonrandomly across each site in order to maximize the recording of activity within each site on the basis of past experience. This included the placement of the detector microphone in areas with a mosaic of habitat types that were open enough to allow bats to fly without much obstruction. We analyzed all files that contained bat calls using Analook™ software. Calls were identified to species unless the calls were too similar to other species. These calls were collected into species groups by the frequency ranges at which that group of species echolocates. We determined that using this method sufficiently characterized the general bat community at each site (Bureau of Reclamation, 2008). The actual number of bats could not be determined using acoustic monitoring. Instead, an index of relative bat activity for each species was created to determine how much each species utilizes each area being surveyed. This index was created by using the number of minutes each species is detected within any given hour so that each species will have no more than 60 "bat minutes" in an hour (Miller, 2001). This eliminated the bias of having multiple calls within a single minute, which may overestimate the activity of that species over the entire night. The proportion of bat minutes for each species was then calculated from the total number of minutes for all species. Table 1 gives the results from 1 year of data for all sites combined in the pilot study. A table of common and scientific names for all bat species identified is given in table 2. Species groups were used for multiple species with similar or overlapping call characteristics.

After the pilot study proceeded for two more seasons, we decided that acoustic monitoring could offer more information than just a general characterization of the bat community. Other researchers have used bat detectors to determine habitat preferences of bats (Menzel and others, 2005; Loeb and O'Keefe, 2006; Ober and Hayes, 2008). In March of 2008, we modified the pilot study protocol to allow for a more statistically robust study design. Our goals were to continue characterizing the bat community while also identifying relations between habitat type and bat activity within the habitat

Table 1. Acoustic monitoring results for all sites from fall 2006 to summer 2007. Lower Colorado River Multi-Species Conservation Program species in bold. Data are from 191 detector nights of six sites.

[kHz, kilohertz]

Common name/group [a]	Total bat minutes	Relative bat activity
45–55 kHz	13,243	44.07%
Canyon bat	7,340	24.43%
25–30 kHz	7,196	23.95%
Cave myotis	618	2.06%
35 kHz	375	1.25%
California leaf-nosed bat	353	1.17%
Greater mastiff bat	322	1.07%
Pocketed free-tailed bat	316	1.05%
Hoary bat	113	0.38%
Western yellow bat	83	0.28%
Western red bat	37	0.12%
Big free-tailed bat	37	0.12%
20–25 kHz	9	0.03%
Townsend's big-eared bat	6	0.02%
Silver-haired bat	2	0.01%
Total	30,050	100.00%

[a] Species included in species groups:

45–55 kHz group: Yuma myotis, California myotis, canyon bat
25–30 kHz group: big brown bat, Brazilian free-tail, pallid bat
35 kHz group: pallid bat, cave myotis
20–25 kHz group: pocketed free-tail, big free-tail, hoary bat, Brazilian free-tail

Table 2. Common and scientific names for bat species identified in the study.

Common name	Scientific name
California leaf-nosed bat	*Macrotus californicus*
Brazilian free-tailed bat	*Tadarida brasiliensis*
Pocketed free-tailed bat	*Nyctinomops femorosaccus*
Big free-tailed bat	*Nyctinomops macrotis*
Western mastiff bat	*Eumops perotis*
Western red bat	*Lasiurus blossevillii*
Western yellow bat	*Lasiurus xanthinus*
Hoary bat	*Lasiurus cinereus*
Silver-haired bat	*Lasionycteris noctivagans*
Townsend's big-eared bat	*Corynorhinus townsendii*
Pallid bat	*Antrozous pallidus*
California myotis	*Myotis californicus*
Cave myotis	*Myotis velifer*
Yuma myotis	*Myotis yumanensis*
Canyon bat	*Parastrellus hesperus*
Big brown bat	*Eptesicus fuscus*

creation areas. If habitat preferences of covered bat species can be discovered, it will help resolve critical management uncertainties.

The new study design, which began with the spring sampling of 2008, is scalable, providing information within individual sites as well as giving us the ability to look at the larger LCR system. Our primary focus will be on habitat use by the four covered species. We will compare bat activity levels between different habitat types as well as how these levels change through time as the habitat matures at each site. Landscape features, such as distance to pooled water, distance to roosts, tree canopy height, and tree density, will also be examined.

We chose five habitat types for monitoring as part of the new study design. At least three of the five habitat types will be monitored per study area. Three bat detectors will be deployed within each habitat type so that at least nine detectors are being deployed per study area. Detector locations will be chosen nonrandomly in areas of the habitat where bats are most likely to be flying. Surveys will be conducted for 2 days every season at each study area. Five study areas were chosen for the study. If two sites were within close proximity to one another, they were combined into a single study area. These areas occur within a 196-mile stretch of the river (fig. 7).

Study Areas and Habitat Types

The seven sites are separated into five study areas as follows:

1. Beal Lake Riparian Restoration (Beal)
2. ‘Ahakhav Tribal Preserve (‘Ahakhav)
3. Palo Verde Ecological Reserve (PVER)
4. a. Cibola Valley Conservation and Wildlife Area
   b. Cibola NWR Unit 1 Conservation Area (Cibola)
5. a. Imperial Ponds Conservation Area
   b. Pratt Restoration Demonstration Area (Imperial/Pratt)

The five habitat types being monitored are as follows (figs. 8–12):

- Sapling cottonwood-willow plantings (average diameter at breast height (DBH) is <8 centimeters)
- Intermediate cottonwood-willow plantings (average DBH ≥8 centimeters)
- Mesquite plantings (average canopy height ≥3 meters)
- Agricultural fields
- Monotypic *Tamarisk* spp. stands

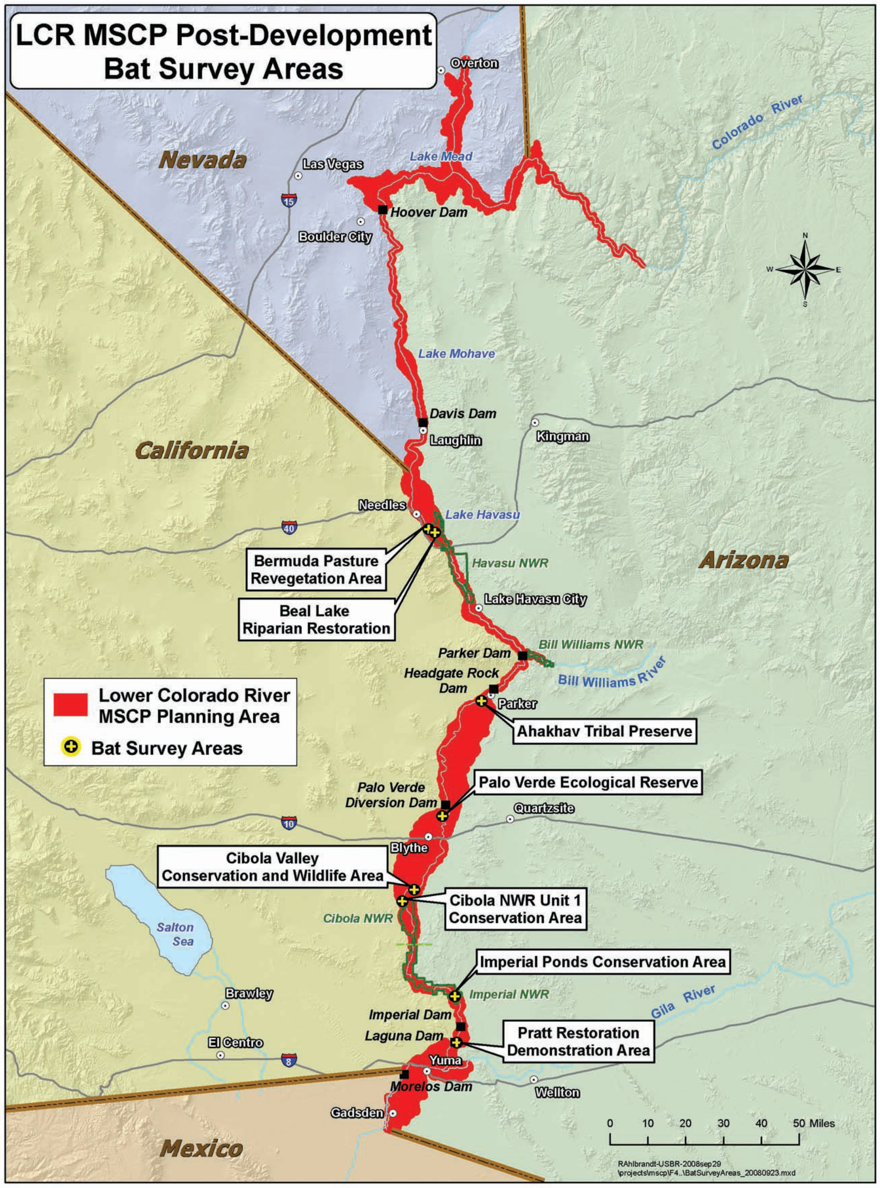

Figure 7. Bat monitoring locations at habitat creation areas along the lower Colorago River. Note that the survey area *Bermuda Pasture Revegetation Area* was not included in acoustic surveys.

Figure 8. Sapling cottonwood willow.

Figure 9. Intermediate cottonwood willow.

Figure 10. Mesquite.

Figure 11. Agricultural field.

Figure 12. Monotypic *Tamarisk* spp. stand.

We separated cottonwood-willow plantings into two classes because of the differences in size and structure that may cause bats to use these two classes differently. For example, the sapling trees will most likely not be used for roosting. Agricultural fields and *Tamarisk* spp. (saltcedar) stands were chosen because they serve as controls of what habitat is predominant along the LCR as well as what habitat is being replaced within habitat creation areas. Table 3 lists which habitat types will be monitored in each study area.

This new study design will continue for an additional 1–2 years. By the end of the project we anticipate that indices of activity will be developed for each habitat type for most bat species. We will also determine the overall bat species assemblage for each habitat creation area. Currently, we are testing the use of a long-term acoustic bat station at the Beal site within Havasu National Wildlife Refuge (NWR). The station has been in operation since April 2008. It consists of a weather proof box that contains the bat detector, battery, and a weather data logger attached to a post that has been cemented into the ground. Also attached to this post is a solar panel that recharges the battery and an anemometer that collects wind data for the weather data logger. The microphone is detached from the detector with a cable that runs up to the top of the pole where it is housed in weatherproof housing (fig. 13). A 1-gigabyte flashcard, which is inserted into the detector and is downloaded every 3–5 weeks. Although data at a long-term station are only being collected at one sampling location within a site, the data are being collected every night. This allows for variation to be seen at multiple scales (nightly, seasonally, and annually). An example of the high variation in bat activity can be seen in figure 14. A long-term station also increases the chances of recording uncommon species that may not be in the area every night. In the future, a system of long-term stations will be established at all habitat creation areas.

Table 3. Study area locations for each habitat type being monitored.

[CW, cottonwood willow]

Study area	CW - saplings	CW - intermediate	Mesquite	*Tamarisk* spp.	Agriculture
Beal	X		X	X	
‘Ahakhav	X	X	X		
PVER	X			X	X
Cibola	X	X	X		X
Imperial/Pratt		X		X	X

Figure 13. Long-term Anabat™ station located in the Beal Restoration site at Havasu National Wildlife Refuge.

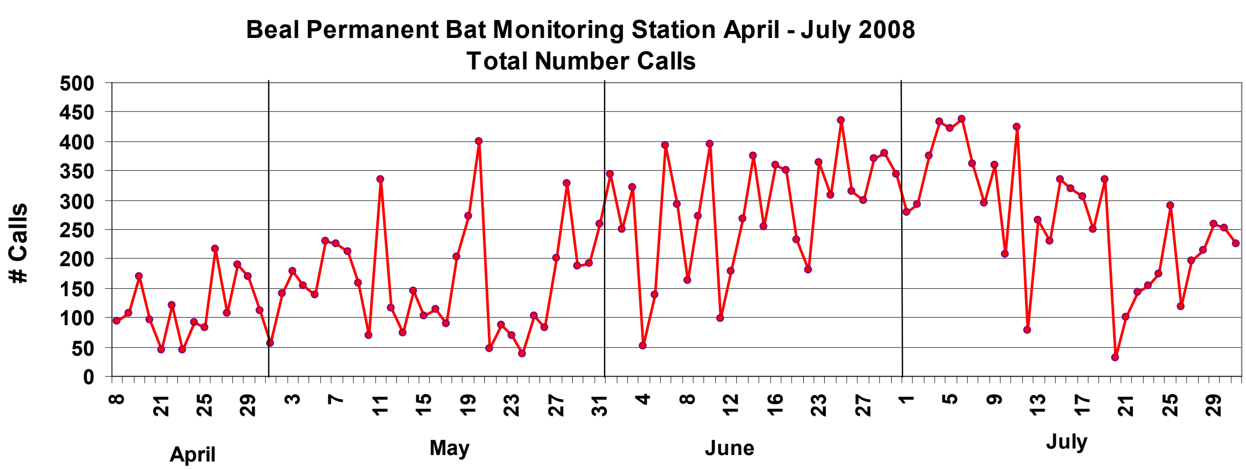

Figure 14. Variation in total bat activity at the Beal Restoration site from April to July 2008.

Capture Methods and Preliminary Results

We initiated a bat capture program in the summer of 2007 to determine the feasibility of capturing bats within habitat creation areas. The capture surveys had three main objectives:

1. Capture covered species and collect reference acoustic calls.
2. Collect information such as age, sex, and reproductive status.
3. Aid the design of future habitat areas by comparing capture success with capture locations.

Because bat echolocation calls can be quite variable, obtaining as many reference calls as possible from each species ensures proper identification. Acoustic monitoring is limited in what type of information can be gathered. Capture surveys allow for information, such as sex, age, and reproductive status, to be collected. Bat capture surveys were conducted using mist nets and harp traps. Bats generally avoid cluttered habitat and use open areas and corridors for flyways (Manley and others, 2006). One challenge of netting within these areas is the ability of the bats to avoid a single net (2.6 meters high) placed across a corridor. Most researchers net over water where bats are determined to reach the water source; however, because our surveys are being conducted within our habitat creation areas, this method is not possible for our study. One way of overcoming this situation is to stack nets on top of each other to reach higher into the canopy. Many of the methods used were learned in a bat conservation and management workshop provided by Bat Conservation International (2007). Generally, the poles used to attach the nets have a pulley system that allows the different nets to be raised and lowered to the appropriate height where bats can be removed from the net (fig. 15). Harp traps were used when the vegetation narrowed to a point where bats were funneled through a smaller area (fig. 16). By setting nets and traps in these types of settings within habitat creation areas, adequate capture rates were possible. Bats were handled with leather gloves by personnel who have had rabies pre-exposure vaccinations. All bats were handled by approved animal care guidelines (Gannon and Sikes, 2007).

Capture Results

Surveys were conducted in April, July, September, and October 2007 and in April, May, July, August, and September 2008. Five sites were surveyed during the 2 years. We captured a total of 263 bats of nine species, including two LCR MSCP species (table 4). Our capture rates were highest when stacked nets were used in defined corridors, which existed because of the original design of the site. Sites that had poorly defined corridors had lower capture rates because of the ability of the bats to avoid nets more easily. In future years, our effort will be focused on sites with higher capture rates, and additional sites may be chosen as new habitat creation areas mature.

Figure 15. A triple high mist-net set up reaching over 8 meters high at the Beal Restoration site.

Figure 16. A harp trap set within a narrow opening at Cibola National Wildlife Refuge.

Table 4. Bat capture results for 2007 and 2008.

[Lower Colorado River Multi-Species Conservation Program species in bold. N = total number of survey nights]

Species	Beal N = 5	Bermuda Pasture N = 4	‘Ahakhav N = 7	Unit 1 N = 7	Pratt N = 8	Total
Pallid bat	3	13	39	14	25	94
Big-brown bat	0	1	9	15	41	66
Yuma myotis	12	3	16	1	4	36
California leaf-nosed bat	0	0	5	18	10	33
Western yellow bat	0	0	8	2	1	11
Cave myotis	1	2	6	0	0	9
California myotis	0	0	2	3	0	5
Myotis spp.	0	0	5	0	0	5
Hoary bat	0	0	0	3	0	3
Brazilian free-tailed bat	0	0	1	0	0	1
Total	16	19	91	56	81	263

Management Considerations

Using both acoustic and capture survey methods gives the best overall picture of bat use in an area. These survey methods will accomplish the monitoring goals set forth in the HCP. Understanding how bats use these sites will aid the design of future habitat creation areas. Adaptive management is only possible when enough information is gathered to make recommendations. One example is how bat captures were highest where there were defined corridors. This information may be used in the future to include “bat corridors” into the design of habitat creation areas to allow bats additional areas to forage as well as to aid monitoring efforts. These survey methods are adaptable so that they may be used for similar resource management projects. Monitoring a variety of wildlife, including bats, allows for a better understanding of how different species are affected by different measures that may take place within an area.

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Little Colorado River Lower 1,200-Meter Long-Term Fish Monitoring, 1987–2008

By Brian C. Clark,[1] William R. Persons,[2,3] and David L. Ward[4]

Abstract

The Arizona Game and Fish Department has been monitoring the status of the endangered humpback chub (*Gila cypha*) and other fishes in the lower Little Colorado River in Grand Canyon since 1987. Thirteen hoop nets are set in standardized locations and checked daily for 20–30 days each spring. This monitoring program is one of the most consistent long-term sampling efforts for fish in Grand Canyon. The catch rate of humpback chub, as well as other fishes, is an important tool to estimate the number of individual fish within the populations. Recent increases in catch rates of native species, such as flannelmouth (*Catostomus latipinnis*) and bluehead (*Catostomus discobolus*) suckers, indicate that populations of these species are increasing.

Introduction

In 1987, the Arizona Game and Fish Department (AGFD) began monitoring of fishes in the lower 1,200 meters (m) of the Little Colorado River (LCR) to assess population trends and status of the endangered humpback chub (HBC; *Gila cypha*). In 2000, the AGFD lower 1,200-m monitoring project was discontinued and reinstated beginning in 2002. The confluence of the LCR and Colorado River is approximately 61 river miles[5] downstream from the boat launch ramp at Lees Ferry, within Grand Canyon National Park. The LCR is one of the primary tributaries to the Colorado River. It is a primary spawning site for the HBC and is an important spawning location for other native species. The LCR is the only known HBC aggregate in the Colorado River ecosystem within Grand Canyon from which fish are known to recruit into the adult population (Valdez and Ryel 1995; Coggins and others, 2006). Other native fishes, bluehead sucker (*Catostomus discobolus)*, flannelmouth sucker (*Catostomus latipinnis*), and speckled dace (*Rhinichthys osculus*) spawn in the LCR (Robinson and others, 1998) as do nonnative species, including channel catfish (*Ictalurus punctatus*), fathead minnow (*Pimephales promelas*), red shiner (*Cyprinella lutrensis*), common carp (*Cyprinus carpio*), black bullhead (*Ameiurus melas*), and plains killifish (*Fundulus zebrinus*). The lower 1,200-m monitoring site of the LCR is a deeply entrenched channel located in a vertical-walled canyon that, in places, narrows to less than 50 m in width (fig. 1). The LCR channel contains runs, riffles, deep pools, and small rapids. Substrates are primarily silt and sand with scattered large boulders and travertine dams.

In order to compare data over several years, it is important to plan monitoring events in a consistent manner. The standardization of the LCR fish monitoring project included deploying the same size and style of hoop nets as well as similar placement of the nets within the LCR on each sampling occasion. Hoop nets are considered to be a passive capture technique that entraps fish without the nets being actively moved by humans. Fish swimming upstream freely swim into the nets and become trapped and cannot escape. The hoop nets deployed in the LCR are a cylindrical net 5 m in length and 1 m in diameter, distended by a series of

Figure 1. Lower 1,200 meters of Little Colorado River.

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[5] By convention, river mile is used to describe distance along the Colorado River.

seven metal hoops covered by 6.3 millimeter (mm) mesh web netting. Also, the lower 1,200-m project conducts sampling every year in the spring in an attempt to capture native fish that return to the LCR in the spring to spawn. Because of the spatial and temporal nature of the lower 1,200-m monitoring, it is possible that some fish migrate upstream from the lower 1,200-m reach from the mainstem Colorado River before AGFD personnel arrive in the spring and deploy hoop nets and, therefore, are not susceptible to capture. The hoop nets are effective at capturing adult and juvenile fish, and the use of hoop nets minimizes physical harm and stress to the fish.

Methods

The Arizona Game and Fish Department has been monitoring the status and trends of the endangered humpback chub and other fishes in the lower Little Colorado River in Grand Canyon since 1987. Thirteen hoop nets are set in standardized locations and checked daily for 20–30 days each spring. Nets are set at 100, 119, 137, 165, 420, 480, 500, 577, 675, 1,045, 1,110, 1,160, and 1,195 m upstream from the confluence. Nets are set as close as possible to those used in previous sampling efforts (Brouder and Hoffnagle, 1998). All fish captured are handled following protocols in Ward (2002). Physical property data are collected for turbidity (nephelometric turbidity units) and temperature (degrees Celsius (°C)) during the monitoring period by AGFD personnel using a Hach 2100P turbidimeter and a hand-held Cooper Model DPP400W thermometer every morning before checking hoop nets. Flow data are collected from the U.S. Geological Survey (USGS) real-time water data station USGS 09402300, which is located within the 1,200-m reach of the LCR.

Long-Term Trends

Since the beginning of the AGFD lower 1,200-m project in 1987, considerable numbers of native and nonnative species have been captured (table 1). The species composition of fish captured since 1987 has been dominated by native species (>80 percent), in general, with the exception of 1997 and 2006 when fathead minnows dominated the total catch. Catch rates of native species vary from year to year; however, within the last 2 years, flannelmouth sucker (FMS) and bluehead sucker (BHS) mean catch per hour has increased to levels greater than previous years (fig. 2). In 2008, the total catch of BHS and the mean catch rate (fish per hour) was the highest recorded since

Table 1. Total catch of species by year, Little Colorado River standardized hoop-net monitoring.

[BBH, Black bullhead; BHS, Bluehead sucker; CCF, Channel catfish; CRP, Common carp; FHM, Fathead minnow; FMS, Flannelmouth sucker; GSF, Green sunfish; HBC, Humpback chub; PKF, Plains killifish; RBT, Rainbow trout; RSH, Red shiner; SPD, Speckled dace; SUC, unidentified sucker]

Species	BBH	BHS	CCF	CRP	FHM	FMS	GSF	HBC	PKF	RBT	RSH	SPD	SUC
1987	0	39	5	2	1	81	1	396	0	0	0	132	0
1988	0	65	8	1	12	91	0	596	0	0	0	192	0
1989	0	72	41	0	17	28	0	548	0	1	2	204	0
1990	0	25	2	0	10	30	0	418	0	0	0	90	3
1991	0	106	4	0	3	106	0	316	0	1	0	1,003	0
1992	0	19	8	0	1	25	0	199	0	0	0	110	0
1993	0	44	0	0	1	50	0	431	0	2	0	455	1
1994	0	64	5	0	265	88	0	657	0	0	0	1,022	0
1995	1	32	1	1	19	65	0	243	0	1	0	488	0
1996	0	413	1	8	237	237	0	359	0	8	14	741	2
1997	1	45	12	60	726	97	0	123	97	1	74	417	0
1998	1	27	5	0	52	6	0	132	1	4	8	106	0
1999	0	61	10	5	14	21	0	156	0	6	70	187	0
2002	0	122	1	0	46	79	0	130	1	3	3	115	0
2003	3	93	3	7	42	256	0	157	0	0	13	116	0
2004	5	154	7	7	91	357	0	743	52	5	65	1,918	0
2005	4	347	3	1	0	192	0	344	0	1	0	445	0
2006	12	395	13	19	1,286	483	0	587	9	1	44	3,173	0
2007	9	304	3	13	17	644	0	266	12	0	8	1,644	0
2008	19	568	3	1	62	596	0	507	0	0	0	1,288	0

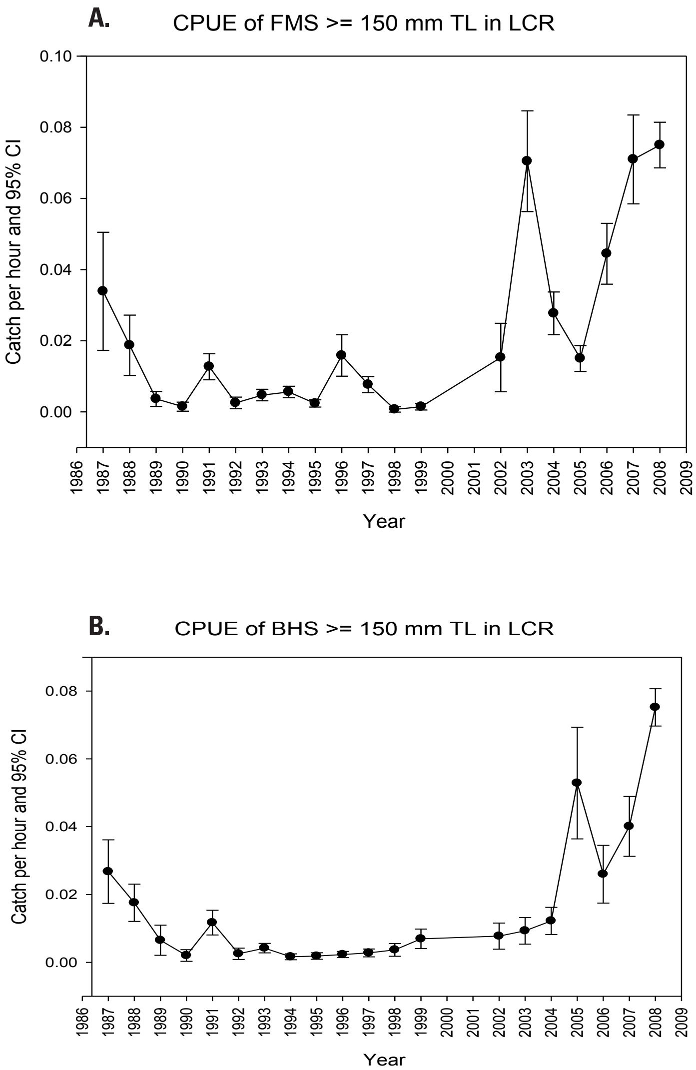

Figure 2. Mean catch per hour (CPUE) of (A) flannelmouth sucker (FMS) ≥150 millimeters (mm) total length (TL) and (B) bluehead sucker (BHS) ≥150 mm TL in the Little Colorado River (LCR) during Arizona Game and Fish Department lower 1,200-meter monitoring, 1987–2008.

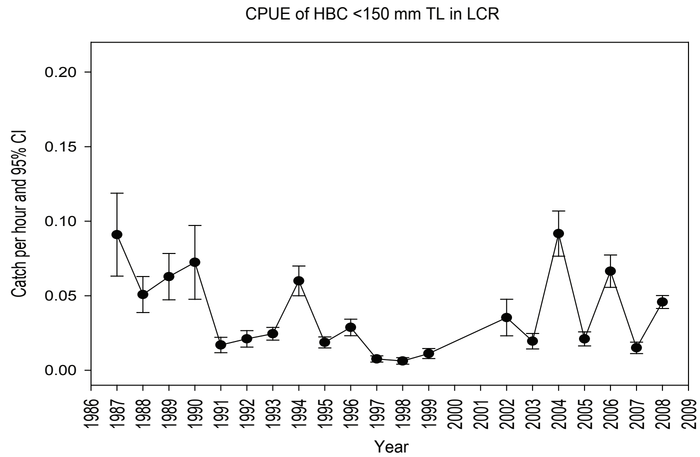

Figure 3. Mean catch per hour (CPUE) of humpback chub (HBC) <150 millimeters (mm) total length (TL) in the Little Colorado River (LCR) during Arizona Game and Fish Department lower 1,200-meter monitoring, 1987–2008.

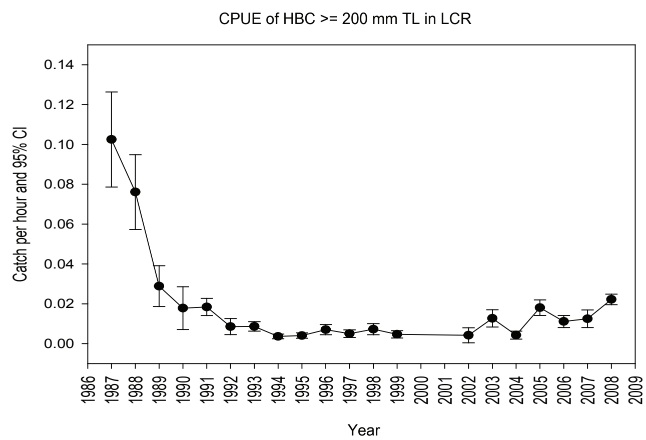

Figure 4. Mean catch per hour (CPUE) of humpback chub (HBC) ≥200 millimeters (mm) total length (TL) in the Little Colorado River (LCR) during Arizona Game and Fish Department lower 1,200-meter monitoring, 1987–2008.

AGFD monitoring began in 1987. In 2007, the total catch of FMS was the highest recorded during AGFD lower 1,200-m monitoring, and in 2008 the mean catch per hour was the highest recorded since AGFD lower 1,200-m monitoring began in 1987. The catch per hour of juvenile HBC (<150 mm) total length (TL) varies from year to year (fig. 3). Since 2006, the mean catch per hour of adult HBC (≥200 mm TL) appears to have stabilized at levels similar to the early 1990s (fig. 4). Small-bodied nonnative species catch rates vary from year to year possibly because of flooding events from LCR high-flow events, which displace those species into the mainstem Colorado River. Typically, once small-bodied, introduced species such as fathead minnow or red shiner appear, those species, which are adapted for more stable systems, gradually increase in abundance over time until they numerically dominate a fish assemblage (Marsh and Pacey, 2005). Several factors may prevent these species from becoming well established, such as the extreme flood regime, high turbidity, and high salinity of the LCR during spring and late summer (Minckley and Meffe 1987; Ward and others, 2003). The catch per hour of common carp also varies from year to year. Adult carp are not captured frequently in hoop nets, although smaller juvenile carp are captured more commonly. Therefore, catch rates of common carp are not a good index of the LCR carp population. Catch per hour of channel catfish are generally low, and most often the channel catfish captured are juvenile or sub-adults. Black bullhead mean catch per hour has increased over the last 3 years.

The increases in catch rates of suckers may be attributed to warmer mainstem water temperatures caused by lower water levels in Lake Powell. When Lake Powell was at full pool, the water released from the Glen Canyon Dam through the penstocks was cold (<11 °C) throughout the year. Because of recent drought conditions resulting in lower lake levels, the water being released from Glen Canyon Dam has been warmer than average (1990–2002) during the summer and fall (10–16 °C) (Grand Canyon Monitoring and Research Center, 2003). Another factor that may have been beneficial to sucker populations was an experimental nonnative mechanical removal project on the Colorado River near the confluence of the LCR. The removal project started in 2003 and ended in 2006; the removal project targeted nonnative species approximately 5 miles above and below the LCR confluence. The removal project was successful in reducing the number of rainbow trout (*Oncorhynchus mykiss*) in the vicinity of the LCR confluence. Larger adult rainbow trout are capable of preying upon smaller fishes such as juvenile HBC (Paukert and Petersen, 2007). In addition to predation upon juvenile native fishes, nonnative species compete for food resources with the native species (Paukert and Petersen, 2007).

Management Implications

The lower 1,200-m hoop-net monitoring represents one of the longest ongoing trend indices for Grand Canyon fishes. The real strength of this dataset is the long length of time over which the data have been collected in a consistent manner. The lower 1,200-m monitoring project allows researchers to track trends in relative abundance and catch rates of native and nonnative fishes, as well as potential early detection of rare nonnative species that may enter the Colorado River ecosystem by way of the LCR (fig. 5). The trend indices of multiple size classes of native fishes are useful in aiding researchers in following recruitment of juvenile and sub-adult fishes into the adult population. Catch per hour indices derived from the lower 1,200-m monitoring is a valuable method to confirm output of age-structured mark-recapture open population models.

Figure 5. Little Colorado River.

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The Humpback Chub of Grand Canyon

By David R. Van Haverbeke[1]

Abstract

Anyone gazing into Grand Canyon invariably wonders… "*what's down there*?" Among the Canyon's myriad secrets, one is the endangered humpback chub (*Gila cypha*). Many hikers and rafters venture into the depths of Grand Canyon each year, but few glimpse this rare and fascinating animal. Even so, this fish represents a core natural value of Grand Canyon.

The U.S. Fish and Wildlife Service (USFWS) conducts research on humpback chub in the Little Colorado River in Grand Canyon. Scientists documented a substantial decline of humpback chub during the 1990s, but recent efforts show them making a comeback. In the past 2 years, the numbers of spawning adults and year-round residents in the Little Colorado River have significantly increased.

The USFWS also conducts a project involving translocation. Since 2003, juvenile humpback chub have been moved from lower reaches of the Little Colorado River to previously unoccupied habitat higher in the watershed. Some of the fish have remained where relocated, displayed high growth rates, and may be partially contributing to the overall increase in population size of humpback chub. This project is unique in that it represents a natural rearing situation, without hatchery-reared fish.

Figure 1. Humpback chub captured and released in Little Colorado River in early 1990s. Photograph by David Van Haverbeke.

Introduction

The humpback chub was described by Miller (1946) from a specimen taken near the mouth of Bright Angel Creek in Grand Canyon National Park. Humpback chub have a unique body shape (fig. 1) and are located only in the Colorado River Basin (Minckley, 1991). Their origins extend as far back as Miocene, or more than 5 million years ago (Minckley and others, 1986). The species is a member of a relict native fish community, many of which are locally extinct or declining. Three of eight native fish species have become extinct in Grand Canyon since the closure of Glen Canyon Dam in 1963, including the Colorado pikeminnow (*Ptychocheilus lucius*), bonytail (*Gila elegans*), and roundtail chub *(Gila robusta*). A fourth, the razorback sucker (*Xyrauchen texanus*), may also be extirpated in Grand Canyon (Minckley, 1991). Humpback chub are also found in the upper Colorado River Basin, including Black Rocks, Westwater, and Cataract Canyons (upper Colorado River); Desolation/Gray Canyon (Green River); and Yampa Canyon (Yampa River; U.S. Fish and Wildlife Service, 2002). The species was listed as endangered in 1967 (Federal Register, v. 32, no. 48, p. 4001).

In Grand Canyon, humpback chub occupy unusual habitat relative to other populations in the watershed. They largely inhabit the Little Colorado River (fig. 2), a saline tributary to the Colorado River. Most humpback chub spawn and rear in the Little Colorado River. As they approach adulthood, many

Figure 2. Little Colorado River, April 2007. Photograph by Brian Healey.

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leave the Little Colorado River to inhabit the larger Colorado River. Adults return to spawn in the Little Colorado River during the spring season (Douglas and Marsh, 1996).

Spawning and Over-Wintering Abundances of Humpback Chub in the Little Colorado River

Background

In order to successfully track the abundance of a population, scientists generally employ mark-recapture techniques. In the Little Colorado River, the technique we use is called a closed population model (Seber, 2002). In very general terms, this technique involves capturing a portion of the animals in the population and "marking" them with individually numbered tags. Passive Integrated Transponder (PIT) tags, which are very small glass encapsulated microchips, are inserted into the body cavity. Once a portion of the population is marked, the animals are released and allowed to mix with the population at large. After mixing, biologists capture a portion of the population again, some of which will already be marked. Using these numbers, a population estimate is generated.

Mark-recapture efforts to determine the abundance of humpback chub in the Little Colorado River began in the 1980s (Kaeding and Zimmerman, 1983; Minckley, 1988, 1989) and were refined in the early 1990s (Douglas and Marsh, 1996). Efforts to reliably determine the population size of the species are necessary to understand the status of the species and to provide information to meet recovery criteria for the species (U.S. Fish and Wildlife Service, 2002). In 2000, in cooperation with the U.S. Geological Survey's (USGS) Grand Canyon and Monitoring Research Center (GCMRC), the USFWS reinitiated the focus on determining the population size of humpback chub in the Little Colorado River as a research objective. These efforts have provided annual estimates of the number of humpback chub ≥150 millimeters (mm) total length, as well as the number of adult humpback chub ≥200 mm that are spawning in the Little Colorado River each spring. These efforts also provide an estimate of the number of humpback chub that are presumably year-round residents in the Little Colorado River. Finally, these data are used to help generate an age-structured mark-recapture model, inclusive of not only humpback chub in the Little Colorado River, but also in the Little Colorado River inflow region of the mainstem Colorado River (Coggins and others, 2006; Coggins and Walters, 2009).

Methods

We use the Chapman modified Petersen two-sample mark-recapture model (Seber, 2002; eq 1). Depending on several factors (e.g., the proportion of the population originally marked, the number of marked fish that were recaptured), biologists place confidence intervals on their estimate of abundance (eq 2), which quantifies the degree of certainty of the estimate.

$$N^* = \frac{(M+1)(C+1)}{R+1} \quad (1)$$

$$V\left[N^*\right] = \frac{(M+1)(C+1)(M-R)(C-R)}{(R+1)^2(R+2)} \quad (2)$$

where:

N^* is the estimated number of fish in the population,

$V\left[N^*\right]$ is the estimated variance of the number of fish in the population,

M is the number of fish marked during the marking event,

C is the number of fish captured during the recapture event, and

R is the number of fish recaptured from the marked population during the recapture event.

Because we are also interested in the abundance of a particular size class, we make use of what is commonly known as the "proportion method," which calculates the *proportion* of humpback chub that are ≥200 mm out of the total abundance of humpback chub ≥150 mm. Making use of this proportional method incorporates a larger and more robust set of data (Seber, 2002). Equation 3 is used to calculate the estimate for a particular size class of fish, and equation 4 calculates the variance.

$$N_x^* = \frac{M_x + C_x - R_x}{M + C - R} N^* = P_x(N^*) \quad (3)$$

$$V\left[N_x^*\right] = N_x^{*2}\left[\frac{1}{R} + \frac{2}{R^2} + \frac{6}{R^3}\right] + \frac{N_x^*\left(N^* - N_x^*\right)}{(M+C+1)} \quad (4)$$

where P_x indicates the proportion of fish within a particular size class, and the subscript x indicates fish that belong to a particular size class (e.g., ≥200 mm).

To estimate the abundance of spawning humpback chub in the Little Colorado River each year, USFWS conducts two trips each spring. These trips are timed to coincide with the peak of the spring spawning activity and occur in April and May. To track the abundance of humpback chub presumably residing year round in the Little Colorado River, USFWS conducts two more trips during the fall each year after most migrating spawners are believed to have vacated the Little

Colorado River (Gorman and Stone, 1999). These trips occur during September and October. Each of the four trips is approximately 10 days. We allow 2 weeks to pass between any given "marking" and "recapture" trip. This helps to ensure that marked fish mix into the population in between the two trips and helps to reduce the chance for movement of fish in and out of the Little Colorado River.

Because we sample a 13.57-kilometer (km) stretch of river, three camps are established during each trip. These camps are referred to as the Boulders, Coyote, and Salt Camps and are located respectively 1.9, 9.0, and 10.4 km upriver from the confluence with the Colorado River. The Bureau of Reclamation (USBR) provides helicopter support to fly personnel and gear to each campsite. Once in the canyon, each camp is supervised by a USFWS biologist and includes two volunteers. Each camp is responsible for fishing about a 4.5-km stretch of river (i.e., Boulders 0 to 5 km, Coyote 5 to 9.6 km, and Salt 9.6 to 13.57 km; fig. 3). Daily afternoon water temperature data are collected near Salt Camp, and turbidity is measured with a Hach 2100P turbidimeter. Provisional streamflow data (maximum and mean daily discharge in cubic feet per second) are downloaded (*http://nwis.waterdata.usgs.gov*) from USGS streamgaging station 0940200 located upriver on the Little Colorado River near Cameron, AZ.

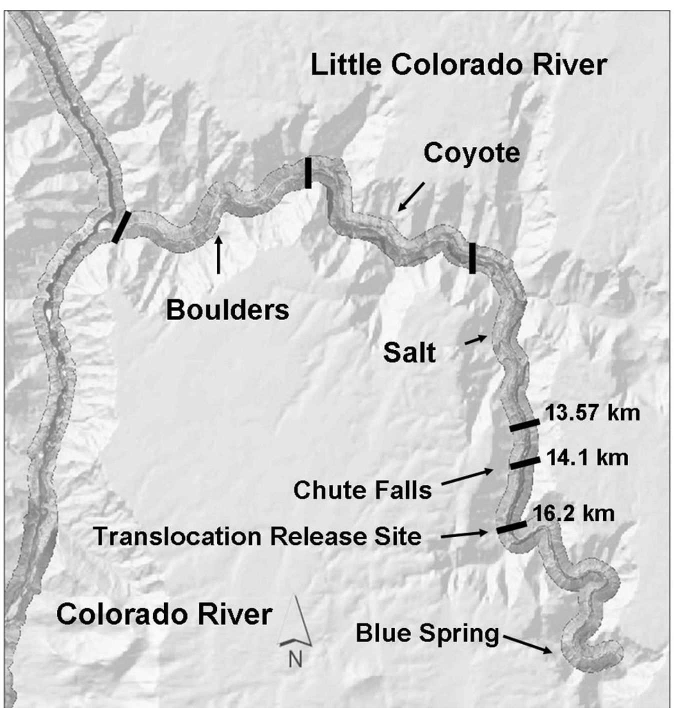

Figure 3. Study sites in Little Colorado River showing: (1) Salt, Coyote, and Boulders reaches (study areas of spring and fall mark-recapture efforts), (2) release site of translocated humpback chub at 16.2 km, and (3) the two reaches (lower and upper) of the Chute Falls mark-recapture efforts. (Note: Lower reach extends from 13.57 to 14.1 km, and upper reach extends from 14.1 to 18 km.)

Hoop nets are used to capture fish (fig. 4). Hoop nets are effective at capturing humpback chub and are a relatively benign sampling method. The mesh nets are barrel shaped with a funnel-shaped entrance that tends to direct fish into the net and prevent their escape. The dimensions of the hoop nets are 0.5–0.6 meter (m) diameter, 1.0 m length, 6 mm mesh, with a single 0.1 m throat, and three steel hoops (Memphis Net and Twine, Inc.). Hoop nets are set for approximately 24 hours each and are fished along shorelines, cut banks, and behind boulders, in areas suspected of yielding high catches of humpback chub.

Fish are removed from the nets daily, identified to species, measured for length (in millimeters), and checked for other characteristics (e.g., sexual condition, external parasites). All large-bodied native fish (humpback chub, bluehead sucker [*Catostomus discobolus*], and flannelmouth sucker [*Catostomus latipinnis*]) ≥150 mm are implanted with a TX1411SST, 134.2 kHz PIT tag (Biomark, Inc.) and released. More specifics on methods can be found in Van Haverbeke and Stone (2009).

Figure 4. Setting a hoop net in the Little Colorado River. Photograph by Michael J. Pillow.

Results

During spring trips from 2001 to 2008, we deployed 9,080 hoop-net sets in the lower 13.57 km of the Little Colorado River, which yielded 211,527 hours of fishing effort. We captured 53,308 fish, of which 25,442 (46 percent) were humpback chub. Native fish made up 89 percent of the overall spring catches, while nonnative fishes made up the remaining 11 percent. Nearly 4,400 humpback chub ≥150 mm received PIT tags. From 2001 to 2006, the spring abundance estimates for humpback chub ≥150 mm ranged between 2,082 and 3,419 (fig. 5). For 2007 to 2008, the spring abundance estimates for humpback chub ≥150 mm increased to 5,124 and 5,850, respectively (fig. 5). For adult humpback chub (≥200 mm) from 2001 to 2006, the spring abundance estimates ranged

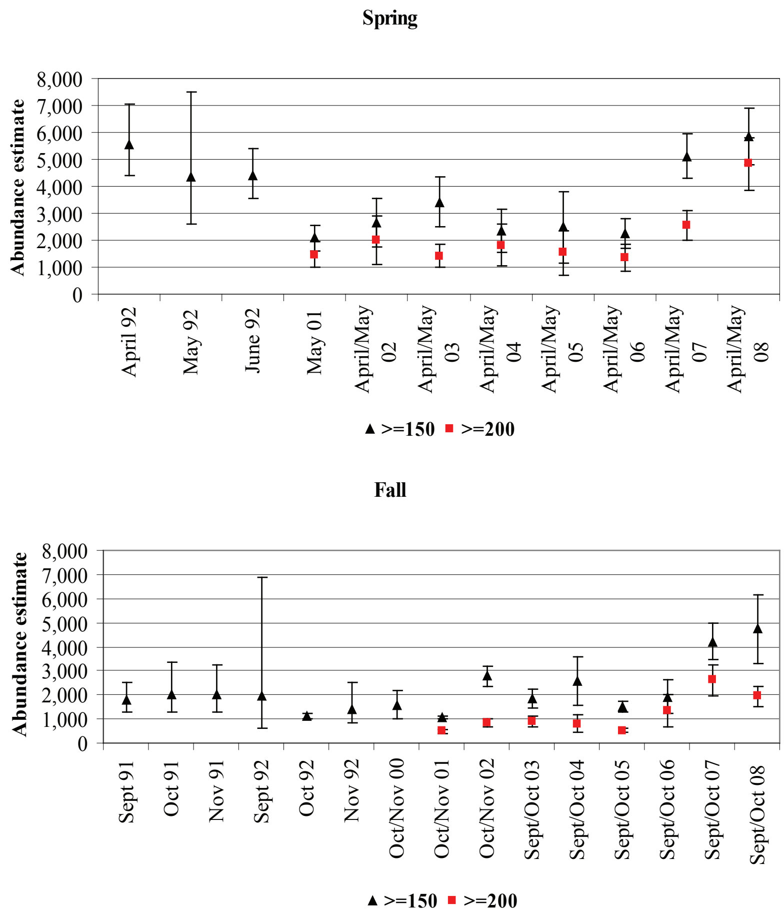

Figure 5. Spring and fall abundance estimates (with 95 percent confidence level intervals) of humpback chub ≥150 mm, and ≥200 mm in the lower 13.57 km of the Little Colorado River. All pre-2000 estimates are from Douglas and Marsh (1996). All other estimates from Van Haverbeke and Stone (2009).

between 1,339 and 2,002. In 2007 and 2008, the abundances rose to 2,544 and 4,831, respectively (fig. 5). In addition, we witnessed the abundance estimates of bluehead sucker increase from 12,295 in 2006 to 74,655 in 2008 (Van Haverbeke and Stone, 2009).

During the fall trips from 2000 to 2008, we deployed 9,996 hoop-net sets, yielding 233,436 hours of fishing effort. We captured 35,709 fish, of which 24,836 (70 percent) were humpback chub. Native fish again made up 89 percent of the overall catches. Nearly 4,700 humpback chub ≥150 mm received PIT tags. Between 2000 and 2006, the fall abundance estimates for humpback chub ≥150 mm ranged between 1,064 and 2,774 (fig. 5). In the fall of 2007 and 2008, the abundance estimates for humpback chub ≥150 mm increased to 4,079 and 4,750, respectively (fig. 5). For adult humpback chub (≥200 mm) between 2001 and 2006, the fall abundance estimates ranged between 483 and 1,347. In 2007 and 2008, the abundances increased to 2,247 and 1,936, respectively (fig. 5).

Chute Falls Translocation and Monitoring

Background

A question long intriguing fish biologists in Grand Canyon is why humpback chub have not recently been found in the Little Colorado River above Chute Falls, which is a naturally occurring travertine dam structure (fig. 6). The river originates as snowmelt from Mt. Baldy and continues as a perennial stream in eastern Arizona where it becomes intermittent below St. Johns, AZ, and is confined to subsurface channels during dry months. The river becomes perennial again at Blue Springs (21 km above the confluence with the Colorado River) where, combined with other springs, discharges are about 6.30 cubic meters per second (m^3/s) (Cooley, 1976). Historical evidence indicates that a native fish community previously resided well above Blue Springs to Grand Falls, a

Figure 6. Little Colorado River and Chute Falls (14.1 km).

stretch of the watershed now seasonally dry and reaching to nearly 140 km above Blue Springs. Colorado pikeminnow and bonytail were both reported from the Little Colorado River above Blue Springs in the late 1800s (Minckley, 1973). Miller (1963) reported that Colorado pikeminnow and bonytail were captured at the base of Grand Falls in the early 1900s. Additionally, skeletal remains of Colorado pikeminnow, razorback sucker, bonytail, and humpback chub have been recovered from the Homol'ovi archaeological ruins near Winslow, AZ (Strand, 1998). Widespread devastation to extensive grassland communities, erosion of topsoil, and increased variation to flows (Abruzzi, 1995) are all factors implicated in the transformation of the river between Blue Springs and Grand Falls from a formerly perennial system (Colton, 1937) to a seasonally dry sand bed. However, this does not explain the absence of humpback chub in the historically perennial reach from Chute Falls to Blue Springs (14 to 21 km). Rather, biologists have attributed the absence to Chute Falls being an impassable barrier for humpback chub (Robinson and others, 1996) or to high carbon dioxide levels in the water (Mattes, 1993).

In 2002, a conservation action was identified by USFWS, USBR, GCMRC, and the National Park Service to translocate (move) small humpback chub from the lower reaches of the Little Colorado River to above Chute Falls. This conservation action was intended to offset any potentially detrimental impacts to humpback chub from experimental releases from Glen Canyon Dam and from a project to remove nonnative fish by electrofishing in the Colorado River. It has long been assumed that small humpback chub in the lower reaches of the Little Colorado River may have poor survival rates because many are flushed into the Colorado River during monsoon flood events in the Little Colorado River. Once in the Colorado River, they are subject to cold water temperatures, low growth rates (Clarkson and Childs, 2000), and predation by nonnative salmonids (Valdez and Ryel, 1995). Because the translocated fish were moved to above Chute Falls, they were presumed less likely to be flushed into the mainstem Colorado River. Additionally, the fish were exposed to warm spring-fed water temperatures where growth rates were expected to be higher. And it was hoped that they would colonize the new habitat, thereby increasing the range of the species.

Methods

Translocations

In July 2003, 300 humpback chub (50–100 mm) were collected over a 3-day period by using seines and baited hoop nets in the lower 2.7 km of the Little Colorado River and placed in holding nets in the river. The fish were then anaesthetized, implanted with an elastomer tag, and allowed to recover overnight in an aerated tank. The following day they were moved by helicopter in an oxygenated tank to the release site at 16.2 km (fig. 3). At the release site, the fish were tempered by exchanging one-third of the oxygenated water with fresh river water every 15 minutes until carbon dioxide levels in the tank were within 10 milligrams per liter of the release site. The fish were then placed in mesh bags in the river, monitored, and allowed to acclimate overnight until release the next morning. This initial action was followed by the translocation of 300 humpback chub (50–100 mm) in July 2004, another 567 (50–100 mm) in July 2005, and another 299 (86–136 mm) in July 2008. Because of their small size, these 1,150 translocated fish were not initially implanted with PIT tags upon release, but rather were tagged with Visible Implant Elastomer (VIE) tags. Further information on these translocations is presented in D.M. Stone, U.S. Fish and Wildlife Service, written commun., 2005, and Holton (2008).

Translocation Site Monitoring

During the summers of 2006, 2007, and 2008, supplemental mark-recapture efforts were conducted above Chute Falls in the Little Colorado River between 14.1 and 18.2 km in order to track the abundance of the translocated humpback chub released at 16.2 km (upper reach; fig. 3). The supplemental mark-recapture efforts also included a small portion of river between 13.57 and 14.1 km (lower reach; fig. 3) which is not included in our primary spring and fall mark-recapture efforts because flooding in the Little Colorado River prohibited safe working conditions during those seasons. Methods for these mark-recapture efforts are nearly identical to the previously described spring and fall mark-recapture efforts and are presented in Van Haverbeke and Stone (2009).

Results

During the mark-recapture trips, 899 hoop-net sets were deployed, yielding 21,012 hours of fishing effort. We captured 34,496 fish, of which 2,960 (9 percent) were humpback chub and 31,156 (90 percent) were speckled dace (*Rhinichthys osculus).* Native fish made up 99 percent of the catches and nonnatives the remainder. Nearly 780 humpback chub ≥150 mm received PIT tags. In 2006, we estimated (by use of eqs 3 and 4) that there were 125 humpback chub ≥200 mm in the reach of river above Chute Falls where the translocated fish were released (fig. 7). The number of humpback chub

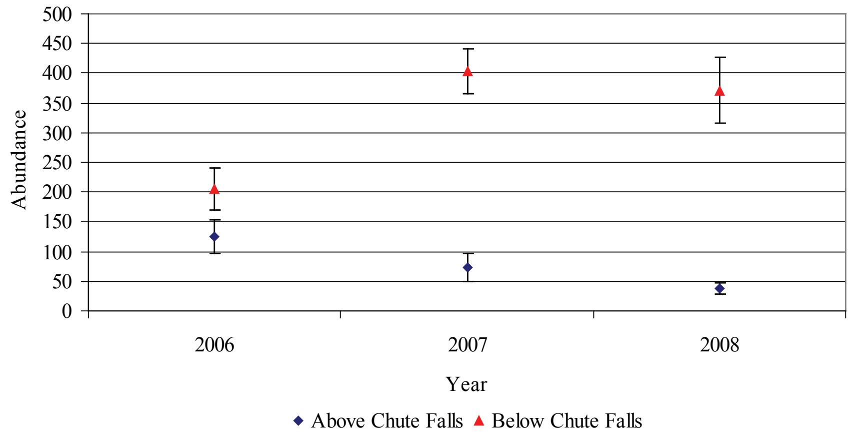

Figure 7. Abundance estimates of humpback chub ≥200 mm immediately below Chute Falls (13.57 to 14.1 km) and above Chute Falls (14.1 to 18.2 km), Little Colorado River.

in that reach declined to 37 by 2008. This suggests that the translocated fish grew to adulthood and dispersed downriver, consistent with the life history of the species. For 2006, we estimated that there were 206 humpback chub ≥200 mm in the small stretch of river (13.57 to 14.1 km) immediately below Chute Falls (fig. 7). This number increased to 403 in 2007 and was 371 in 2008. The increase in 2007 is believed to be caused by translocated humpback chub that had moved (or been displaced) downriver to immediately below Chute Falls and grew into adulthood by 2007.

Thus far, we have directly tracked 10 percent of the 2003 to 2005 translocated humpback chub to adulthood. This number is based on recapturing 112 of the total of 1,150 translocated fish by fall 2008. The recaptured fish were all ≥200 mm (i.e., reached adult size).

Discussion and Implications for Management

Mark-Recapture and the Increases in Abundance

Our mark-recapture efforts in the lower 13.57 km of the Little Colorado River demonstrate that there has been a recent increase in the abundance of adult humpback chub in the Little Colorado River in both the spring spawning season and fall. This increase is a positive sign for recovery of the species and we are cautiously optimistic. However, humpback chub still face threats, including habitat alteration, parasite infestation (e.g., the Asian tapeworm, *Bothriocephalus acheilognathi*), and predation by nonnative fish. In addition, fishery biologists are uncertain as to specifically why humpback chub are increasing in abundance. The increase of native bluehead sucker spring spawning abundance in the Little Colorado River (Van Haverbeke and Stone, 2009) would suggest some ecosystem-wide change has occurred that not only influences humpback chub abundance, but influences the native fish community as a whole. Some factors in the Colorado River that could be increasing survivorship and recruitment of native fish include a reduction in the magnitude of fluctuating flows compared to pre-Environmental Impact Statement levels (U.S. Department of the Interior, 1995), a warming trend in water temperature of the Colorado River because of drought, and a decline in the abundance of nonnative predacious salmonids (trout) in the Colorado River. It is also possible that the increases we are witnessing in native fish abundances could be partially resulting from factors associated with the Little Colorado River, such as by its hydrograph.

Translocation

The translocation efforts have been productive. We have learned that Chute Falls is not an impassable physical barrier to humpback chub, albeit this is only based on documenting four humpback chub ascending the falls. We have recorded high growth rates of the translocated humpback chub (D.M. Stone, U.S. Fish and Wildlife Service, written commun., 2005), and have directly tracked 10 percent of the 1,150 translocated humpback chub to adulthood. Possibly most important, translocating humpback chub to above Chute Falls gives them a natural rearing environment, functioning as a "wild" hatchery—a scenario much preferred to augmentation involving artificial hatchery propagation. Dexter National Fish Hatchery and Technology Center assists in the translocation project by providing guidance and by monitoring for any potential genetic consequences of the action. Finally, we have demonstrated that humpback chub can successfully be translocated, which may prove very useful for future translocations to other tributaries in Grand Canyon.

Acknowledgments

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Razorback Sucker Population Status in Lake Mohave: Monitoring, Database, Analysis, and Repatriation Program Optimization

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Abstract

A razorback sucker (*Xyrauchen texanus*) monitoring program in Lake Mohave, Arizona-Nevada, was initiated by W.L. Minckley and colleagues in the 1960s. As the razorback sucker population dwindled, the lower Colorado River Lake Mohave Native Fish Work Group (NFWG) was formalized in 1990 with representation from a suite of concerned academicians, agency biologists, and other interested parties. Primary missions of the NFWG are to capture and rear razorback sucker for repatriation, track population and genetic status, and develop management strategies. Field data accessioned into a central repository database now at Marsh & Associates, LLC, was an integral part of the lower Colorado River native fishes conservation program in general and the Lake Mohave razorback sucker program in particular. As data were accumulated and analyzed, the NFWG recommended incremental increases in total length for repatriates because length was the most important determinant of post-stocking survival. The most recent increment of 15-centimeters was from 35 to 50 centimeters, but too few monitoring data were available to assess the benefit of the last stocking size. The wild razorback sucker population in Lake Mohave is fewer than 50 individuals, and the 2007–2008 repatriate population estimate is 1,232 fish. Additional stockings of larger fish are predicted to increase the repatriate population size.

Introduction

Lake Mohave, Arizona-Nevada (fig. 1), once was occupied by the largest remaining population of wild razorback sucker (*Xyrauchen texanus*) (fig. 2). Historically, this population was made up of more than 100,000 fish, but numbers have dwindled dramatically during the past two decades, and the current estimate is fewer than 50 remaining individuals (Marsh and others, 2003; Kesner and others, 2007; Turner and others, 2007). Razorback sucker, like many other native fishes of Southwestern United States, is on a trajectory that without intervention soon will lead to its extirpation in the wild.

Arizona State University (ASU) served for nearly 20 years as a central repository of field data gathered by the lower Colorado River Lake Mohave Native Fish Work Group (NFWG), which formed in 1990 with representation from Arizona Game and Fish Department (AZGFD), ASU, U.S. Geological Survey (USGS), Nevada Department of Wildlife (NDOW), Bureau of Reclamation (BR), U.S. Fish and Wildlife Service (FWS), and National Park Service (NPS). The primary mission of the NFWG is to capture and rear native lower Colorado River fish for repatriation, in particular razorback sucker (Mueller, 1995). Wild-produced larvae are collected annually from the Lake Mohave shoreline during the winter-spring spawning season and reared initially in protective captivity at Willow Beach National Fish Hatchery (NFH) in Arizona. Off-site rearing locations historically included Boulder City, Nevada, golf course ponds and wetland ponds. Some fish are (or were) stocked directly into

Figure 1. Lake Mohave, Arizona-Nevada. Photograph by Abraham Karam.

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Figure 2. Wild razorback sucker (*Xyrauchen texanus*) captured in Lake Mohave, Arizona-Nevada. Photograph by Abraham Karam.

the lake from these sites, while others are retained at Willow Beach NFH or are transferred to various grow-out locations, including predator-free lakeside backwaters such as Yuma and Davis Coves in Arizona and Dandy and Chemehuevi Coves in Nevada, all on Lake Mohave. Once the fish attain a size thought to be relatively safe from predation (initially a nominal size of 30 centimeters (cm)), fish are PIT (Passive Integrated Transponder) tagged, measured, and stocked into the lake.

In addition to capturing larvae, the NFWG continues to oversee and implement Lake Mohave monitoring programs that periodically assess population status of wild adult and repatriated razorback sucker and other components of the fish community. W.L. Minckley at ASU and his colleagues initiated these efforts in 1968 (Minckley, 1983). Members of the NFWG annually revisit the same localities at the same times of year and deploy the same kind of collection devices, capturing untagged and previously PIT-tagged native fishes as well as many nonnative species. Field expeditions typically occur in March (also referred to as the razorback roundup), May, and November, generally targeting spawning, post-spawning, and pre-spawning periods, respectively, and employing several fishing methods, primarily trammel netting and electrofishing. During these expeditions, repatriates are captured and (or) recaptured, generally as mature adults, as they co-mingle with other repatriates and any remaining wild adults on spawning grounds, but also as juveniles at scattered locations.

Field data from stocked repatriates and adult monitoring were regularly received at ASU until the Native Fish Laboratory (NFL) was privatized in 2008 to become the NFL at Marsh & Associates, LLC (M&A). Samples are regularly received at M&A, and data are manually entered into electronic Excel (Microsoft®Excel 2003, ©1985–2003 Microsoft Corporation) spreadsheets or directly into an Access (Microsoft®Access 2003, ©1992–2003 Microsoft Corporation) database; electronic field data files generally are received in Excel spreadsheets. Data generally include collection or stocking date, collection location, stocking or rearing site with associated State and river mileage (north from Davis Dam, for Lake Mohave), Global Positioning System (GPS) coordinates in either Universal Transverse Mercator (UTM) coordinates or in latitude/longitude (in degrees/minutes), agency, gear, PIT-tag number, total length (TL, in millimeters or centimeters), weight (in grams or pounds), gender, status, and field comments. Gender categories are defined as "juvenile" (a young fish that has not attained sexual maturity and does not exhibit external secondary characters that allow reliable sex determination), male, female, and "unknown" (an adult-size fish whose gender cannot reliably be determined). Status refers to fish capture, recapture, or stocking history, and field comments are generally related to fish health but also may indicate mortality or involvement in an in-situ or hatchery research study.

All manually entered PIT-tagging data are proofed using text to speech software (Zoom Text®8.1, ©2003–2004 Ai Squared) before they are imported into the NFWG database maintained in Access; electronic field data files are generally sorted for duplicates, but not proofed. All razorback sucker data plus information on other PIT-tagged native fishes from reservoirs Mead, Mohave, and Havasu and in the Colorado River below Parker Dam are maintained in this single database, using a species/reservoir identification key to differentiate among reservoirs and a record identification number to identify each individual record regardless of location. These areas correspond with reaches 1, 2, 3, and 4/5, respectively, of the Lower Colorado River Multi-Species Conservation Program (LCR MSCP), which supports much of the on-going work on native fishes. Data queries are initiated on the basis of information requirements and generically written to accommodate any reservoir.

Several dozen requests for specific searches each year from biologists working for a suite of State and Federal entities were typically handled by NFL staff at ASU until access to the database through the Internet was made available in 2005. This change made retrieval of fish capture histories more convenient and faster for NFWG members, as the database in its entirety was no longer available to members in any software format because of its complexity and size. Currently, the Web site is managed by M&A on an externally hosted server (Hostmonster.com). In 2007, formatting changes allowed members to search for as many as three PIT-tag numbers at one time versus the previous format of searching for only one tag at a time, and an online accessible annual release summary table also was made available. Additional enhancements are in development.

In 2007, NFWG members began double tagging fish such that fish captured with older 400 kilohertz (kHz) tags generally received new 134.2 kHz tags. In the Access database, a new field was added for these latter tags, and the data were amended (release and (or) capture records) to include this new tag. This addition allows NFWG members to search the online database for either old or new tag numbers, and the complete capture history associated with both tag frequencies is returned.

A number of adjustments have been made to the NFWG program that incorporate information adapted from summarizing the database in an attempt to increase survival of stocked fish (e.g., Marsh and others, 2005). This report provides a brief summary of wild adult and repatriated population status as of March 2008 and recites general findings of recent studies by Kesner and others (2007) and Karam and others (2008).

Methods

We summarized captures of PIT-tagged wild and repatriated razorback sucker in Lake Mohave from 1990 to March 2008 using the NFWG Access database. For most of these years and for most wild razorback sucker captured, fish without PIT tags were marked and noted in the database as wild; however, beginning in 2006, this method was abandoned, and we began marking any untagged fish "repatriate." As used below, "short-term recapture(s)" were recaptures within 7 days of initial capture. For methods related to Kesner and others (2007) and Karam and others (2008), see those papers directly.

Results and Discussion

Wild Fish

During the 19-year period from 1990 to March 2008, the NFWG contacted 9,662 wild razorback sucker, and 4,101 of these were contacted two or more time(s), which also included short-term recaptures. Further analysis relied on March-only data because the most consistent and uniform field effort is applied by the NFWG during this month. Based on this dataset, the NFWG collected 2,112 fish with paired-capture data, meaning the database contained mark and any subsequent contact data for each fish. Using these March-only, paired-capture data, we found approximately 13 percent of the total (*N*=272) were at large longer than 5 years compared to the remainder (*N*=1,840) that were at large 0 (less than a year) to 5 years; 23 fish were at large from more than 10 to 15 years, and 249 fish were at large from more than 5 to 10 years. One of the first wild fish PIT tagged by the NFWG, originally marked in 1991, was not captured again until 2006, 15 years between handlings. McCarthy and Minckley (1987) estimated fish in their samples were 24 to 44 years old at the time of their capture in 1981 to 1983, making it possible that this single fish could have been 39 to 59 years old in 2006. Of the 1,840 fish at large less than 5 years, 443 fish were captured again within the same month of their marking.

Minckley (1983) and McCarthy and Minckley (1987) predicted wild razorback sucker in Lake Mohave would disappear before the year 2000. Their estimates were eerily accurate as the 2007–2008 wild population estimate is 47 individuals (24 to 175 95-percent confidence interval; single-census, Chapman modification of the Peterson method (Ricker, 1975)). In 1991–1992, more than 42,000 wild razorback sucker were estimated to persist in Lake Mohave; six times more than the number estimated 6 years later in 1997–1998 (7,196 fish estimated) and almost 900 times more that our current estimate only 16 years later.

Repatriated Fish

With the exception of a few untagged escapees, all repatriated razorback sucker were PIT tagged before stocking into Lake Mohave (table 1). With the exception of three out of 15 years, the average TL at release was approximately 30 cm even though target length was 25 cm for 1999 and previous

Table 1. Stocking summary of PIT-tagged razorback sucker repatriated into Lake Mohave, 1992–2007 (total *N* = 127,842).

[TL, total length; cm, centimeter; *N*, number; Avg, average; SD, standard deviation; Min, minimum; Max, maximum]

Year	*N* Fish[a]	TL (cm)			
		Avg	SD	Min	Max
2007	1,282	40	7	23	59
2006	11,341	38	3	23	56
2005	12,208	37	3	14	55
2004	17,268	35	3	21	58
2003	16,844	33	3	18	53
2002	10,978	32	3	14	55
2001	11,431	32	3	21	55
2000	7,160	30	5	21	55
1992–1999	39,330	18–35	3–5	10–27	43–62

[a] Total *N* fish from 2000 to 2007 = 88,512.

years. Over time, the NFWG recommended incremental increases in TL at release because length was found to be the most important determinant of post-stocking survival (Marsh and others, 2003, 2005; Kesner and others, 2007). Approximately in year 2000, target size was increased to 30 cm, followed by 32.5 cm in 2003, 35 cm in 2004, and finally it was raised to 50+ cm in 2006. As target size increased, fewer fish were stocked because there was a lag time for grow-out facilities to rear their fish to the new, larger sizes.

From 1992 to March 2008, the NFWG captured 2,667 razorback sucker and 1,917 of these were contacted again (including short-term recaptures). From the March-only captures, 1,209 fish had paired-capture data. Similar to wild fish, we tracked time at large for the repatriates, and in some cases when year class was known, we also knew the exact age of the fish. Twenty-three percent of stocked fish (*N*=274) were more than 5 years at large, with the remainder 0 (less than a year) to 5 years at large (*N*=938). Three fish were at large between 15 to 16 years.

In reviewing population estimates for repatriated razorback sucker (table 2), it appeared that NFWG effort over the years was only maintaining the population and not necessarily moving toward a larger population size. Contrary to predictions (Marsh and others, 2005), increases in size (TL) at stocking did not measurably increase population estimates. Confidence intervals were relatively narrow, so we are reasonably confident in our estimates. As a result, with overall survivorship declining even though fish stocking continued, we explored the fate of repatriated fish, other than the obvious consumption by nonnative aquatic species.

One assumption was that survival of repatriated fish, once they joined the adult population, would be higher than the estimated survivorship of the wild fish (approximately 75 percent annually) because the wild fish were believed to be reaching the end of their life span. There was no detectable decline in wild fish population size in the 1980s, which indicated adult razorback sucker survival was much higher at that time. However, our data from March-only samples and a basic mark and recapture model (fig. 3) showed that annual survival in Lake Mohave of repatriates at 45 and 50 cm at their first capture was similar to wild fish. We also found that annual capture was about 10 percent of each spatially defined

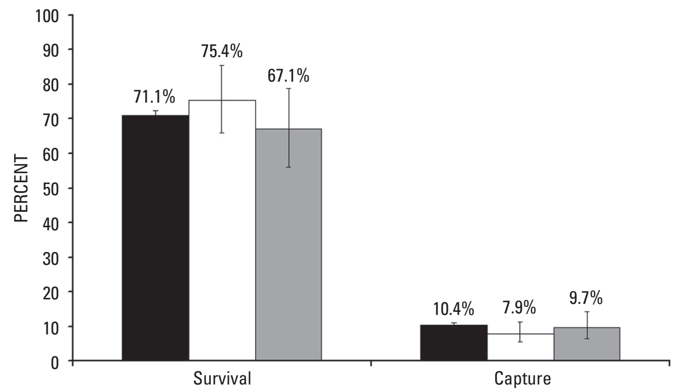

Figure 3. PIT-tagged wild (black), repatriated adults more than 450 mm TL (open), and repatriated adults more than 500 mm TL (grey) annual survival and capture in Lake Mohave.

group in the population (fig. 4; see Kesner and others (2007) for more detailed information).

NFWG members were concerned that using March-only sampling data could bias the population and survival estimates. We, therefore, conducted a mark-recapture analysis that incorporated year-round capture data with some level of site identification (Kesner and others, 2007). The model focused on captures from the three central zones because these zones represent the majority (80 percent) of captures and also have the most consistent year-around capture data. We used capture events that were summarized for each zone by month and months where all three zones were sampled. Figure 4 depicts Lake Mohave in its entirety, from Hoover Dam to Davis Dam, the three central zones (Yuma, Tequila, and Nine Mile), and six other zones above and below the central area. Analysis represented the period 1996–2008, during January, February, March, April, and November (summer months generally were not sampled) of each year. A total of 1,659 fish were captured: 514 in Nine Mile zone, 475 in Tequila zone, and 670 in Yuma zone. Estimates of annual survival for two of the three zones were similar to those estimated from the March-only, nonsite-specific mark-recapture analyses. Transition rates demonstrated that razorback sucker readily moved from one zone to the next. Even though survival in the Tequila zone was elevated, fish did not remain in any one zone long enough to enjoy the benefits of that zone. These results demonstrate that the March-only, nonsite-specific analysis is unbiased and adequately represents the Lake Mohave population at large.

We also wanted to assess the relation between post-stocking repatriate survival and size at release (see Karam and others (2008) for more detailed information). Toward that end, three acoustic telemetry studies (2006–2007, 2007–2008, and 2008–2009) were initiated using two size groups of razorback sucker: sub-adults (TL = 38 cm) and adults (TL = 50+ cm). All inactive fish were investigated and their transmitters were recovered using SCUBA and an underwater diver receiver (Sonotronics, Inc.). Concurrent with the first year of field study, 20 razorback sucker were implanted with acoustic

Table 2. PIT-tagged repatriated razorback sucker population estimates in Lake Mohave.

[CI, confidence interval]

Data years[a]	Population estimate	Lower CI at 95%	Upper CI at 95%
2007–2008	1,232	662	2,318
2003–2004	1,508	663	3,660
1998–1999	1,173	482	3,118

[a] March-only data using single-census, Chapman modification of the modified Peterson method (Seber, 1973).

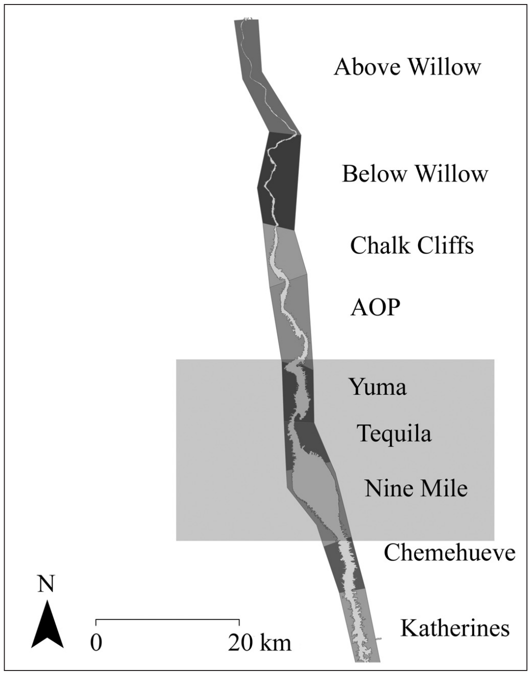

Figure 4. Lake Mohave and primary repatriate capture zones: Yuma, Tequila, and Nine Mile.

transmitters and held in an outdoor raceway for 3 months. All captive fish remained healthy, their growth was positive, and some individuals showed obvious reproductive signs (milt and egg production), indicating that our surgical procedures did not compromise fish health or behavior. Additionally, no transmitters were shed during the captive fish study, which suggests that recovered transmitters from the telemetry work in the lake represent fish mortality and not transmitter loss. Preliminary field results indicated 6-month survival for sub-adults was low (between 7 and 16 percent). Six-month survival for adults (36 percent) was five times greater than for sub-adults during 2007–2008. Weekly survival of adults was always higher than sub-adults. Survival estimates, based on weekly survival rates for all groups of fish, indicated a significant difference in survival between adult and sub-adult fish. A subsequent study (2008–2009) will compare hatchery sub-adults with adults reared in lakeside backwaters.

Implications for Management

The NFWG has been monitoring razorback sucker for nearly 20 years, and its database currently maintains almost 150,000 PIT-tag records. The wild population estimate decreased from tens of thousands to fewer than 50 individuals during this time, while large repatriated fish were stocked by the thousands. The NFWG actively reviews monitoring data and analyzes those data for the optimization of the repatriation program; however, for now, too few monitoring data were available to assess the benefit of increased stocking size to the recommended minimum of 50 cm.

Conservation plans for big-river fishes in the lower Colorado River (Minckley and others, 2003; U.S. Fish and Wildlife Service, 2005) incorporate a population component that will occupy the mainstream, but it may be impractical or impossible to accommodate that plan. For example, it is documented that long-term persistence is near zero for razorback sucker stocked into the lower Colorado River downstream of Parker Dam (Schooley and others, 2008). If main channel populations cannot be developed and maintained, conservation of razorback sucker in the lower river may depend entirely on populations in off-channel habitats that are free of nonnative fishes. An objective of this continuing research is to provide information needed to determine how each of these strategies should contribute to maintenance of razorback sucker in Lake Mohave and throughout the lower Colorado River. Moreover, our results will provide critical demographic information and management recommendations to help ensure the long-term persistence of a genetically viable stock of adult razorback sucker in Lake Mohave.

Acknowledgments

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Colorado River Campsite Monitoring, 1998–2006, Grand Canyon National Park, Arizona

By Matt Kaplinski,[1] Joseph E. Hazel, Jr.,[1] and Rod Parnell[1]

Abstract

Recreational use along the Colorado River in Grand Canyon is highly dependent on sandbars used as campsites. Campsite area changes in Grand Canyon National Park were studied between 1998 and 2006 by comparing annual surveys and visual observations of campsite area. High-elevation campsite area was surveyed at 38 sandbars commonly used as campsites by river runners and hikers. The results show that during the 8-year period of study the total amount of campsite area decreased by 56 percent. The primary factors in campsite loss are riparian vegetation growth and sandbar erosion, but the effects vary, largely depending on river width and sandbar size.

Statistical trend analysis shows that the decrease in campsite area is significant despite a 29 percent increase in area between 2003 and 2005. The increase occurred as a result of sand deposition and some vegetation burial or removal during a November 2004 high-flow release. The continued existence of sandbars suitable for camping depends on high flows to redeposit sediment eroded by dam releases and bury or scour established vegetation. The creation and maintenance of open sandbar areas are required to offset increasing riparian vegetation increases along the river banks; otherwise, management goals for campsite availability in this system will not be met.

Introduction

Visitors to the Colorado River in Grand Canyon National Park typically use sandbars as campsites. The presence and operation of Glen Canyon Dam has eroded sandbars and has reduced the sand available for maintaining them (Rubin and others, 2002; Wright and others, 2005). Closure of Glen Canyon Dam in 1963 not only cut off the upstream supply of sediment but also the flood flows that annually reorganized the configuration of sandbars and scoured riparian plants from the banks of the river. The number and size of plants quickly increased, colonizing areas previously available for camping (Turner and Karpiscak, 1980; Kearsley and Ayers, 1996; Webb and others, 2002). This interrelated effect of the changed hydraulic regime of Glen Canyon Dam, along with other contributing factors such as surface-water runoff (Melis and others, 1994), aeolian processes (Draut and Rubin, 2008), and human impact (Phillips and others, 1986), has substantially reduced the area available for camping (Kearsley and others, 1994; Kaplinski and others, 2005).

Because the interest in recreation in Grand Canyon National Park has risen dramatically since the mid-1960s, modern river management is concerned about the relative size, distribution, and quality of campsites along the river corridor (U.S. Department of the Interior, 1995; Glen Canyon Dam Adaptive Management Program, 2001; U.S. Department of the Interior, 2005). Following the Record of Decision (ROD) for the Final Environmental Impact Statement for Glen Canyon Dam operations in 1996 and the establishment of the Glen Canyon Dam Adaptive Management Program (U.S. Department of the Interior, 1996), a campsite monitoring project was initiated in 1998 by the U.S. Geological Survey's (USGS) Grand Canyon Monitoring and Research Center (GCMRC) (Kaplinski and others, 2005). The goal of the monitoring project is to evaluate the management objectives of the program, specifically management objective 9.3 to "*increase the size, quality, and distribution of camping beaches in critical and non-critical reaches in the mainstem…*" (Glen Canyon Dam Adaptive Management Program, 2001). Results from the project indicated that the rates of campsite decrease were still high after four decades of flow regulation, with more than half of the camping area under study lost by 2003.

In this paper, we build on the monitoring results of Kaplinski and others (2005) and present a longer term view of changes in the size of camping areas between 1998 and 2006. During this 8-year period, detailed field measurements were made annually or less frequently at as many as 38 sandbars located throughout the 364-kilometer reach of the Colorado River ecosystem (CRE) between Lees Ferry and Diamond Creek, AZ (fig. 1). Changes in campsite area were compared among years and between critical and noncritical reaches. As defined by Kearsley and Warren (1993), a critical reach

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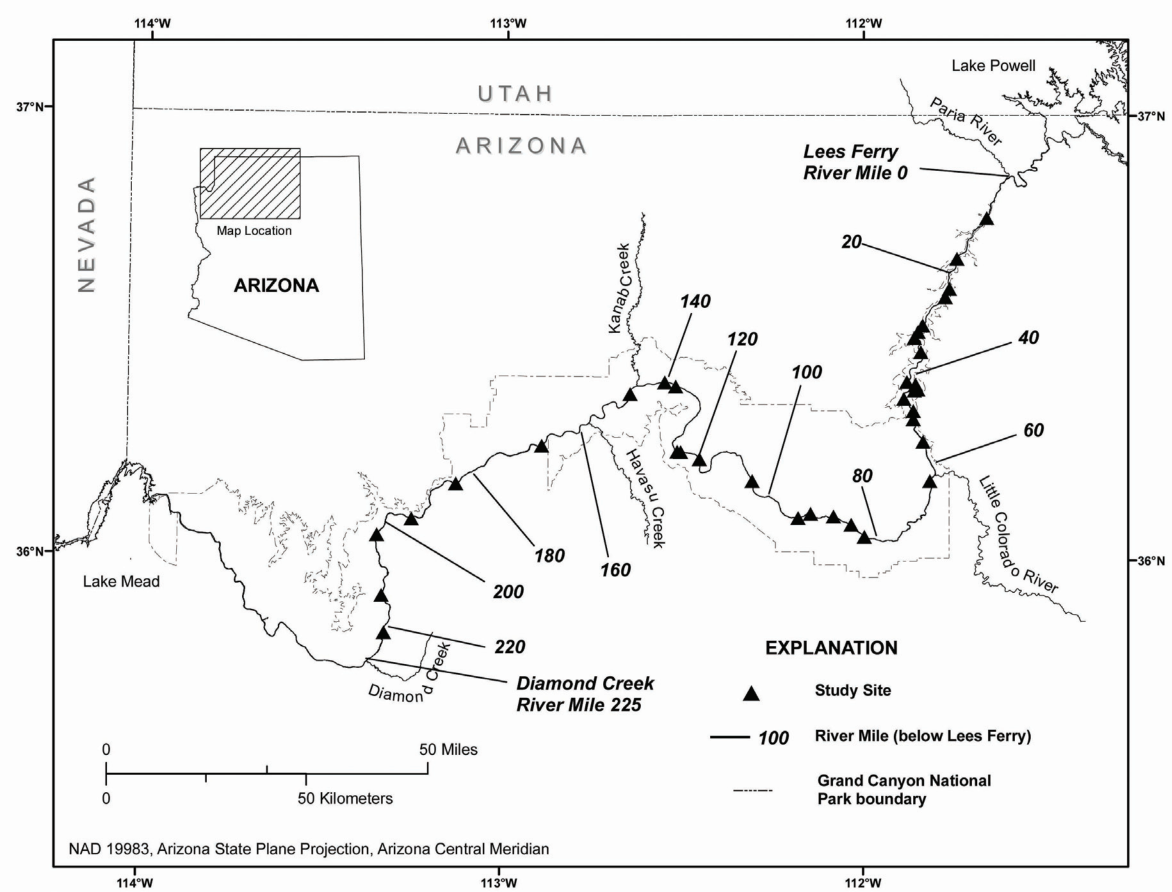

Figure 1. The Colorado River corridor between Glen Canyon Dam and the western boundary of Grand Canyon National Park. Study site locations are indicated with triangles. The use of river mile has a historical precedent and provides a reproducible method for describing locations along the Colorado River in Grand Canyon. Lees Ferry (RM 0) is the starting point.

is any contiguous stretch of the river in which the number of available campsites is limited because of geomorphic setting, high demand for nearby attraction sites, or other logistical factors. Noncritical reaches are those stretches in which campsites are plentiful, resulting in little competition for the majority of sites. In addition, campsite area changes were compared to changes in sandbar size to evaluate the effects of changing sandbar morphology on campsite area. An overview of previous studies of the number, size, and distribution of campsites along the Colorado River can found in Kaplinski and others (2005).

Methods

Study Site Selection

This study evolved from a sandbar monitoring project initiated in 1990 that measured changes in topography and sediment storage at as many as 45 study sites located throughout the CRE (Beus and others, 1992; Kaplinski and others, 1995; Hazel and others, 1999). Beginning in 1998, we measured campsite area at a subset of the study sites, using the same survey techniques employed to survey the topography of sandbars previously. Despite the less than optimal study design, this strategy afforded a number of advantages. By using the same study sites, well-defined stage-discharge relations (Hazel and others, 2006) could be used to partition campsite area changes between discreet stage-elevation ranges, and changes in camping area can be directly compared to sandbar area. In addition, measuring both campsite area and sandbar area on the same river trip resulted in considerable logistical cost savings.

Campsite area measurements were collected at 31 of the 45 sandbar study sites, as several of the sandbar study sites were not suitable for campsite area monitoring. Seven sites were added in 2002 for a total of 38 study sites (table 1). Seventeen sites are located in Marble Canyon (the reach of the CRE located between the Paria River and the Little Colorado River), and 21 sites are located in Grand Canyon, downstream from the Little Colorado River confluence (fig. 1). Nineteen sites are located within critical reaches, and 19 sites are in noncritical reaches (table 1). The study sites are named according to river-mile location. Distances along the Colorado River in Grand Canyon are traditionally measured in river miles (RM) upstream (–) or downstream from Lees

Table 1. Study site location and area changes from 1998 to 2006.

[m^2, square meter; s.d., standard deviation. No data were collected in 2004]

River mile*	Side**	Reach#	1998 area (m^2)	1999 area (m^2)	2000 area (m^2)	2001 area (m^2)	2002 area (m^2)	2003 area (m^2)	2005 area (m^2)	2006 area (m^2)
8.0	L	C					237	468	324	460
16.6	L	C	367	362	395	68	77	89	215	215
16.7	L	C	117	133	180	76	65	76	41	41
22.1	R	C	66	43	152	147	106	74	382	179
23.5	R	C					9	5	21	8
29.5	L	C					182	177	175	153
30.7	R	C	297	352	99	74	35	28	566	270
31.9	R	C	642	675	618	572	315	487	428	420
35.0	L	C	463	542	497	470	442	445	452	475
41.2	R	NC					531	621	409	381
43.4	L	NC	1,105	1,014	933	526	505	126	134	147
44.5	L	NC	599	626	534	453	512	567	644	461
45.0	L	NC					183	84	778	287
47.6	R	NC	765	799		269	199	359	212	272
50.1	R	NC	702	785	755	717	786	534	588	338
51.5	L	NC	1,277	653	544	267	420	228	119	147
55.9	R	NC	548	424	273	195	126	30	119	0
62.9	R	NC	180	172	185	82	46	26	174	53
81.7	L	C	1,167	1,130	1,181	1,111	846	532	959	859
84.6	R	C			97		20	19	13	15
87.7	L	C	200	158	169	123	169	140	90	103
87.8	L	C	313	193	236	151	133	92	160	103
91.7	R	C	286	286	301	307	209	271	280	166
93.8	L	C	204	162	352	210	223	143	184	219
104.4	R	C	133	98	135	158	138	81	80	55
119.4	R	NC	317	300	631	328	177	174	685	156
122.8	R	NC	472	456	289	222	273	373	272	178
123.2	L	NC	376	402	295	224	158	41	180	210
137.7	L	C	627	573	786	685	838	643	630	625
139.6	R	C	323	286	179	61	78	107	71	74
145.9	L	C	118	114	289	178	152	121	182	154
167.1	L	NC					201	162	159	192
183.3	R	NC	146	136	179	143	85	65	144	72
183.3	L	NC	391	114	199	192	176	150	35	40
194.6	L	NC	1,124	817	776	596	723	511	487	416
202.3	R	NC	740	715	526	745	432	383	686	417
213.3	L	NC	411	216	128	78	51	16	28	31
220.1	R	NC	1,600	1,109	1,010	1,140	660	428	232	249
		median	391	362	295	223	183	147	198	179
		s.d.	387	315	295	293	241	201	244	188

* By convention, river mile is used to describe distance along the Colorado River.

** Side of the river as viewed in a downstream direction. L is left, R is right.

C is critical reach, NC is noncritical reach, as defined by Kearsley and Warren (1993).

Ferry, AZ (RM 0), which is the starting point. We adhered to use of the GCMRC mileage system (table 1; U.S. Geological Survey, 2006). This study did not evaluate campsites upstream from Lees Ferry in the Glen Canyon reach (RM –15 to 0) or downstream from Diamond Creek (RM 225).

Data Collection and Analysis

Surveys were conducted using standard total-station survey techniques. The accuracy and precision of these techniques have been assessed by Hazel and others (2008). Individual points collected with total stations in the CRE have a minimum vertical and horizontal error of ±0.05 meters (m). The campsite surveys were accomplished by selecting points that outlined the perimeter of each camping area, as well as collecting points to exclude features such as trees, bushes, and rocks. The perimeter points were then used to define polygons of campsite area (fig. 2). We adopted the criteria of Kearsley (1995) and Kearsley and others (1999) to identify campable area, which was defined as a smooth substrate (most commonly sand) with no more than an 8 degree slope and little or no vegetation. Slope angle was qualitatively determined visually in the field. Campsite area mapping involves a certain degree of subjectivity when mapping selected areas at a given sandbar following the criteria outlined above. Nonetheless, a direct comparison of the campsite maps collected on the same day by two different survey crews yielded a difference in area between the two surveys of less than 3 percent (Kaplinski and others, 1998).

In this paper, we focus on changes above the elevation reached by a discharge of 25,000 cubic feet per second (ft^3/s); this topographic level is the highest reached by normal ROD operations.[2] We use the term high-elevation campsite area to denote camping area above this level and used the stage-discharge relations developed by Hazel and others (2006) for calculating the area above this level. Lower topographic levels may be available for camping during low-flow months but were not mapped in all years because several surveys were conducted at higher flows than others. All surveys were conducted in October with the exception of the May 2005 survey. The interval of time between the surveys and changes in flow regime is shown in figure 3. There were two high-flow events during the study period that exceeded the 25,000 ft^3/s stage elevation reached by ROD operations and were sufficient to inundate or partly submerge high-elevation

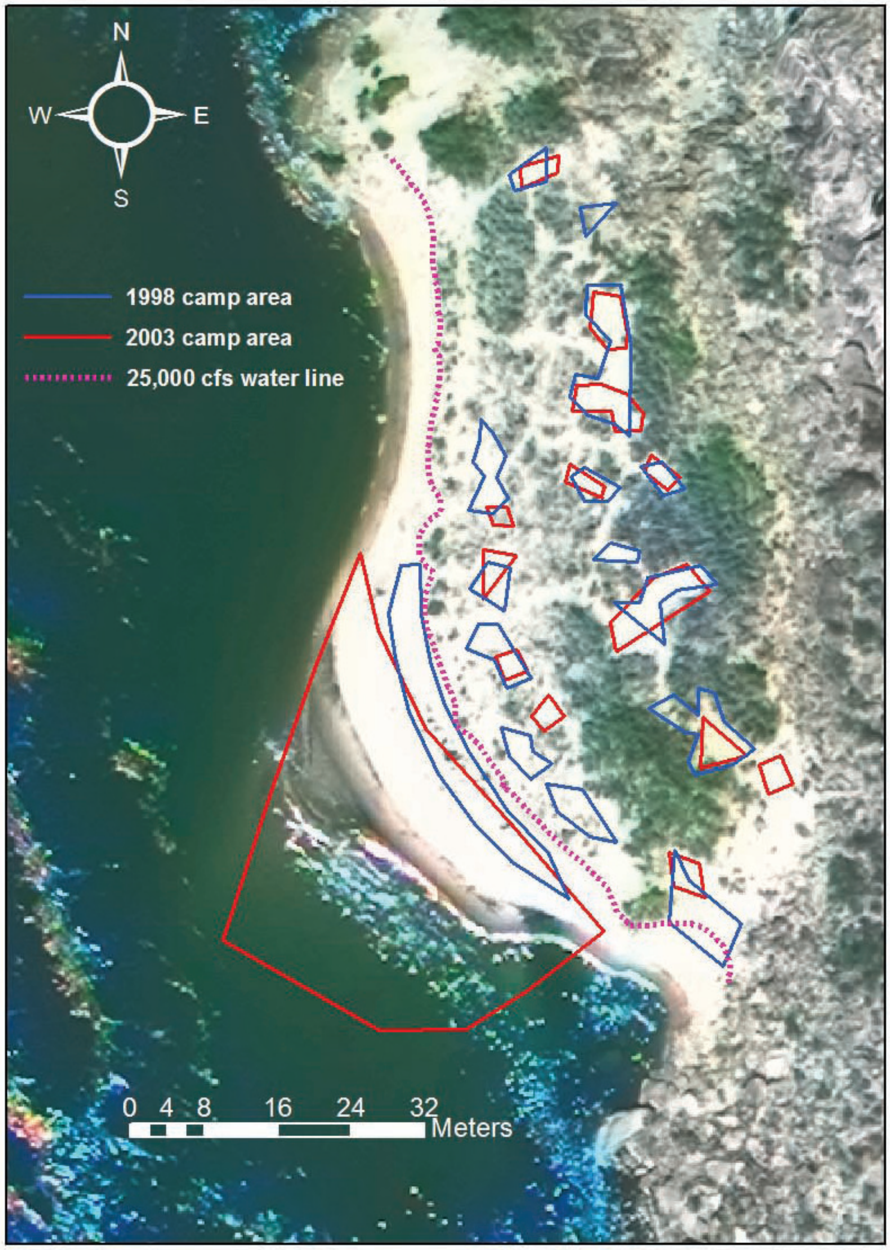

Figure 2. Aerial photograph taken in May 2002 of the 119.4-mile study site at a discharge of approximately 8,000 cubic feet per second (ft^3/s). The campsite area polygons surveyed in 1998 and 2006 are shown by blue and red lines, respectively. Also shown is the location of the 25,000 ft^3/s stage elevation line (purple) in 2006. Note that this 2002 orthophotograph does not reflect the size, height, and morphology of the lower elevations of the sandbar in other years because of inundation and erosion or deposition during flow releases from Glen Canyon Dam.

[2] By convention, cubic feet per second (ft^3/s) is the unit used to measure flow volumes from Glen Canyon Dam and the unit used to specify release volumes in the Record of Decision (U.S. Department of the Interior, 1996).

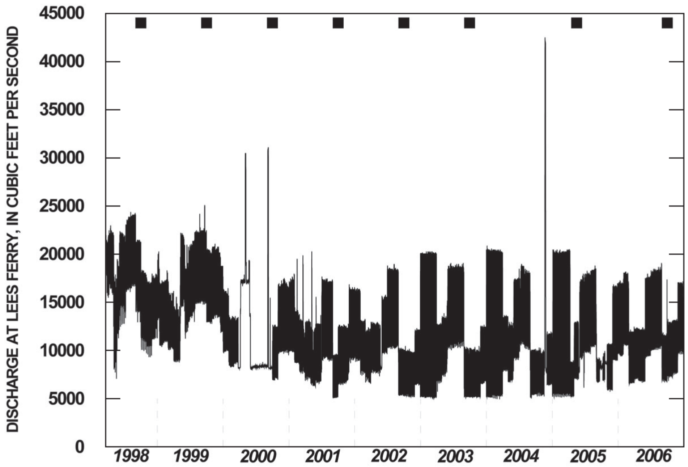

Figure 3. Daily mean discharge hydrograph from the USGS streamgaging station at the Colorado River near Lees Ferry (09380000) during the period of study. The squares indicate the survey date. Note the daily and seasonal fluctuations in flow volume, the May and September 31,000 cubic feet per second (ft^3/s) powerplant capacity flows during the 2000 LSSF, and the November 2004 HFE of 41,000 ft^3/s.

campsite areas, leading to changes that could be confidently attributed to these events. These two events were the low steady summer flow (LSSF) experiment in 2000 that included two high-flow releases in the spring and fall that bracketed a period of low, steady 8,000 ft^3/s flow (no diurnal fluctuation) and the 2004 high-flow experiment (HFE). The two high flows during the 2000 LSSF were 4-day releases of powerplant capacity (≈31,000 ft^3/s) in May and September, respectively; the 2004 HFE consisted of a short-duration (60-hour) release of 41,000 ft^3/s beginning on November 21, 2004. Both experiments were partly designed to test whether or not tributary sediment input could be redistributed to the banks to rebuild eroded sandbars (Topping and others, 2006; Schmidt and others, 2007).

Results

Overview of Campsite Area Changes, 1998–2006

The study sites progressively decreased in campsite area between 1998 and 2006, with the exception of short-lived increases following the 2000 LSSF and 2004 HFE (table 1). Generally, campsite area decreased because of erosion from bank retreat and vegetation growth. The 2004 HFE was the most significant aggradational event to occur during the 8-year study, although the 2000 LSSF did result in minor deposition (Schmidt and others, 2007). Campsite area increases occurred in some years without high flows, a surprising finding that we attribute to human impacts such as trailing and vegetation pruning or removal, surface wind reworking, or survey error. Despite the substantial variability in response from site to site, campsite area declined steadily between surveys, with the exception of area increases observed following the 2004 HFE (table 1). Twenty-six out of the original 31 sandbars were smaller in 2006 than in 1998. The median size of campsites in 2006 was only slightly greater than that measured in 2003 (a year before the 2004 HFE) and less than the size in any other year except 2003 (table 1).

Responses at Specific Campsites

We attribute the variability in campsite area decrease to the compounding effects of vegetation growth and sandbar deposition and erosion. The changes at RM 202.3 are typical of campsite loss caused by vegetation growth (fig. 4). This site is located in a wide, noncritical reach in western Grand Canyon. In 1998, the camp extended the 130 m length of the sandbar, and three stands of mature tamarisks (*Tamarix* spp.) were present. Aeolian dunes were present at higher elevations behind the tamarisk with scattered woody vegetation. By

Figure 4. Repeat photographs of the sandbar and campsite located at RM 202.3. Flow in main channel is from right to left. Between 1998 and 2006, woody vegetation, primarily arrowweed, had expanded to cover large areas of the formerly sandy, unvegetated sandbar (photographs A and B). The indicated flows at the time of the photographs are estimates on the basis of travel time between USGS streamgaging stations on the Colorado River.

2006, arrowweed (*Pluchea sericea*) had colonized and expanded to a dense thicket along the front of the bar and in between the tamarisk stands. The tamarisk stands increased only slightly in size, and the sandbar was relatively stable during the 8-year period. The 2004 HFE aggraded the bar at high elevation to such an extent that it was still greater in both area and volume in 2006 than in 1998 (Hazel and Kaplinski, Northern Arizona University, unpub. data, 2009). Despite the gains in sandbar area and volume, the campsite area in 2006 was 44 percent less than that measured in 1998 (table 1). About one-half of the unvegetated sandbar shown in the 1998 photograph was densely vegetated, and the downstream end of the camp was largely abandoned (fig. 4).

The changes at RM 30.7 illustrate campsite area increase resulting from deposition during flooding and subsequent loss from erosion (fig. 5). This site is located in Marble Canyon in a critical reach characterized by a narrow, bedrock-defined channel. Several mature individual tamarisks are located at the sand/talus slope interface, but the sandbar is largely devoid of vegetation. The sandbar was substantially aggraded during the 1996 high-flow release that peaked at 45,000 ft^3/s. The size and volume was more than double that measured in 1990, at the beginning of the sandbar monitoring project (Hazel and others, 1999). Subsequent reworking by medium- to high-volume (10,000 to 25,000 ft^3/s) operations in 1997 and 1998 reduced the sandbar area and volume such that by 1998 the high-elevation campable area was limited to a relatively small area located above a 1.5-m cutbank on the

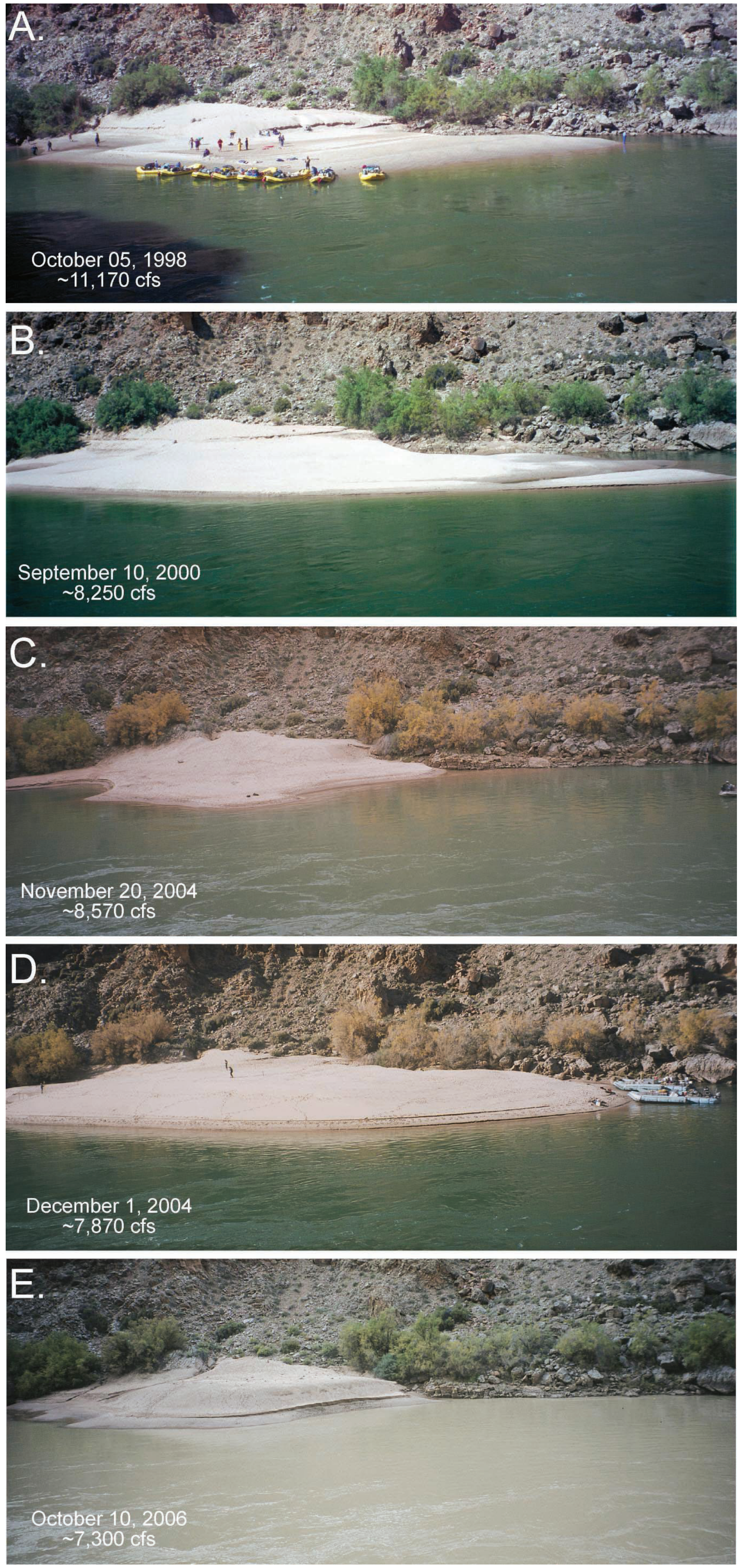

Figure 5. Time series of repeat photographs of the sandbar and campsite located at RM 30.7 illustrating campsite area changes at a nonvegetated sandbar. Flow in main channel is from right to left. The photograph in A shows the bar in 1998 after 2 years of erosion following the 1996 high-flow release that substantially aggraded the sandbar. The photographs in B and D were taken shortly after the 2000 LSSF powerplant capacity flows and the 2004 HFE, respectively. Subsequent erosion following the high-flow events are shown in C and E. The indicated flows at the time of the photographs are estimates on the basis of travel time between USGS streamgaging stations on the Colorado River. (cfs is cubic feet per second)

upstream end (fig. 5). Below the cutbank, a broad, gently sloping surface provided campable area during low-flow months. The high flows during the 2000 LSSF did not result in high-elevation deposition at this site, and high-elevation campsite area decreased by 253 square meters (m^2) (table 1). Between 1998 and 2003, high-elevation campsite area decreased from 297 to 28 m^2 (table 1). The November 20, 2004, photograph shows a small bar with little remaining high-elevation campsite area (fig. 5). Shortly thereafter, the 2004 HFE resulted in substantial rebuilding of the eroded bar (Topping and others, 2006) with a corresponding campsite area increase of 538 m^2 measured 5 months later in May 2005 (table 1). Subsequent erosion between 2005 and 2006 decreased the campsite area by 296 m^2 to levels similar to those measured in 1998. Surface-water runoff and gully formation on the downstream end of the sandbar also contributed to the loss of high-elevation campsite area during this time (fig. 5E).

Temporal Patterns of Campsite Area Change

Total campsite area changes for the CRE were derived by summing the campsite area measurements for all sites that could be compared for the 8-year study (fig. 6). Between 1998 and 2006, the total campsite area decreased by 56 percent. Despite the site-to-site variability, the total campsite area decrease was fairly consistent between surveys, with the exception of the increase from 2003 to 2005. Between 1998 and 2003, campsite area declined by an average of 14.5 percent per year. Because of deposition by the 2004 HFE, campsite area increased by 29 percent between 2003 and 2005. These gains were short-lived, however, and campsite area decreased by 24 percent between 2005 and 2006, effectively eliminating the positive effects of the 2004 HFE. Although campsite area at lower elevations increased because of deposition from high-flow events associated with the LSSF experiment in 2000, high-elevation campsite area was largely unaffected except at a few sites (table 1).

We conducted a trend analysis on the campsite area data versus time in order to test the statistical significance of the observed decrease in campsite area (Helsel and Hirsch, 2002). We tested the trend for all sites combined (fig. 6) and for critical and noncritical reaches (fig. 7). First, we tested the null hypothesis that the data are normally distributed using the Shapiro-Wilk statistic, r (Shapiro and others, 1968). For all reaches, n = 8 and the critical statistic value at 95-percent confidence is 0.906 (Helsel and Hirsch, 2002, table B3). The Shapiro-Wilk statistic for each reach (Lees Ferry to Diamond Creek r = 0.938, critical r = 0.959, noncritical r = 0.962) was greater than the critical value. Therefore, we fail to reject the null hypothesis and can test the trend using a parametric regression.

A linear regression line was constructed for the campsite area data for each reach (figs. 6 and 7). The linear regressions were tested for significance using a one-way analysis of variance (ANOVA) procedure. The results of the trend analysis show that the trend lines are significant to the 95-percent confidence level, with the exception of the trend line in critical reaches, which was significant to the 93-percent level. Despite the slightly lower significance level in critical reaches, we reject the null hypothesis that no trend exists in the data and conclude that, between 1998 and 2006, there is a significant decreasing trend in the total amount to campsite area for all reaches.

Spatial Patterns in Campsite Change

In a study of campsite area using aerial photographs taken between 1973 and 1991, Kearsley and Warren (1993) found that campable area in critical reaches decreased primarily because of erosion; in noncritical reaches, decrease in campsite area was attributed to increased vegetative cover. We separated the study sites in this study into the same critical and noncritical reaches of Kearsley and Warren (1993) to examine if this response pattern was still prevalent during our study. The results indicate a similar response as that observed by Kearsley and Warren (1993) between campsite changes in critical and noncritical reaches, but differences were found in the magnitude of loss (fig. 7). From 1998 to 2006, total campsite area in noncritical reaches decreased by 71 percent; whereas, in critical reaches the change was much less, with a total decrease of 25 percent. In critical reaches, high-elevation deposition during the 2000 LSSF and 2004 HFE is reflected by a

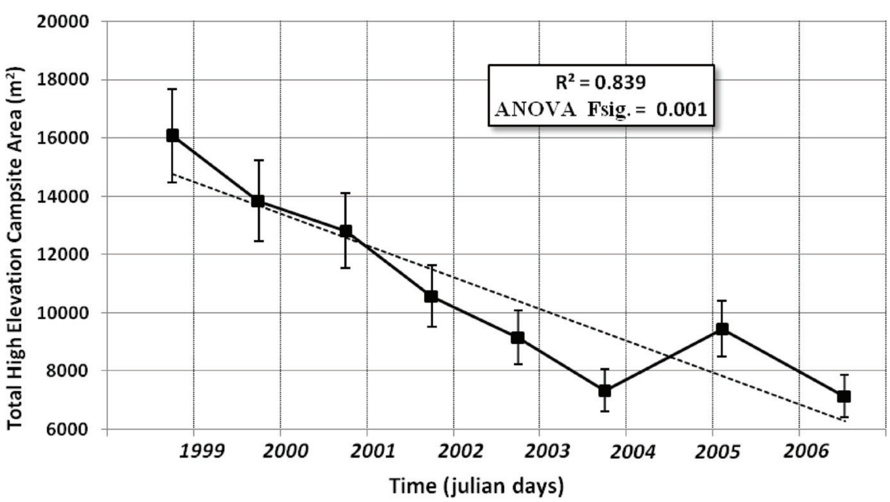

Figure 6. Total high-elevation campsite area for each survey between 1998 and 2006 (with 10 percent uncertainty). The dashed line shows the linear regression fit. Regression coefficient of determination and significance of one-way analysis of variance (ANOVA) are also shown.

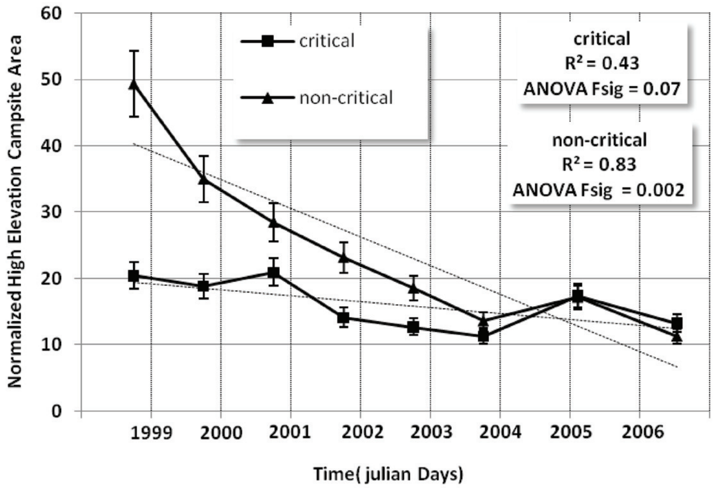

Figure 7. High-elevation campsite area in critical and noncritical reaches between 1998 and 2006 (with 10 percent uncertainty). The dashed lines show the linear regression fit. Regression coefficient of determination and significance of one-way analysis of variance (ANOVA) are also shown.

9 and 41-percent increase, respectively. In contrast, noncritical campsite area decreased 15 percent between the surveys bracketing the 2000 LSSF, and the increase following the 2004 HFE was much smaller (14 percent). The trend analysis described above shows that the loss in campsite area in both critical and noncritical reaches is significant (fig. 7). These results suggest that campsite area changes in critical reaches were more closely linked to deposition on the sandbars during the 2000 LSSF and 2004 HFE, and even though the bars quickly eroded following the high flows, the rate of campsite loss between 1998 and 2006 was less than that in noncritical reaches. Clearly, both erosion and vegetation growth reduce campsite area, but the processes and their effects are not identical between critical and noncritical reaches.

Comparison of Campsite and Sandbar Areas

In order to explain the difference between erosion and vegetation growth in critical and noncritical reaches, we compared changes in campsite area to sandbar area during the study period (fig. 8). Because there was not complete overlap of topographic and campsite surveys on the same date, this comparison is only possible for data collected between 2001 and 2006. For these surveys we calculated the total amount of high-elevation sandbar area to compare with the campsite area measurements collected on the same day. The results show that campsites in critical reaches make up a greater portion of the sandbar than campsites in noncritical reaches. In noncritical reaches, sandbar areas are 78 percent larger than campsite areas, while the difference is only 46 percent in critical reaches (fig. 8). We quantitatively examined this relation by comparing the total high-elevation sandbar area metric for the same surveys and used the Kendall correlation coefficient (t) to measure the strength of association between the campsite and sandbar area (Kendall, 1975). The Kendall statistic measures whether the pattern of variation is unrelated or if one variable generally increases (or decreases) as the second increases (or decreases). The null hypothesis is that no correlation exists between campsite area and sandbar area. The Kendall coefficients show that campsite area and sandbar area was correlated in critical reaches ($t = 0.8$, $p = 0.084$, $t < p$) but not in noncritical reaches ($t = 0$, $p = 0.6$, $t > p$). Importantly, while the statistical power of this test is rather low, $n = 5$, the result makes intuitive sense when examining the difference between critical and noncritical reaches. Critical reaches, with the exception of the Deer Creek area, are located within narrow geomorphic reaches that typically have smaller and fewer sandbars (Kearsley and others, 1994). Noncritical reaches are characterized by wide, alluvial banks with large and abundant sandbars that typically are covered with riparian and fluvial marsh vegetation (Kearsley and others, 1994). Therefore, campsites within critical reaches, where the campsites constitute a greater percentage of the entire sandbar (approximately 50 percent), correlate to changes in sandbar area, whereas campsites in noncritical reaches, where campsites only occur on approximately 20 percent of the entire sandbar, do not. To put it more simply, erosion and deposition of sandbars is the primary cause of campsite loss in critical reaches, and vegetation encroachment is the primary cause of campsite area loss in noncritical reaches.

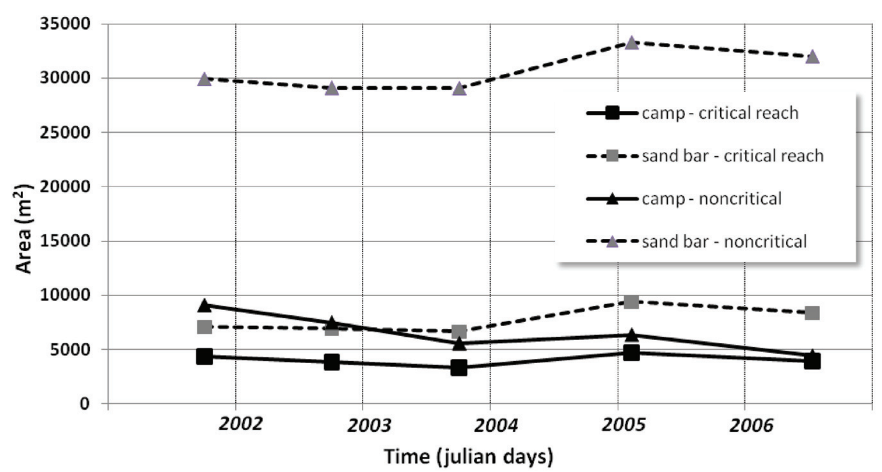

Figure 8. High-elevation campsite area and sandbar area in critical and noncritical reaches between 1998 and 2006 (with 10 percent uncertainty).

Discussion and Conclusions

The data presented above demonstrate that campsite area continues to decline in the CRE, and the objectives of the Glen Canyon Dam Adaptive Management Program with respect to recreational resources are not being met. Our results show that from 1998 to 2006 the total amount of high-elevation campsite area decreased by 56 percent. High-elevation campsite area decreased by 25 and 75 percent in critical and noncritical reaches, respectively. Critical reaches are generally narrower than noncritical reaches; the sandbars tend to be smaller, and there is less space for vegetation expansion. Even though sandbars in noncritical reaches are much larger than sandbars in critical reaches, the campable area only accounts for an average of 20 percent of the entire sandbar. Campsite area within critical reaches covers approximately 50 percent of the sandbar. In critical reaches, campsite area change was statistically correlated to changes in sandbar area, whereas in noncritical reaches, it is not. This suggests that vegetation encroachment is leading to higher rates of campsite area loss than can be attributed to erosion alone. Other factors, such as surface-water runoff, aeolian processes, and human impact, also contribute to campsite loss.

The only systemwide campsite area increase during the study period occurred between 2003 and 2005, as evidenced by the high-elevation campsite area increase of 29 percent. This temporary increase was the result of high-elevation deposition and vegetation burial during the November 2004 HFE. The continued existence of sandbars suitable for camping in this system depends on high flows to redeposit sediment lost through the natural processes of erosion and to bury, scour, or remove vegetation. Therefore, the availability of campsite area is closely linked with the frequency of flood events from Glen Canyon Dam. The results of this study suggest that high flows once every 8 years is not sufficient to restore and maintain high-elevation campsite area. Unless vegetation is physically removed, future high-flow events are the only mechanism by which sandbars used as campsites above the 25,000 ft^3/s stage elevation can be rebuilt and maintained.

Acknowledgments

This project was supported through a cooperative agreement between Northern Arizona University and the Grand Canyon Monitoring and Research Center as part of the Glen Canyon Dam Adaptive Management Program. We thank Grand Canyon National Park for research and river permits, Ruth Lambert and Helen Fairley for supporting this project, Carol Fritzinger and Jeff Behan for their assistance, and all the "Nams" and boatmen who participated in the field surveys.

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Water Use by Riparian Plants on the Lower Colorado River

By Pamela L. Nagler[1] and Edward P. Glenn[2]

Abstract

In many places along the lower Colorado River, saltcedar (*Tamarix ramosissima*) has replaced native shrubs and trees, including arrowweed (*Pluchea sericea*), mesquite (*Prosopis* spp.), cottonwood (*Populus fremontii*), and willow (*Salix gooddingii*). It has been proposed that removing saltcedar and replacing it with native species could result in substantial water savings on western U.S. rivers. We used sap-flow sensors to determine water use by saltcedar and other riparian species at six sites at Cibola National Wildlife Refuge in 2007 and 2008. We also measured leaf area index (LAI) and fractional ground cover (f_c) of saltcedar stands. Saltcedar water use varied among stands, ranging from 2.0 to 9.5 millimeters of water per square meter per day (mm d^{-1}; peak summer values) and averaged 5.7 mm d^{-1}, about one-half of the potential evapotranspiration (ET_o), determined from meteorological data at the site. LAI averaged 2.54 and f_c averaged 0.8 over the flood plain. Mesquite and arrowweed had higher water use than saltcedar. Using a remote sensing method calibrated with ground data, average water use by riparian vegetation over the whole river was 876 millimeters of water per square meter per year. Based on the acreage of riparian vegetation present along the river, we calculated that clearing all riparian vegetation would save about 2 percent of the annual river flow, and clearing saltcedar monocultures would save about 1 percent, assuming no replacement vegetation. Water savings would be less if replacement vegetation was allowed to develop on the flood plains.

Introduction

Riparian corridors account for only 1–2 percent of the land area in the Southwest but are disproportionately important for their ecosystem value and their role in the regional water budget (Poff and others, 1997). Over one-half of Southwest animal species use riparian corridors for all or part of their life cycles. Resident and migratory birds are especially dependent on riparian zones for feeding and nesting habitat (Sogge and others, 2008; van Riper and others, 2008). On the other hand, riparian vegetation consumes large amounts of water, which might otherwise be recovered for human use (Di Tomaso, 1998; Zavaleta, 2000). Therefore, resource managers must balance ecosystem needs with water demands by a growing human population in the Southwest (Shafroth and others, 2005).

The hydrology of Southwest rivers has been greatly altered by construction of dams and diversion of water for irrigation and municipal use over the past 75 years (Poff and others, 1997). Overbank flooding is now rare on regulated river stretches; their terraces have become saline, and aquifers have receded. These changes have been accompanied by the spread of an introduced, salt-tolerant shrub, saltcedar (*Tamarix ramosissima* and related species) (Gaskin and Schaal, 2002), along the rivers (Glenn and Nagler, 2005). Native riparian trees, such as cottonwood (*Populus fremontii*), willow (*Salix gooddingii*), and mesquite (*Prosopis* spp.), have decreased dramatically on many regulated rivers, and saltcedar and native salt-tolerant shrubs, such as arrowweed (*Pluchea sericea*) and quailbush (*Atriplex lentiformis*), now dominate these altered river systems (Pataki and others, 2005; Shafroth and others, 2005).

Two key science questions about saltcedar must be answered to develop adaptive management strategies for these rivers. First, how does saltcedar impact the habitat value of riparian corridors for animal species of concern; and second, how does saltcedar impact the water budget of these river stretches. Starting in the 1970s, ecologists and resource managers became increasingly concerned that saltcedar-dominated rivers provided poor wildlife habitat and that saltcedar might consume large amounts of water compared to native riparian species—water that could be used for human uses (Di Tomaso, 1998; Zavaleta, 2000). In response, saltcedar control programs have been implemented with the goals of improving habitat value and saving water, and the Salt Cedar and Russian Olive Control Demonstration Act (H.R. 2720; Public Law 109–320) has been passed by the U.S. Congress to conduct demonstration control projects for saltcedar and Russian olive (*Elaeagnus angustifolia* L.), another introduced riparian species in the Western United States.

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More recent studies have called these concerns into question. It is now recognized that saltcedar can support wildlife, especially in mixed stands with a minority of native trees and with a source of water nearby (Sogge and others, 2008; van Riper and others, 2008). Furthermore, saltcedar water use appears to be within the range of other riparian species (Nagler and others, 2004, 2005, 2008, 2009; Glenn and Nagler, 2005; Owens and Moore, 2007). However, definitive studies on these concerns are still lacking.

In this paper, we describe research conducted at Cibola National Wildlife Refuge (CNWR) on the lower Colorado River, where we measured water use by saltcedar and native plants. We used ground and remote sensing methods to estimate evapotranspiration (ET) of single plants, stands of plants, and whole river reaches (Nagler and others, 2008, 2009). Measuring ET at multiple scales is important in understanding how the physiological controls on ET at the leaf level translate into water-use characteristics of vegetation over whole river systems. We have found that saltcedar water use is low to moderate in comparison to other riparian species, and saltcedar occupies saline niches, which are now controlled by saltcedar establishment but are no longer habitable by mesic native trees.

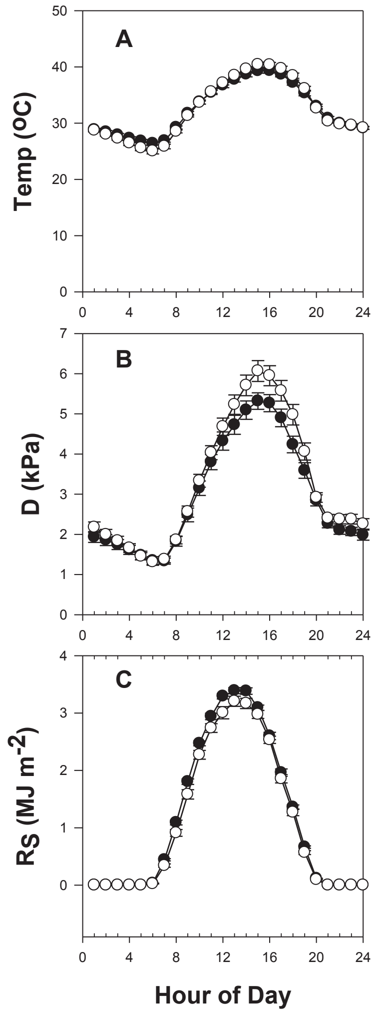

Figure 1. Diurnal patterns of (A) air temperature, (B) vapor pressure deficit, and (C) solar radiation at the Parker AZMET station near Cibola National Wildlife Refuge on the lower Colorado River during sap-flow measurements of transpiration in 2007 (closed circles) and 2008 (open circles).

Methods

Study Site. CNWR is located between Yuma, AZ, and Blythe, CA, on the lower Colorado River. Annual rainfall is less than 100 millimeters of water per square meter per year (mm yr^{-1}), occurring as occasional winter rains augmented by summer monsoon rains in July and August (Arizona Meteorological Network, 2008). The hottest month of the year is August with an average maximum daily temperature of 38 degrees Celsius (°C), and the coolest month is December with an average minimum daily temperature of 4 °C. Saltcedar is deciduous in this climate, losing leaves in November and initiating new leaves in March (growing season is about 230 days). Daily curves of air temperature, solar radiation, and vapor pressure deficit for June–August 2007 and 2008 are shown in figure 1 from data collected at the Parker, AZ, Arizona Meteorological Network (AZMET) station (Arizona Meteorological Network, 2008).

CNWR contains approximately 6,000 acres (ha) of riparian vegetation of which 4,000 ha is classified as saltcedar near-monocultures (>90 percent saltcedar), and the remainder is saltcedar with native trees including cottonwood, willow, honey mesquite (*Prosopis glandulosa*), and screwbean mesquite (*Prosopis pubescens*) or native shrubs including arrowweed, quailbush, and fourwing saltbush (*A. canescens*) (Bureau of Reclamation, 1996). The study site was on a floodplain terrace on which six plots were established at different distances from the active channel of the river (names and locations of plots are given in figure 2) (Nagler and others, 2008, 2009). Saltcedar was the dominant plant at each site, growing in dense stands interrupted by areas of light, sandy soil, with occasional arrowweed, creosote, and quailbush shrubs and stunted screwbean mesquite trees occurring in the more open areas. These sites differed in distance from the river, depth and salinity of the aquifer, soil texture, and plant density and were chosen to represent the range of conditions in which saltcedar grows at CNWR (Nagler and others, 2008, 2009).

Measuring Transpiration and Stomatal Conductance. We measured transpiration of saltcedar, mesquite, and arrowweed by heat-balance, sap-flow sensors attached to plants at the study sites. Measurements were made in the summers of 2007 and 2008 as described in detail in Glenn and others (2008) and Nagler and others (2007, 2009) Heat-balance sensors introduce a constant amount of heat into the plant through a heating wire wrapped around a branch. Transpiration is then measured by the rate at which

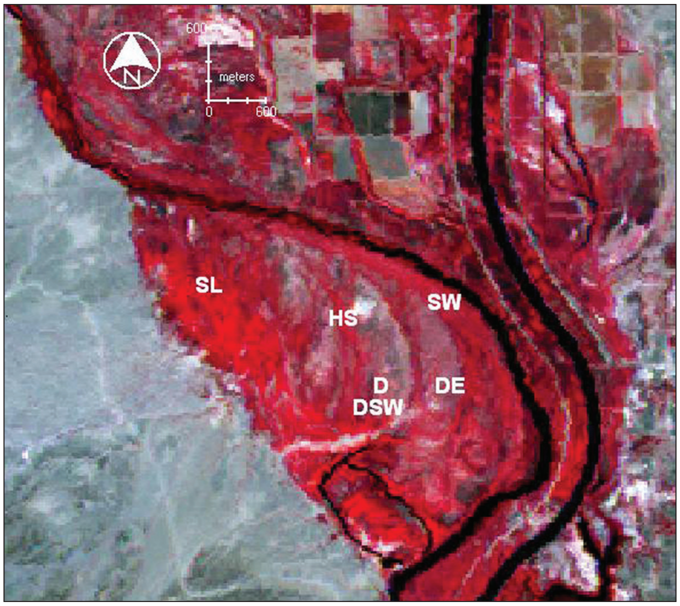

Figure 2. Approximate location of sap-flow study sites at Cibola National Wildlife Refuge on the lower Colorado River. Sites were named Slitherin (SL), Swamp (SW), Diablo Tower (D), Diablo South West (DSW), Diablo East (DE), and Hot Springs (HS). Sites are displayed on a Landsat ETM+ image. Data for SL, SW, and DE were collected in 2007 and are reported in Nagler and others (2009); data for DT, DSW, and HS were collected in 2008 and are reported here for the first time.

heat is dissipated away from the heat source by convection in the transpiration stream (Kjelgaard and others, 1997; Grime and Sinclair, 1999). Heat dissipation is measured by thermocouples placed at the heating wire, upstream and downstream from the wire, and by a thermopile placed around the insulation layer around the sensor. Heat is dissipated both by convection in the transpiration stream and by diffusion in the woody tissues around the heating wire and in the insulating material. A method is needed to calculate diffusion heat losses in the absence of transpiration, which is then subtracted from total heat dissipation to calculate transpiration. Diffusion heat losses typically are calculated using 2:00 a.m. values of heat loss, because most plants do not transpire at night, hence convection heat loss because of transpiration should be zero at 2:00 a.m. However, saltcedar can have considerable night-time transpiration (Moore and others, 2008), so we used an alternative procedure to calculate zero values for transpiration. At the end of the measurement period, typically 2–6 weeks, the gaged branches were harvested by cutting them above the point of sensor attachment, and the cut end was sealed with parafilm. After cutting the branches, sensors were allowed to collect data for an additional 3 hours to estimate diffusional heat loss in the absence of transpiration.

The surface area of the leaves is determined by weighing the leaves, then determining the specific leaf area (SLA) (square meters of leaves per gram) for a subsample of leaves (Nagler and others, 2004, 2007). Plant transpiration on a leaf-level basis (E_L), by convention, is expressed in units of millimoles of water per square meters of leaf area per hour or millimeters of water per square meter per day. Sap-flow readings were made at different sites during June–August in 2007 and 2008. In 2007, saltcedar transpiration was measured on eight plants at Slitherin from July 20 to September 2, five plants at Diablo East from June 22 to July 8, and seven plants at Swamp from June 20 to July 17. Mesquite and arrowweed transpiration was measured on 10 and 8 plants, respectively, at Diablo East from July 7 to August 2, 2007. In 2008, saltcedar transpiration was measured on 8 plants at Diablo Tower from August 8 to August 16 and on 11 plants at Diablo Southwest and 10 plants at Hot Springs from July 3 to July 18, 2008. Mesquite and arrowweed transpiration was measured on three plants each at Hot Springs from July 3 to July 18, 2008.

Scaling E_L to Whole Plants and Stands of Plants. Sap-flow sensors provide direct, real-time measurements of plant water use. Leaf-level measurements can be scaled to ground-area estimates by first determining the leaf area index (LAI) (square meters of leaf area per square meters of ground area) for the flood plain over the river reach of interest. In our study, this was accomplished by measuring plant-specific leaf area index (LAPS) of individual plants and the proportion of vegetation and bare soil (fractional cover, f_c) over the site by using high-resolution satellite and aerial imagery (Nagler and others, 2009). Then LAI was calculated as:

$$LAI = LAPS \times f_c. \quad (1)$$

Transpiration of individual plant canopies (E_C) was calculated as:

$$E_C = E_L \times LAPS. \quad (2)$$

Transpiration of stands of plants on a ground-area basis (E_G), which included the area of bare soil between plants, was calculated as:

$$E_G = E_L \times LAI. \quad (3)$$

Note that E_G is different from ET because it only includes plant transpiration, whereas ET also includes evaporation from other sources, such as bare soil after a rain event. However, given the scant rainfall at CNWR, E_G and ET are considered nearly equivalent in this study.

Optical measurements of LAI were made under several hundred plants during two summer growing seasons (2007–2008) by using a Licor 2000 leaf area index meter, which was calibrated by leaf harvesting of selected plants of each species. Measurements were concurrent with measurements of sap flow, as LAI and SLA can change during a season. Fractional cover was determined on high-resolution aerial photographs or Quickbird satellite images of the study site by using a visual, point-intercept method in which the image was overlaid with a 200-point grid, and each grid intersection was scored as 250-meter (m) plots centered on each sap-flow site.

Expressing ET as Fractional ET Based on ET_o. Sap-flow measurements typically provide measurements of E_G for a relatively short period of time (a few weeks) at a specific point in the landscape. The measurements must be scaled over longer time periods and larger land areas to be used in riparian water budgets. Temporal scaling was accomplished by calculating the ratio of actual E_G measured by sap-flow sensors to ET_o:

$$\text{ET-F} = E_G/ET_o. \tag{4}$$

ET-F typically is considered to be constant for a given crop or plant type, hence short-term measurements of E_G can be divided by ET_o to get ET-F. Meteorological data can then be used to project E_G for an annual cycle for a given plant species by multiplying annual ET_o determined at the AZMET station by E_G/ET_o determined in the field (Allen and others, 1998; Groeneveld and others, 2007). Two methods were used to calculate ET_o. The first method used the FAO-56 formula, which is based on the Penman-Monteith equation (Allen and others, 1998). $ET_{o\text{–}PM}$ is an estimate of ET from a hypothetical well-watered grass crop, and it is used as a measure of the maximum ET that can be supported in a given set of ideal meteorological conditions. $ET_{o\text{–}PM}$ values were obtained from the Parker, AZ, AZMET station (Arizona Meteorological Network, 2008). The second method used the Blaney Criddle formulation of ET_o ($ET_{o\text{–}BC}$), which is a simplified formula based on mean monthly temperature and mean daily percentage of annual daytime hours (Brouwer and Heibloem, 1986). Although $ET_{o\text{–}PM}$ is generally the preferred method for calculating ET_o (Allen and others, 1998), temperature data are much more widely available than the full meteorological data needed to calculate $ET_{o\text{–}PM}$. In Arizona, for example, there are nearly 500 cooperative National Oceanic and Atmospheric Administration (NOAA) stations reporting temperature and precipitation throughout the State, but only 27 AZMET stations reporting $ET_{o\text{–}PM}$. Hence, $ET_{o\text{–}BC}$ could be a valuable method for scaling ET over large landscape areas.

Scaling E_G Over River Stretches. Spatial scaling of E_G over large river stretches was accomplished by using remote sensing (Choudhury and others, 1994). Ground measurements of E_G were converted to ET-F using equation 4, then were regressed against values of the enhanced vegetation index (EVI) from the MODIS sensors on the Terra satellite. MODIS EVI values have a resolution of 250 m and are collected on a near-daily basis and delivered as pre-processed, 16-day composite values (Huete and others, 2002). Once relations between EVI and the biophysical variables are determined, EVI can be used to scale E_G over large river areas. In this study, we used a scaled vegetation index (VI; EVI*), in which values were scaled between 0 (representing bare soil) and 1.0 (representing maximum greenness) on the basis of a previous extensive dataset collected on western rivers (Nagler and others, 2005).

Determining EG for the Lower Colorado River. We estimated EG over the major riparian terraces on the lower Colorado River by sampling MODIS pixels to determine EVI* and AZMET data from the Mohave, Parker, and Yuma AZMET stations to determine ET_o. We sampled pixels in the following river reaches (north to south; fig. 3): Mohave, Havasu National Wildlife Refuge (HNWR), Bill Williams River at its confluence with the Colorado River, Imperial National Wildlife Refuge (INWR), Mittry Lake, and the confluence of the Colorado River with the Gila River in Yuma. The Bill Williams River delta at the Colorado River contains an extensive stand of mature cottonwoods, which were sampled during the study. All the other sample sites were dominated by saltcedar, similar to CNWR. We did not sample narrow stretches of the river because the MODIS pixels would include nonriparian land-cover classes. At each sampling site, we extracted pixels on a grid pattern using the Oak Ridge National Laboratory Distributed Active Archive

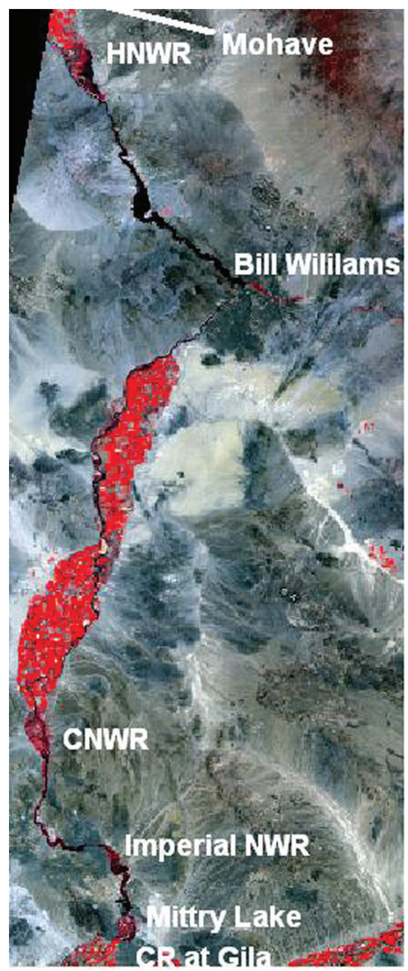

Figure 3. Location of wide-area sampling sites for estimating ET from MODIS EVI* pixels on the lower Colorado River. The Mohave site is not visible on this composite Landsat ETM+ scene, but it is just north of the irrigation district at the upper edge of the image.

Center (DAAC) site, which displays the MODIS pixels overlain on a high-resolution Quickbird image (Oak Ridge National Laboratory Distributed Active Archive Center, 2008). If a sampled pixel contained water or nonriparian landscape features (agricultural fields, desert), it was excluded. From 12 to 16 pixels were extracted per study area. We used this pixel sampling method rather than preparing a mask of the whole area of interest to ensure that only riparian landscape was measured. Water, in particular, can skew results because it has negative EVI values.

Results and Discussion

LAPS, f_c, E_L, E_C, and E_G at Individual Sites. Results for saltcedar values are summarized in figure 4. LAPS for saltcedar ranged from 2 to 4 among sites, and f_c ranged from 0.54 to 0.95 (fig. 4A, B). The Slitherin site had the highest LAPS and f_c. On the other hand, the Hot Springs site had notably lower f_c than the other sites. This site is adjacent to a bare area where geothermal water (about 50 °C) approaches the soil surface, and the plants at this site likely were negatively affected by the water source. Numerous dead plants occurred at this site. E_L ranged from 1.0 to 2.9 millimeters of water per square meter per day (mm d^{-1}) among the sites, with the lowest value occurring at Hot Springs. E_G was highest at Slitherin (9.5 mm d^{-1}) and lowest at Hot Springs (1 mm d^{-1}), spanning nearly a 10-fold range of values. Clearly, saltcedar water use is not uniform over CNWR.

Results for mesquite and arrowweed values are shown in figure 5. These plants grew as isolated plants within larger saltcedar stands, hence it was not possible to calculate LAI or E_G. LAPS ranged from 1.3 to 2.6 for mesquite, 1.6 to 2.0 for arrowweed, and 1.5 for creosote at the one site where it occurred. In general these plants had lower LAPS than saltcedar, though the ranges overlapped. On the other hand, E_L ranged from 2.8 to 11.5 mm d^{-1}, much higher than saltcedar values. These plants were all surrounded by bare soil and were illuminated from all sides, which presumably resulted in higher transpiration rates on a leaf-level basis than saltcedar growing in closed or nearly closed canopies.

Figure 6 shows canopy-level rates of transpiration among sites and species. Saltcedar values ranged from 1.5 to 10.0 mm d^{-1}, whereas possible replacement species

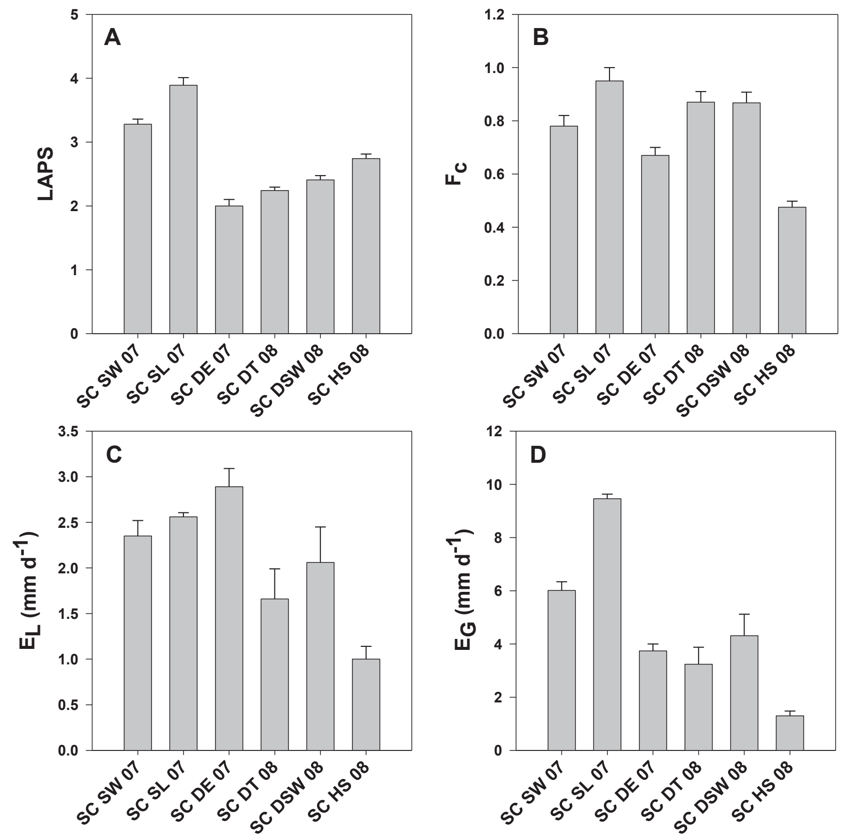

Figure 4. Values of saltcedar (SC) plant-specific (A) leaf area index (LAPS); (B) fractional cover (f_c); and (C) transpiration on a leaf-level (E_L) and (D) ground-level (E_G) basis. The x-axis shows the site (SL = Slitherin; SW = Swamp; DT = Diablo Tower; DSW = Diablo Southwest; DE = Diablo East; HS = Hot Springs) and year (2007 or 2008) when measurements were made. Error bars are standard errors of means.

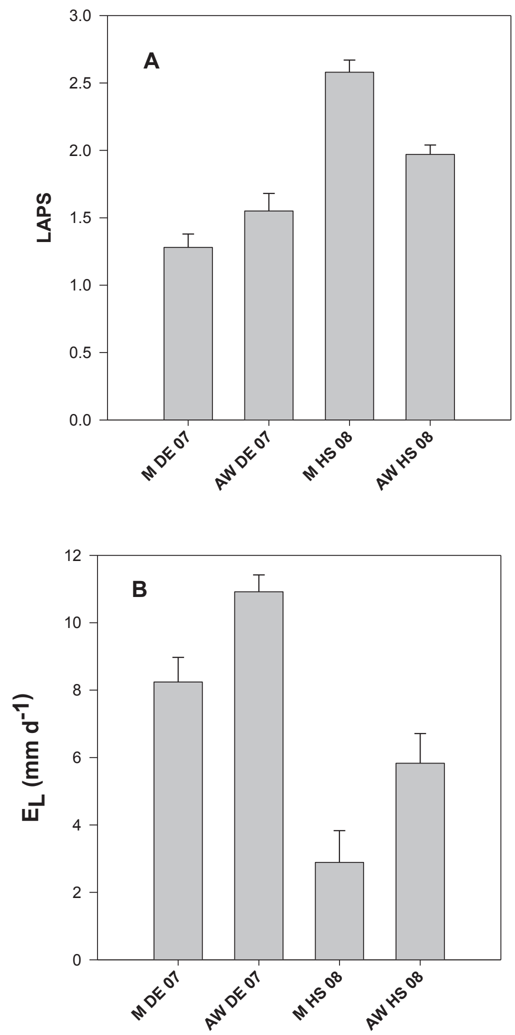

Figure 5. Plant-specific (A) leaf area index (LAPS) and (B) leaf-level transpiration (E_L) of mesquite (M) and arrowweed (AW) at different sites (HS = Hot Springs, DE = Diablo East) at Cibola National Wildlife Refuge in 2007 and 2008. Error bars are standard errors of the means.

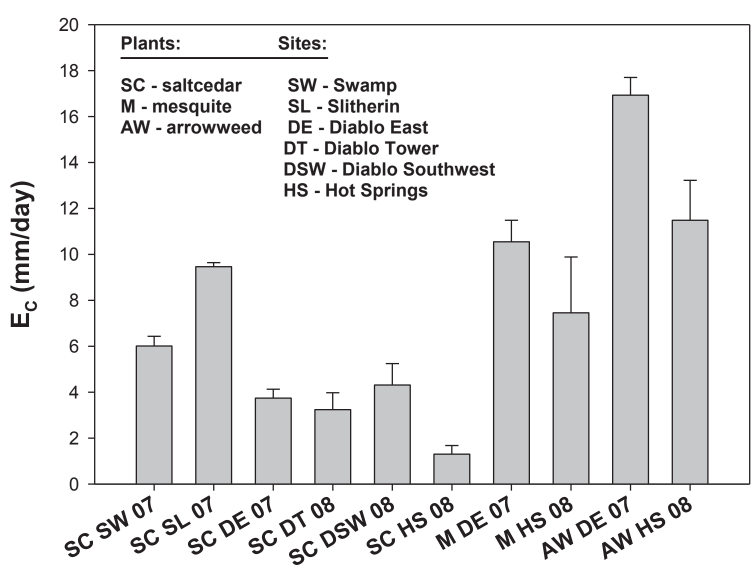

Figure 6. Canopy-level transpiration rates (E_C) of saltcedar (SC), mesquite (M), and arrowweed (AW) at different sites (SL = Slitherin; SW = Swamp; DT = Diablo Tower; DSW = Diablo Southwest; DE = Diablo East; HS = Hot Springs) at Cibola National Wildlife Refuge in 2007 and 2008.

ranged from 5.5 to 16.0 mm d^{-1}. All the species showed high variability among sites, but saltcedar clearly did not have higher E_C than possible replacement species at CNWR. Over wider areas, saltcedar could have higher E_G than mesquite or arrowweed owing to differences in plant spacing. We were not able to resolve this variability at CNWR because mesquite and arrowweed intergrew with saltcedar at CNWR, so E_G could not be determined for these plants.

Scaling E_G/ET_o by MODIS EVI*. ET-F was plotted against EVI* for saltcedar at CNWR and for other plants on the lower Colorado River that were determined in other studies (fig. 7). Alfalfa ET was measured on three occasions at a control alfalfa field (Hay Day Farms, Blythe, CA) near the river by using a neutron hydroprobe to measure water depletion in the root zone following irrigation events (Hay Day Farms, unpub. data, 2007–2008). Soil moisture was measured at 0.3-m intervals from 0.3- to 2.0-m depths above the water table 2 days and 8 days after irrigation of the field to calculate ET by the difference in soil moisture content at the two dates. Cottonwood E_G was measured in a planted field near the river by using sap-flow sensors (Nagler and others, 2007). Saltcedar and arrowweed ET at Havasu National Wildlife Refuge were measured using Bowen Ratio moisture flux towers in 2005 and 2006 (Nagler and others, 2005; Westenberg and others, 2006). ET-F by saltcedar at the Hot Springs site is plotted in figure 7, but was not included in the regression analyses because the high-temperature water clearly affected the plants at this site, creating aberrant growth conditions.

Linear regression equations were significant for both ET_{o-BC} and ET_{o-PM} ($P < 0.01$), but y-intercepts were small and nonsignificant ($P = 0.69$ and 0.84, respectively). This is expected because the scaling procedure sets EVI* for bare soil at 0. Therefore, regression equations were passed through the origin to determine the final algorithms for scaling E_G or ET. ET_{o-BC} (fig. 7A) clearly gave a better fit of data than ET_{o-PM} (fig. 7B). The standard error of the mean increased with increasing ET-F, as expected for regression through the origin. At ET-F = 1.0, the error around the mean for the expression using ET_{o-BC} was about 20 percent, compared to 25 percent for ET_{o-PM}.

Extrapolating E_G Over the Lower Colorado River. We used the regression equation in figure 7A to extrapolate E_G from EVI* over the whole river (table 1). Mean ET for the Hay Day Farms field was 2,082 mm yr^{-1} (excluding 2005 when the field was replanted), 1.11 times higher than the mean ET_o of 1,873 mm yr^{-1} calculated by AZMET. This is expected, because alfalfa ET typically is higher than ET_o for a grass reference crop used to calculate ET_o (Hunsaker and others, 2002). Mean ET at the riparian sites was 816 mm yr^{-1}, with E_G/ET_o equal to 0.44. ET of cottonwood at the Bill Williams river delta was 1,105 mm yr^{-1}, higher than ET at any of the saltcedar sites, which ranged from 434 to 1,057 mm yr^{-1}).

Comparison with Other Studies. Values of LAI, ET, and salt-tolerance limits of saltcedar and possible replacement species are given in table 2. Smith and others (1998)

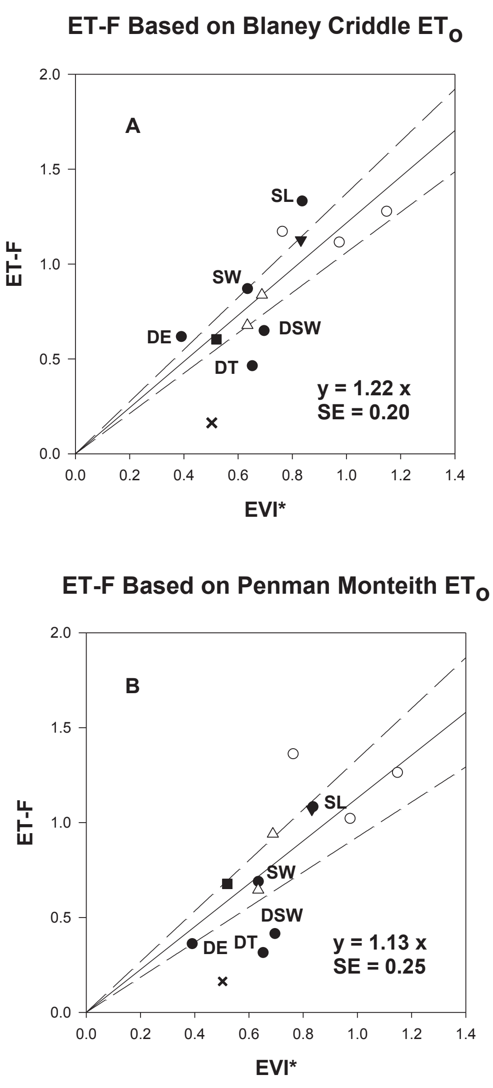

Figure 7. Ratio of actual ET to reference crop ET (ET-F) for plants on the lower Colorado River. Plants and locations are: saltcedar at Slitherin (SL), Swamp (SW), Diablo East (DE), Diablo Southwest (DSW), and Diablo Tower (DT) at Cibola National Wildlife Refuge (closed circles); saltcedar at Hot Springs at Cibola National Wildlife Refuge (cross); saltcedar at Havasu National Wildlife Refuge in 2002 and 2003 (open triangles); arrowweed at Havasu National Wildlife Refuge in 2003 (closed square); and alfalfa at Palo Verde Irrigation District on three dates. Data are from the present study and previous studies (cited in the text). Hot Springs was not included in the regression analyses. Regression equations were passed through the origin, and dashed lines denote 95-percent confidence intervals. ET_o was calculated by the (A) Blaney Criddle method and the (B) Penman Monteith method.

Table 1. Annual transpiration (mm yr $^{-1}$) at wide-area sites along the lower Colorado River determined by the crop coefficient method, in which sap flux measurements were regressed against scaled EVI at Cibola National Wildlife Refuge. At each site, 10–16 MODIS pixels were selected in a grid pattern to represent the area of interest. We used this method rather than a mask approach, because the areas contain open water which interferes with wide-area ET estimates. By selecting individual pixels based on a Quickbird image from Oak Ridge National Laboratory, we were able to exclude water pixels, which had negative EVI values.

Site	2000	2001	2002	2003	2004	2005	2006	2007	2008	Mean
Mohave ET_o	2,075	1,908	1,968	1,745	1,853	1,693	1,843	1,978	1,805	1,874
Parker ET_o	2,183	2,030	2,028	1,858	1,900	1,920	1,988	2,075	1,945	2,183
Yuma ET_o	1,952	1,753	1,950	1,815	1,830	1,768	1,908	1,020	1,788	1,952
Hay Day Alfalfa	2,178	2,102	1,724	1,419	1,651	833*	2,581	2,510	2,612	2,082
Havasu	863	770	775	687	674	616	658	774	724	727
Mohave	347	431	401	408	410	558	444	457	458	434
Bill Williams	1,532	1,159	1,230	962	935	850	1,020	1,148	1,111	1,105
Cibola	1,117	989	873	836	818	893	709	638	699	841
Mittry	1,033	837	873	804	738	626	793	761	752	802
Imperial	1,245	1,047	1,084	1,018	955	931	1,070	1,072	1,091	1,057
LCR-Gila	1,000	824	883	840	770	683	592	515	602	745
Mean riparian	1,019	865	874	794	757	736	755	766	776	876

*Field was replanted in 2005—omitted from mean value.

speculated that saltcedar might have higher LAI than other riparian species, leading to higher rates of water use. However, based on the studies presented in table 2, saltcedar LAI and EG are within the range of other species. The values differed considerably among and within studies and were more closely related to local site conditions than to the species measured. The main difference between saltcedar and possible replacements species was in their degree of salt tolerance, which was much higher for saltcedar than mesquite, arrowweed, or cottonwood.

Implications for Management

As on other regulated arid zone rivers (Jolly and others, 2008), the aquifer and flood-plain soils have become salinized at CNWR, with groundwater salinities ranging from 2,000 milligrams per liter (mg l $^{-1}$) near the river to >10,000 mg l $^{-1}$ away from the river (Nagler and others, 2008, 2009). Results are similar at other locations on the river and at other regulated river reaches in the Western United States (Glenn and Nagler, 2005). Hence, saltcedar at CNWR now occupies niches that are no longer available to mesic trees, such as cottonwood, and are only marginally habitable by mesquites. This study produced no evidence that saltcedar has unusually high water use compared to native plants. Over the lower Colorado River, saltcedar monocultures cover 18,200 ha, and total riparian vegetation covers 34,000 ha (Bureau of Reclamation, 1996). Based on an annual water use of 876 mm yr $^{-1}$ (table 2), consumptive water use is 158,776,000 cubic meters per year (m^3 yr $^{-1}$; 128,772 acre-feet) for saltcedar monoculture and 296,645,000 m^3 yr $^{-1}$ (240,588 acre-feet) for all riparian vegetation. Although these are large amounts of water, they represent less than 1 percent of the annual flow in the river for saltcedar monocultures and less than 2 percent for all riparian vegetation. These volumes could only be salvaged if saltcedar plants or all vegetation were removed and no replacements plants were allowed to grow back. However, maintaining bare riverbanks would lead to severe erosion problems, and this study shows that replacement vegetation would likely have equal or higher rates of water use as saltcedar.

Table 2. Leaf area index (LAI), evapotranspiration (ET), and the salinity that produces half-maximal growth for selected species on Western U.S. rivers. Literature values were selected to represent the range of conditions reported on different river systems, including both stressed and unstressed plants.

	Saltcedar	Mesquite	Arrowweed	Cottonwood
LAI	2.8[a]	1.9[e]	3.7[a]	3.5[a]
	1.5–3.3[b]	1.5[c]	1.6[c]	3.1–3.8[g]
	2.0–3.9[c]	1.9–2.4[f]		2.5–3.5[h]
	0.9–4.1[d]			1.75–2.75[i]
ET (mm d^{-1})	5.3–11.5[b]	5.6[l]	6.0[m]	6–12[g]
	2.0–9.5[c]	7.5–8.2[c]	8.5–16.9[c]	8–9[j]
	6.0–9.0[j]			4.8–9.3[h]
	6–10[k]			3.1–5.7[i]
Salt tolerance (mg l^{-1} total dissolved solids)	35,000[n]	6,000–12,000[o]	16,000[n]	5,000[n]
				2,000–5,000[p]

[a] Mean of values at eight sites on the lower Colorado River (Nagler and others, 2004).

[b] Range for salt-stressed and unstressed plants on a tributary of the lower Colorado River (Sala and others, 1996).

[c] This study.

[d] Range for plants on the Middle Rio Grande, New Mexico (Cleverly and others, 2002, 2006).

[e] *Prosopis velutina* in a Sonoran Desert riparian corridor (Stromberg and others, 1993).

[f] Savanna mesquites (Ansley and others, 2002).

[g] Range for water-stressed and unstressed, irrigated plots (Nagler and others, 2007).

[h] Salt-stressed and unstressed plants on the lower Colorado River (Pataki and others, 2005).

[i] Range for water-stressed and unstressed plants on Upper San Pedro River (Gazal and others, 2006).

[j] Range on the Middle Rio Grande (Cleverly and others, 2006).

[k] Range for unirrigated and irrigated on the Virgin River, Nevada (Devitt and others, 1997, 1998).

[l] Woodland and shrubland mesquites on the Upper San Pedro, Arizona (Nagler and others, 2005; Scott and others, 2008).

[m] Dense stands on the lower Colorado River (Westenberg and others, 2006).

[n] Greenhouse salt-gradient study (Glenn and others, 1998).

[o] Greenhouse study salt-gradient study (Felker and others, 1981).

[p] Range of salinities in an aquifer producing half-maximal ET of trees on lower Colorado River (Pataki and others, 2005).

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Causes, Management, and the Future of Exotic Riparian Plant Invasion in Canyon de Chelly National Monument, Arizona

By Lindsay V. Reynolds[1] and David J. Cooper[1]

Abstract

In the Southwestern United States, two exotic plant invaders of riparian habitats are tamarisk (*Tamarix ramosissima* Ledebour, *T. chinensis* Loureiro, and their hybrids) and Russian olive (*Elaeagnus angustifolia* L.). These plants were introduced by humans throughout the Southwest around 1900, and their success spreading across the region has coincided with human land-management activities such as river regulation. Both tamarisk and Russian olive have invaded Canyon de Chelly National Monument in Arizona. We addressed three broad research topics: the history of invasion, seedling establishment requirements, and the effectiveness of exotic plant removals. Our results indicate that the majority of tamarisk and Russian olives established in the mid to late 1980s, long after the original plantings and dam construction in Canyon de Chelly. This suggests that exotic plant invasion is most closely tied to precipitation and available seedling habitat, rather than river regulation or purposeful plantings. We also found that Russian olive can establish in shaded sites where seedlings do not have access to the water table and where tamarisk and native riparian plant species cannot establish. In sites where tamarisk and Russian olive were removed, native plants are most successful following cut-stump treatments where soil disturbance was minimized. Russian olive will likely continue to increase in dominance in this region while tamarisk decreases, except where cut-stump removals are successfully implemented.

Introduction

The ecological, economic, and social impacts of invasive plant species on the integrity of native communities have stimulated broad concern among researchers, land managers, and the general public. Invasive plants often exclude native plants, threaten biodiversity, and alter physical and ecological processes (Simberloff, 2005). Riparian areas have been invaded by exotic plants disproportionately more than other habitats world wide (Hood and Naiman, 2000; Friedman and others, 2005). Riparian areas are ecologically important because they support high biodiversity despite covering a small percentage of the landscape (Stohlgren and others, 1998; Richardson and others, 2007). In the Southwestern United States the two most abundant invaders of riparian habitats are the exotic woody plant species tamarisk (*Tamarix ramosissima* Ledebour, *T. chinensis* Loureiro, and their hybrids) and Russian olive (*Elaeagnus angustifolia* L.) (Friedman and others, 2005). These species were introduced in many river systems and have spread naturally through the Southwest, including into protected areas such as national parks and monuments (Graf, 1978; Katz and Shafroth, 2003).

In addition to introducing exotic plants, humans have dramatically altered the flow regime of rivers throughout the Southwestern United States with dams and flow diversion structures (Poff and others, 1997; Graf, 1999). Dams are reported to have facilitated exotic plant establishment (Shafroth and others, 2002; Stromberg and others, 2007). Annual precipitation is highly variable in the Southwestern United States and directly influences flood events of southwestern rivers (Hereford and Webb, 1992; Woodhouse and others, 2006). The pattern and timing of precipitation and flow events influence riparian species distribution on southwestern flood plains (Stromberg, 1997; Levine and Stromberg, 2001). Precipitation patterns, in addition to plantings and dams, likely influence the spread of exotic plants in the Southwestern United States (Katz and others, 2005; Birken and Cooper, 2006).

Historically, southwestern flood plains lacked trees or were populated by stands of native cottonwood trees (*Populus deltoides* spp. *wislizeni*) and willows (*Salix* spp.). Life-history differences between tamarisk, Russian olive, and native plants have allowed the exotic plants to increase along southwestern flood plains (Stromberg, 1997; Cooper and others, 2003; Rood and others, 2003). For example, the seeds of cottonwood and willow species disperse aerially in late spring and early

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summer to coincide with peak river run off, are viable for 4 to 6 weeks, and require bare, moist substrate for germination (Cooper and others, 1999). Tamarisk and Russian olive seeds typically require similar post-flooded substrate for germination, but tamarisk stands have much higher densities of seed rain than cottonwood or willow, and the period of seed dispersal extends from early summer through fall (L.V. Reynolds, unpub. data, 2007; Cooper and others, 1999). Russian olive seeds mature in the fall, overwinter on trees, disperse in spring, and are viable for up to 3 years (Katz and Shafroth, 2003). Established tamarisk and Russian olive plants can tolerate long periods without available groundwater, whereas native cottonwood and willow cannot (Brotherson and Winkel, 1986; Katz and Shafroth, 2003).

Both tamarisk and Russian olive were purposefully introduced into Canyon de Chelly National Monument (Canyon de Chelly) in Arizona by the U.S. Soil Conservation Service beginning in 1934 to stabilize erosion around cliff dwellings and Navajo agricultural fields (U.S. Soil Conservation Service, 1934). These invasive species have spread throughout the canyons, and the vegetation and stream channel have changed. The historic streambed in Canyon de Chelly was a wide, open, braided channel. Today the streambed is channelized and deeply incised in most of the monument (Rink, 2003; Cadol, 2007). This stream downcutting has greatly lowered the riparian water table, making irrigation and traditional farming practices of the Navajo residents nearly impossible. Dramatic changes in the stream channel and riparian vegetation and the effects on the Navajo residents and visitors have led the National Park Service (NPS) to initiate a study to determine the causes of exotic plant invasion and channel incision and to identify and implement management solutions.

Tamarisk and Russian olive plantings, dam construction, and precipitation variability all occurred in the recent history of Canyon de Chelly. The primary causes of tamarisk and Russian olive invasion, however, are unknown. The first goal of this study was to investigate the history of invasion by testing whether plantings, dams, or precipitation was the primary trigger of exotic plant invasion into Canyon de Chelly.

Our second goal was to test the seedling requirements of tamarisk, Russian olive, and the native cottonwood tree. Tamarisk and cottonwood provide shaded habitat suitable for Russian olive establishment but unsuitable for cottonwood, willow, or tamarisk seedlings (Sher and others, 2000; Sher and others, 2002). In addition, Russian olive can potentially establish in habitats too dry for either tamarisk or cottonwood seedlings. One study investigated the establishment requirements of tamarisk and cottonwood seedlings and showed that under ideal conditions cottonwood can out-compete tamarisk (Sher and others, 2002). Other studies found that Russian olive seedlings were more successful than cottonwood seedlings in shadier environments (Shafroth and others, 1995; Lesica and Miles, 1999; Katz and others, 2001). However, there have been no comparisons of the seedling requirements of tamarisk, Russian olive, and cottonwood simultaneously in a controlled experiment. We asked the following question: Can Russian olive establish in shadier and drier environments than both tamarisk and cottonwood? We used a controlled experiment to identify the flood-plain habitats where these species can establish. Researchers have suggested that tamarisk invasion in the Southwestern United States has nearly ended because it has filled most suitable flood-plain habitats (Friedman and others, 2005). However, tamarisk and cottonwood stands may provide ideal habitat for an ongoing Russian olive invasion.

Our final and ongoing goal is to compare two removal methods of tamarisk and Russian olive. Canyon de Chelly National Monument is implementing a large-scale tamarisk and Russian olive removal project. To test the effectiveness of invasive plant removal techniques and their influence on future plant community development, the treatments are being conducted in an experimental design framework. We are comparing cut-stump with herbicide application and whole-plant removal methods to assess the effectiveness of these removals and the subsequent recovery of the native plant community. Efforts to control exotic riparian plants have been implemented in many areas of the Southwest (Shafroth and others, 2008). Most efforts have targeted tamarisk-infested stands, and little documentation of Russian olive control efforts exists (Harms and Hiebert, 2006). The general goals of tamarisk control include restoring native plant communities, increasing water yield in rivers, and improving riparian habitat for wildlife (Shafroth and others, 2005). Scientists and managers disagree on the success of different control strategies for meeting restoration goals (Shafroth and others, 2005; Harms and Hiebert, 2006). A key problem is the lack of post-treatment monitoring, which limits our understanding of plant community response to the treatments. In our third research goal, we are addressing two questions related to exotic plant removal: (1) What are the effects of different removal methods on the future riparian vegetation composition, and (2) what physical conditions facilitate the restoration of native plant species instead of exotic plant species?

The aim of this research is to study the patterns, processes, and causes of exotic plant invasion into Canyon de Chelly National Monument. We address three broad subjects: the history of invasion, exotic and native seedling requirements, and the effectiveness of exotic plant removal methods. We hope to inform both theory and management. Our results address the process and mechanisms of exotic plant invasion. We describe Russian olive ecology in more detail than has been previously attempted and outline the ongoing threat of Russian olive invasion into southwestern riparian habitats. Finally, we address the management issue of exotic plant removal along southwestern rivers and attempt to determine effective removal methods.

Study Site

Canyon de Chelly National Monument is located in northeastern Arizona within the Navajo Indian Reservation. The monument includes two main canyons: Canyon de Chelly to the south and Canyon del Muerto to the north, both of which drain the western side of the Chuska Mountains. The two canyons meet 8.5 kilometers (km) east of Chinle, AZ, forming Chinle Wash, which is tributary to the San Juan River. Chinle receives an average of 33 centimeters (cm; 9 inches) of rain per year produced largely by late summer monsoon rains. The area receives an average of 30.5 cm (12 inches) of snow each winter.

Chinle Wash is an ephemeral stream with a bimodal flow pattern. Discharge peaks occur in spring driven by mountain snowmelt and in late summer driven by monsoon rains. Within the monument, our study area included the lower 25 km of Canyon de Chelly, the lower 17 km of Canyon del Muerto, and the first 10 km of Chinle Wash (fig. 1.)

The U.S. Soil Conservation Service began planting tamarisk and Russian olive in Canyon de Chelly in 1934 to protect ancient Puebloan ruins and modern farms from river-bank erosion (Rink, 2003; Cadol, 2007). Tamarisk and Russian olive now dominate the study area riparian vegetation. Historically, streambeds in Canyon de Chelly, Canyon del Muerto, and Chinle Wash were wide, shallow, and braided, and Chinle Wash remains that way today. However, in the two upper canyons, the stream has incised 1–5 meters (m) over the last 50 years.

Methods and Results

Methods: History of Invasion

We sampled tamarisk and Russian olive plants in four study sites in Canyon de Chelly along transects established for geomorphic research purposes. Transects were perpendicular to the stream channel and were spaced systematically every 50 m within each of four 10-acre tamarisk and Russian olive removal areas. We subjectively selected one transect for plant-aging purposes in each study site; transects were selected on the basis of backhoe availability and accessibility. All exotic plants within 3 m of each transect were excavated by using a backhoe and hand shovels (N = 58 Russian olive, 72 tamarisk). Elevation of each plant along transects was determined by surveying. Extracted plants were dried, cross sectioned with a chainsaw, and sanded. The germination point

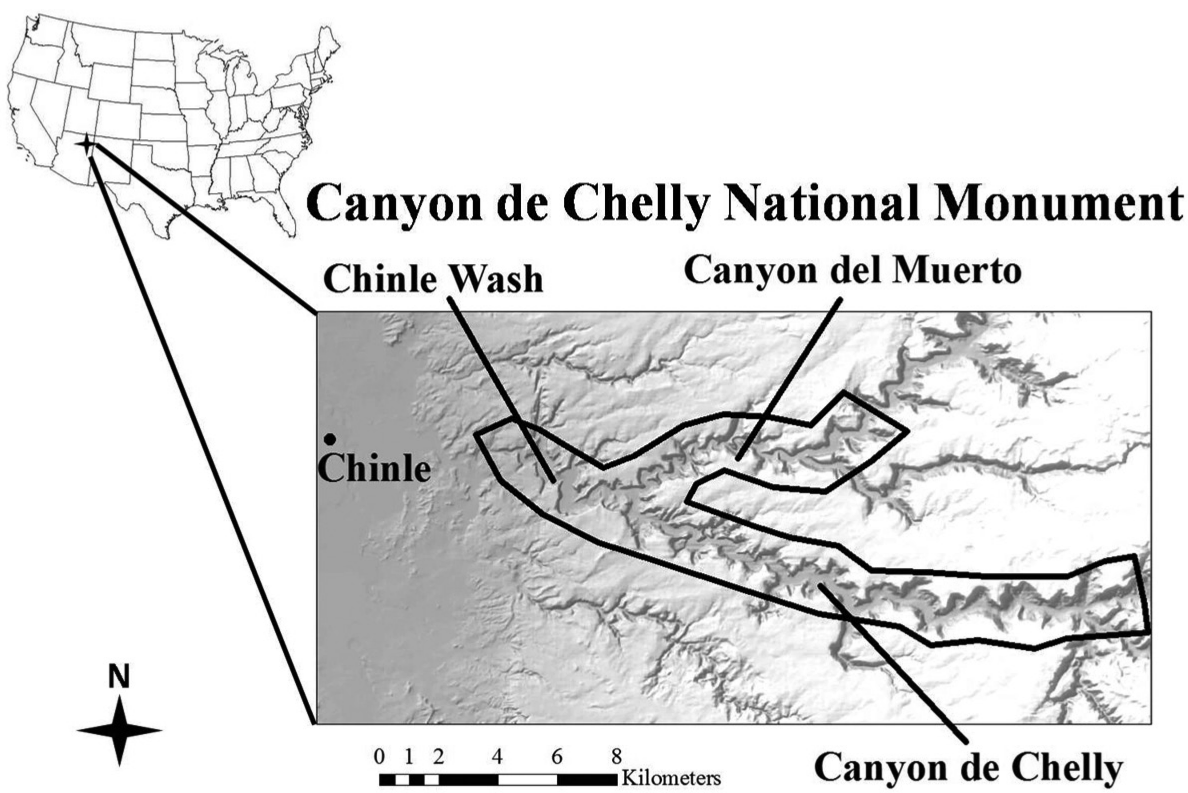

Figure 1. Canyon de Chelly National Monument. Our study area within Canyon de Chelly was limited to riparian areas within the canyon and is outlined in bold on the map.

was identified as the point where the pith originates. The depth below ground surface (and elevation) of the germination point was determined by analysis of the plant cross sections and topographic survey data. Plant cross sections were analyzed by using a precision binocular microscope to count annual growth rings. Methods for dating tamarisk and cottonwood and identifying germination points are based on Cooper and others (2003) and Birken and Cooper (2006).

To understand the effect of climate on establishment patterns of riparian trees, it is best to use river stage records for the study area of interest. However, there are no long-term records of river stage for Chinle Wash in Canyon de Chelly. A U.S. Geological Survey streamgage at the mouth of Canyon de Chelly was in operation from November 1999 through July 2006. We attempted to recreate river stage patterns by using local precipitation data. The closest weather stations to Canyon de Chelly are in Chinle and Lukachukai, AZ. The Chinle weather station is located at the mouth of the canyon system, and Lukachukai is at the base of the Chuska Mountains, which drain into Canyon de Chelly. We used a regression model to analyze the pair-wise relations between Chinle precipitation, Lukachukai precipitation, and stream discharge of Chinle Wash at Chinle between 1999 and 2006. Based on low R^2 values, we found no relation between precipitation in Chinle and Lukachukai ($R^2 = 0.004$, $F = 9.70_{2474}$, $P = 0.0019$), precipitation and stream discharge in Chinle ($R^2 = 0.023$, $F = 58.6_{2473}$, $P < 0.0001$), and precipitation and stream discharge in Lukachukai ($R^2 = 0.0004$, $F = 0.112_{2473}$, $P = 0.7382$). The lack of data from both Chinle and Lukachukai between 1951 and 2007 may account for the poor statistical relation between stations.

Because of the sporadic nature of the local data and because a relation between streamflows and precipitation could not be established, we turned to regional records of wet and dry periods in northeastern Arizona as a proxy for streamflow. We used divisional climate data for Arizona from the National Climate Data Center. The northeastern Arizona division (AZ, division 02) includes 114 stations in operation between 1930 to 2007. Currently, there are 50 active stations. The numbers of stations changed over time as some became operational or were terminated. Divisional data are compiled from all precipitation gages in a climate division region (Guttman and Quayle, 1996). All stations within the northeastern Arizona division were averaged for each month of the record (1895–2006). Before 1931, monthly averages were calculated from regression equations developed from State averages and station averages 1931–1986 (Guttman and Quayle, 1996).

To test the effect of annual rain on plant establishment, water year[2] precipitation was calculated by summing precipitation for the months October through September for all years in which we had establishment of tamarisk and Russian olive in our study sites (1966–1998). We used a multiple regression model with Poisson errors to estimate the relation between plant establishment and precipitation in the year of establishment, precipitation in the previous year, and precipitation in the following year.

[2] Water year is the period October 1 to September 30 and is defined by the year in which the period ends.

Results: History of Invasion

Annual precipitation showed significant relations with establishment. Russian olive establishment in a given year was positively related to annual rainfall that year ($F = 9.72$, $P = 0.001$) as well as the previous year's precipitation and the following year's precipitation ($F = 7.77$, $P = 0.005$ and $F = 8.13$, $P = 0.004$). Tamarisk establishment in a given year was positively related to annual rainfall in the year of establishment ($F = 2.632$, $P = 0.008$) and the previous year's precipitation ($F = 4.32$, $P < 0.001$). The majority of tamarisk and Russian olive plants in our plots established in the mid to late 1980s. Based on the positive relation between establishment and rainfall, the pulse of establishment in the 1980s appears to be related to consecutive high rain years in the 1980s (fig. 2). The oldest tree we found in our study sites dated to 1966, which is surprising since plantings started in Canyon de Chelly in the 1930s. One explanation is that large-scale flooding in the 1980s caused mortality of older trees, simultaneously creating conditions for tamarisk and Russian olive seedling establishment. A more plausible explanation is that the invasion was slow and dispersed until the 1980s, when favorable conditions facilitated widespread invasion. This second explanation is supported by an aerial photograph analysis of Canyon de Chelly where photographs from the 1930s through 2004 were analyzed for riparian vegetation cover. Vegetation cover slowly increased between the 1930s and the 1970s, and then between the 1970s and 2004 a dramatic increase in riparian vegetation cover took place (Cadol, 2007).

Methods: Seedling Survival

We compared seedling establishment requirements for tamarisk, Russian olive, and cottonwood in a controlled experiment with four water treatments (shallow water table, low, average, and high monsoon rain), split into three shade treatments (full sun, partial shade, and full shade). Each water/shade treatment consisted of one plot with 12 replicates of each species (cottonwood, tamarisk, and Russian olive) randomly distributed within the plot. Seeds were collected in May and June 2007. Tamarisk and cottonwood seeds are germinable when they disperse in early summer. Russian olive seeds ripen late in the summer and require scarification during freezing winter temperatures. We collected Russian olive seeds from the 2006 crop that over-wintered on trees. We germinated seeds of all species under saturated soil conditions and allowed the seedlings to grow for 4 weeks before transplanting them into treatment conditions. Seedlings were grown individually in 5-cm x 5-cm x 25-cm tubes. Sandy soil

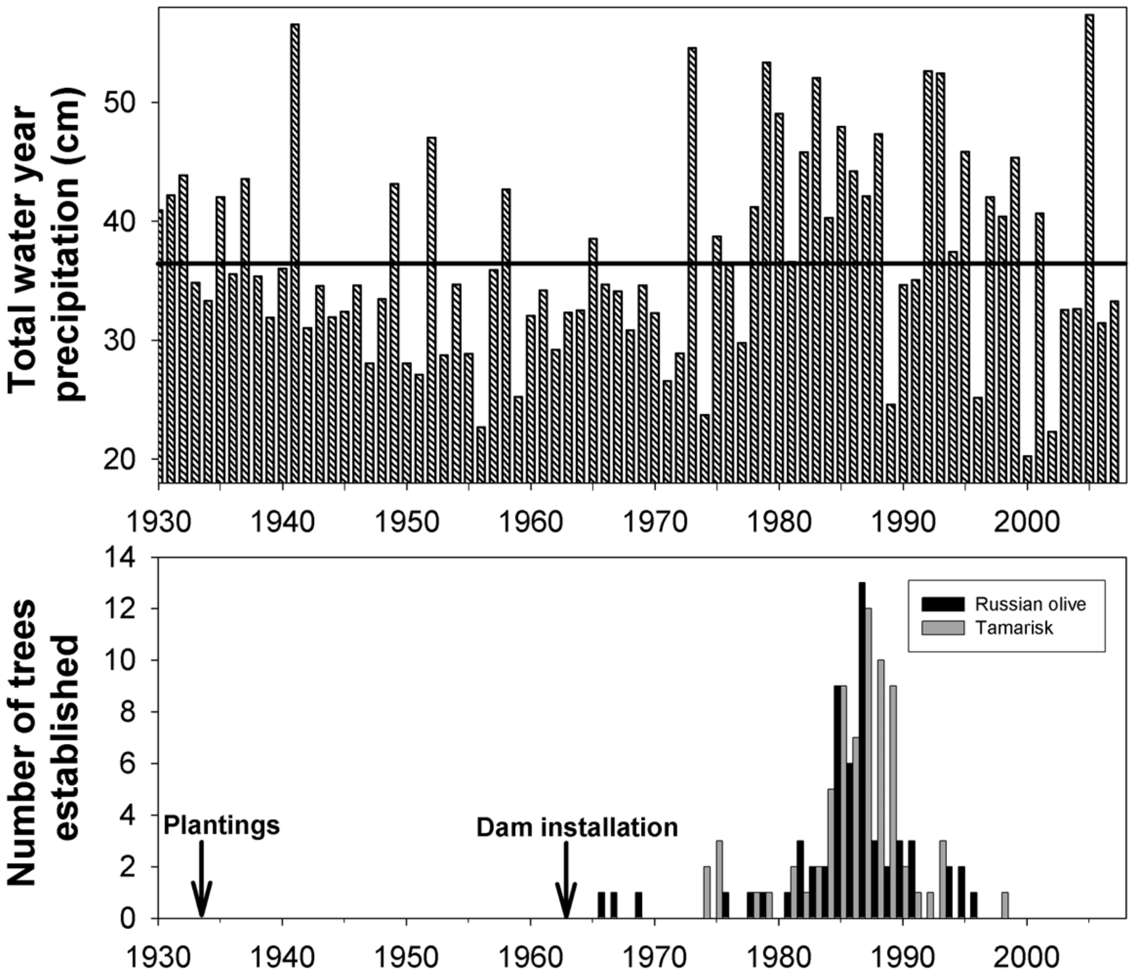

Figure 2. Total water year (October–September) precipitation (cm) for years 1930–2007 (top panel). The bold horizontal line indicates the 100-year average for total water year precipitation (37 cm/year). The bottom panel indicates the number of trees established in our study sites in each year.

collected from Chinle Wash was placed into each tube as a growing medium. Soils used in the experiment had a grain size distribution of 94 percent sand, 2 percent silt, 1.6 percent clay, and 1.5 percent gravel by dry weight. Each plot contained 36 tubes with one seedling in each tube. All treatments were located together in an outside fenced environment in Chinle, AZ.

We measured photosynthetically active radiation (PAR, micromoles (µmol)) in the field under dense stands of tamarisk and cottonwood and under full sun, and simulated these light levels in the shade treatments by using shade cloth (approximately 99 percent, 90 percent, and 0 percent shade). Shallow water table plots were placed in bins with a water level up to 10 cm below the soil surface to maintain saturation and simulate flooded conditions. In the rain simulation treatments, seedlings were watered from the top. Rain amounts mimicked amounts that occur during a low, average, and high monsoon year on the basis of precipitation data from the Chinle, AZ, rain gage, which has been operating since 1951. Water was applied using a drip hose irrigation system; water amounts were measured for quantity and uniformity by using cups spaced evenly along the hoses. We measured seedling survival and height (in millimeters (mm)) for each plant once per week for 10 weeks from July to September 2007.

Logistic regression was used to analyze the effects of shade, water, and species identity on seedling survival. Very few tamarisk and cottonwood plants survived the low water and low light treatments. Therefore, we analyzed each species separately using two-way logistic regression models to test the effects of shade and water on seedling survival. We tested the difference in growth rates (mm/week) between species across treatments by using an analysis of variance on log-transformed growth rates of surviving plants. We conducted a survival analysis using the Cox proportional hazards model to test the differences between treatments and species on time-to-death (weeks) of the seedlings.

Results: Seedling Survival Experiment

Russian olive grew faster than both cottonwood and tamarisk in nearly all treatments. Mean Russian olive growth rate exceeded that of tamarisk and cottonwood ($F = 163.56$, $P < 0.0001$ and $F = 59.96$, $P < 0.0001$), and cottonwood growth rate exceeded tamarisk ($F = 25.46$, $P < 0.0001$).

Russian olive seedling survival rate exceeded that of tamarisk and cottonwood in all treatment combinations except the shallow water table-90 percent shade treatment where 100 percent of Russian olive and cottonwood seedlings survived. The cottonwood seedling survival rate exceeded tamarisk in all treatments. Tamarisk seedling survival was >50 percent only in the shallow water table-90 percent shade treatment (fig. 3).

Shade and water significantly affected Russian olive survival ($\chi^2 = 34.71_2$ and $\chi^2 = 39.02_3$, $Ps < 0.001$), and the interaction between shade and water was inconclusive

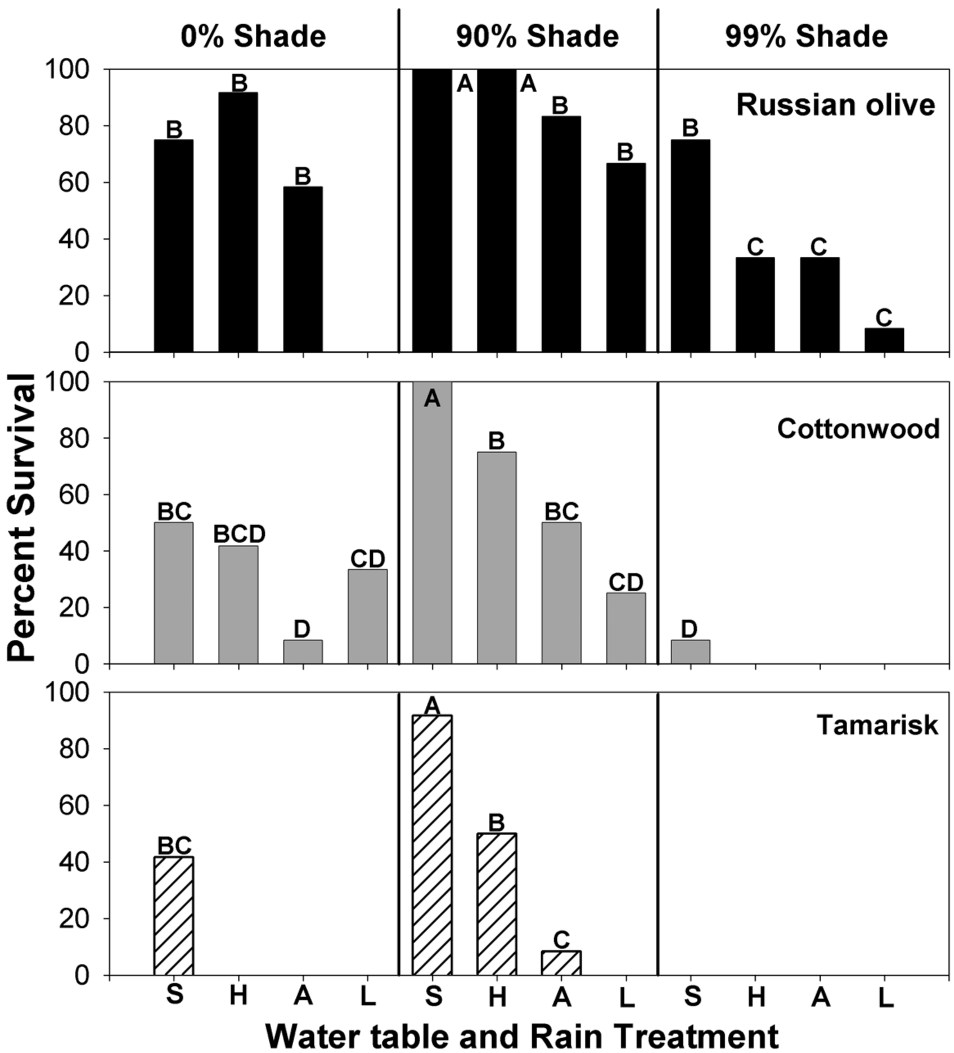

Figure 3. Percent survival of Russian olive (top), cottonwood (middle), and tamarisk (bottom) for each treatment. Shading treatment is indicated by the large boxes: full sun (0% shade), 90% shade, and 99% shade treatment. Watering treatments are indicated along the x-axis: S is shallow water table, H is high monsoon rain, A is average monsoon rain, and L is low monsoon rain. Different letters indicate significantly different survival rates within species, difference of means, and pooled variance.

($\chi^2 = 12.75_6$, P = 0.057). Russian olive survival was similar across water treatments but decreased significantly under low water conditions and in 99 percent shade (fig. 3). Shade and water significantly affected cottonwood survival ($\chi^2 = 12.56_1$, P < 0.001 and $\chi^2 = 20.71_3$, P < 0.001), and there was a significant interaction between shade and water ($\chi^2 = 8.83_3$, P = 0.0316). Cottonwood survival was higher in 90 percent than 0 percent shade and decreased with reduced water, but response to the water treatment varied by shade treatment (fig. 3). Shade and water significantly affected tamarisk survival ($\chi^2 = 11.54_1$, P < 0.001 and $\chi^2 = 24.4_2$, P < 0.001), and there was no interaction between water and shade ($\chi^2 = 0.84_2$, P = 0.658). Tamarisk survival was higher in 90 percent than 0 percent shade and lower in treatments with reduced water availability (fig. 3).

Time-to-death survival analysis generally matched the results of the logistic regression analysis summarized above. Within species, time-to-death increased in treatments receiving more water and increased from 99 percent shade, to 0 percent shade, to 90 percent shade. Tamarisk died 1.77 times faster than cottonwood (z = 4.2, P < 0.001) and 3.96 times faster than Russian olive (z = 8.54, P < 0.001).

Removal Methods and Preliminary Results

Cut-stump and whole-plant removal of tamarisk and Russian olive were compared. Cut-stump removal included cutting all tamarisk and Russian olive trees within the study sites with a chainsaw and applying herbicide Garlon® 4 to the freshly cut stumps. Whole-plant removal included removing all tamarisk and Russian olive trees from the study sites by using a backhoe. The backhoe removed all above-ground and below-ground biomass. To compare the effect of these two removal methods on native vegetation, we sampled vegetation composition within study plots along regularly spaced transects in six study areas. Transects were aligned perpendicular to the general east-west alignment of the canyon and the wash. There were three transects in each treatment (control, cut-stump, and whole-plant removal) spaced 100 m apart, for a total of nine transects in each of six sites (N = 6 x 9 = 54 transects). Transects were as long as the riparian plant community was wide. Along each transect, we sampled vegetation composition within circular nested plots 10 m in diameter and placed adjacent to each other along the transect. If the riparian community transect was 100 m long, 10 plots were sampled. Within each plot we counted and measured the diameter of all shrub and tree stems, estimated percentage canopy cover, and estimated percentage ground cover of herbaceous plants.

We are currently analyzing these data by comparing plant community composition between control, cut-stump, and whole-plant removal sites. Preliminary results indicate that plots in cut-stump treatments have a higher proportion of native plant species than plots in whole-plant removal treatments. This result is likely because of decreased levels of soil disturbance compared to the whole-plant removal sites. Soil disturbance in the whole-plant removal sites may have damaged native seed banks and created a low-competition environment for weeds to invade. Results from a recent study on the bank stability capabilities of tamarisk and Russian olive in Canyon de Chelly show that whole-plant removal sites may also increase erosion of the stream banks (Pollen-Bankhead and others, 2009).

Implications for Management

- Tamarisk and Russian olive require hydroclimatic triggers for establishment. Although we cannot rule out the importance of planting and dam installation in the invasion of tamarisk and Russian olive into Canyon de Chelly, our results clearly indicate that invasion depended on a sequence of years with above-average precipitation. Multiple years of above-average precipitation likely led to flooding conditions that facilitated Russian olive and tamarisk establishment. High precipitation years that lead to large floods along rivers are essential for large pulses of tamarisk and Russian olive invasion. Riparian managers should take action when flooding exceeds average levels for more than 2 years in a row and remove areas of tamarisk and Russian olive seedlings that establish in the available habitat. These flooding conditions will likely be favorable for native cottonwood and willow trees as well, thus careful attention to avoid damaging native plants will also be needed.
- Russian olive can establish in drier and shadier habitat than native cottonwood or tamarisk and can invade under established cottonwood and tamarisk canopies. Also, Russian olive can establish under heavy precipitation events on abandoned flood plains that are disconnected from the riparian water table. Shaded and unflooded habitats represent areas where Russian olive can establish but cottonwood and tamarisk cannot. These results indicate that large areas of potential Russian olive habitat exist along western rivers.
- Our preliminary data suggest that removal methods with the least amount of soil disturbance will help encourage native grass and herb communities. Soil disturbance in whole-plant removal sites may damage native seed banks and create low-competition environments for exotic grasses and herbs to invade. Although only preliminary, our early analyses show that cut-stump with herbicide removal of tamarisk and Russian olive leads to plant communities with higher proportions of native plants than in areas where the soil has been heavily disturbed by removal equipment.

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Geologic Considerations for the Placement and Design of Backwater Restoration Sites Along the Lower Colorado River

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Abstract

In the pre-engineered Colorado River, rapid channel shifting created numerous bodies of still water isolated from the main channel, which provided critical habitat for bird, fish, and other species. Flood-plain lakes formed and disappeared rapidly in the natural river system because of frequent channel shifting and rapid sedimentation. These geologic processes were eliminated by dam and levee construction during the 20th century, preventing the natural formation of this habitat type. The Lower Colorado River Multi-Species Conservation Program (LCR MSCP) includes a provision to restore and maintain 360 acres of "backwater" habitat along the lower Colorado River, seeking to mimic flood-plain lake habitat lost because of river engineering. Both natural and engineered lakes are subject to important geologic controls that are relevant to their creation and maintenance, and consideration of these factors would provide guidance on the proper placement and design of sustainable backwater restoration project sites. One important geologic control is the long-term stability of the main channel of the Colorado River in the proximity of the lake, which controls the local water table. A second important factor is the amount and distribution of sand-rich sediment between the lake and the main channel, which controls groundwater exchange. The sizes and shapes of lakes in the natural system, determined by geologic processes, dictate many biologically important variables to which native species were adapted. We quantify the natural distribution of lake sizes and shapes by using historical maps made in 1902. The natural river system contained many small lakes and a few large lakes. Historical analysis of channel changes and geologic mapping in the vicinity of proposed restoration sites could aid in site selection and design of backwater restoration projects and increase the likelihood that these projects will succeed over the 50-year time horizon of the LCR MSCP.

Introduction

The broad valleys along the lower Colorado River (LCR) (fig. 1) contain many bodies of still water isolated from the main channel (Grinnell, 1914). These water bodies are recognized as important breeding, foraging, and refugia sites for wildlife, including fish and bird species considered by the Lower Colorado River Multi-Species Conservation Program (LCR MSCP) to be threatened and endangered. The chain of flood-plain lakes along the LCR also provide rare and valuable open water for migrating birds along the Pacific Flyway. The ecological significance of these features was recognized by the LCR MSCP Habitat Conservation Plan, which aims to create and maintain 360 acres of "actively managed connected and disconnected backwaters" to provide habitat for razorback sucker (*Xyrauchen texanus*) and bonytail (*Gila elegans*), and "to provide surface and ground-water hydrology in support of existing or created habitat" for covered bird species (Lower Colorado River Multi-Species Conservation Program, 2004, p. 5–16).

Holden and others (1986) developed a classification system for backwater habitats along the LCR and identified some of the most important ecological variables that determine habitat value. Minckley and others (2003) and Mueller (2006) recommended that because native fish are vulnerable to predation by nonnative species in the mainstem, the best locations for sustainable populations of listed fish species would be in water bodies not connected to the main channel by way of a surface-water connection. The adoption of the LCR MSCP in 2004 prompted renewed interest in isolated backwater habitat along the LCR, and BIO-WEST, Inc. (2007) recently proposed an updated classification system for determining the biological suitability of isolated backwater habitats. This classification system includes indicators of cover, water depth, and the

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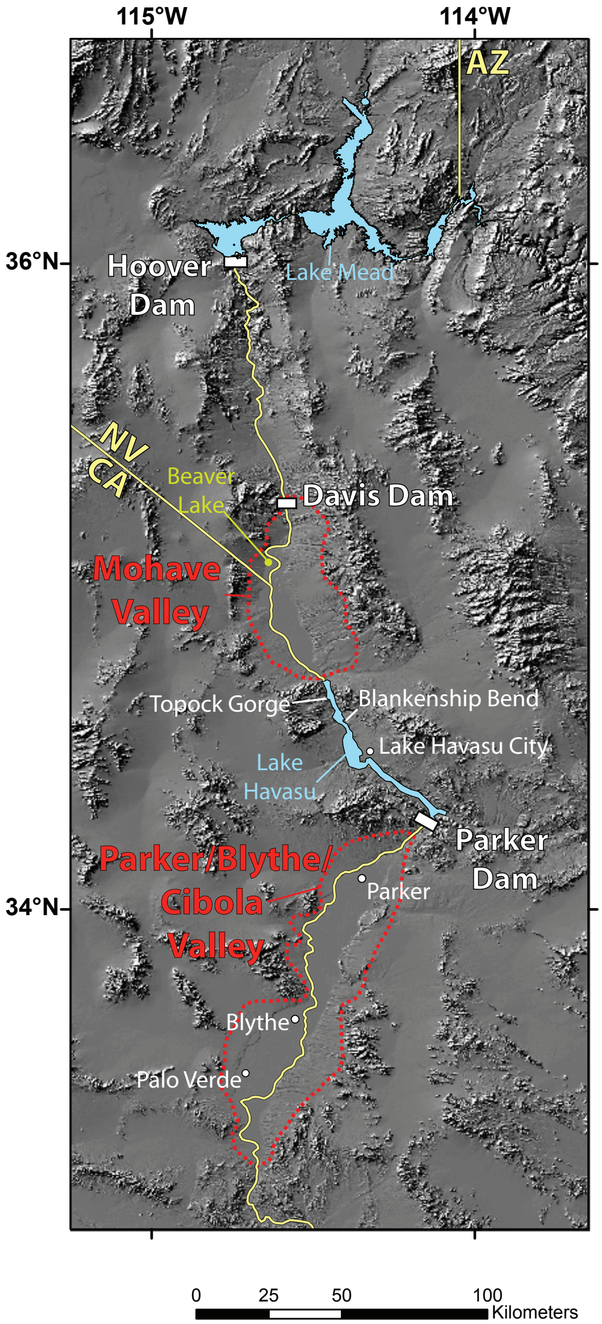

Figure 1. Hillshade image of the lower Colorado River valley based on a digital elevation model.

particle size of substrate material, as well as water conditions, such as pH, turbidity, salinity, temperature, dissolved oxygen, selenium, chlorophyll, and cyanobacteria content.

The ability of these projects to maintain these biologically appropriate conditions over several decades or longer is closely linked to geologic factors in the vicinity of the lakes. In the natural river system, large disconnected backwaters formed primarily in abandoned channels of the Colorado River, so they commonly occupied curved and elongated depressions in the flood-plain surface. The sizes and shapes of these features affect water temperature, evaporation rate, salinity, and other biologically important variables. In addition, the amount and distribution of sand, silt, and clay substrates within and around the lake directly affect both habitat suitability and the rate of groundwater exchange.

The current site-selection process for backwaters does not provide any guidance on the geologic context for site selection. The aim of this paper is to partially fill this gap by discussing the most important geological considerations relevant to the placement and design of isolated backwater restoration projects in the LCR valley. After considering the main controls on flood-plain lakes in the natural system, we discuss four important geological factors that should be considered while designing backwater restoration projects in the LCR: (1) the vertical stability of the main channel in the vicinity of the project reach, (2) material properties between the lake and the main channel, (3) the sizes and shapes of natural lakes, and (4) new lake formation caused by fluvial and deltaic sedimentation near the upstream ends of major reservoirs.

Lakes in the Natural River System

Infilled flood-plain lakes are common in the geologic record of the Colorado River. The remains of flood-plain lakes are particularly prominent in widespread exposures of Pleistocene (1.8 million to 10,000 year-old) Colorado River deposits on the margins of the valley. These deposits have yielded fossilized remains of birds, fish, turtles, and other species (Metzger and others, 1973), demonstrating that flood-plain lakes have been an important component of the Colorado River's natural riparian ecosystem for at least tens of thousands of years. Flood-plain lakes were described and mapped by explorers, scientists, and engineers in the 19th and early 20th centuries and are recorded in Tribal histories. By all accounts the river before 1935 was a very dynamic system, in which "there are few places in the bottom lands that may not, during any season, be overrun" (Ives, 1861). Owing to the tendency of the river to frequently flood the valley from wall to wall, the typical lifespan of a natural flood-plain lake was short, on the order of several decades (Ohmart and others, 1975). We digitized lakes in a series of maps of the Colorado River valley made in 1892, 1902, 1950, and 1975. In these maps, almost none of the large, named lakes in the river valley persisted in more than one of the map sets (T. Felger, unpub. data, 2006).

The evolution of Beaver Lake (fig. 2; see fig. 1 for location), a crescent-shaped lake formed in a former channel of the Colorado River in the northern part of Mohave Valley, illustrates the life cycle of a naturally formed flood-plain lake. The lake was first described in an 1859 newspaper article as a 3-mile-long, crescent-shaped lake containing abundant duck and beaver (Ohmart and others, 1975). The lake was shown in a map by Wheeler (1869) and mentioned by Stanton (1890).

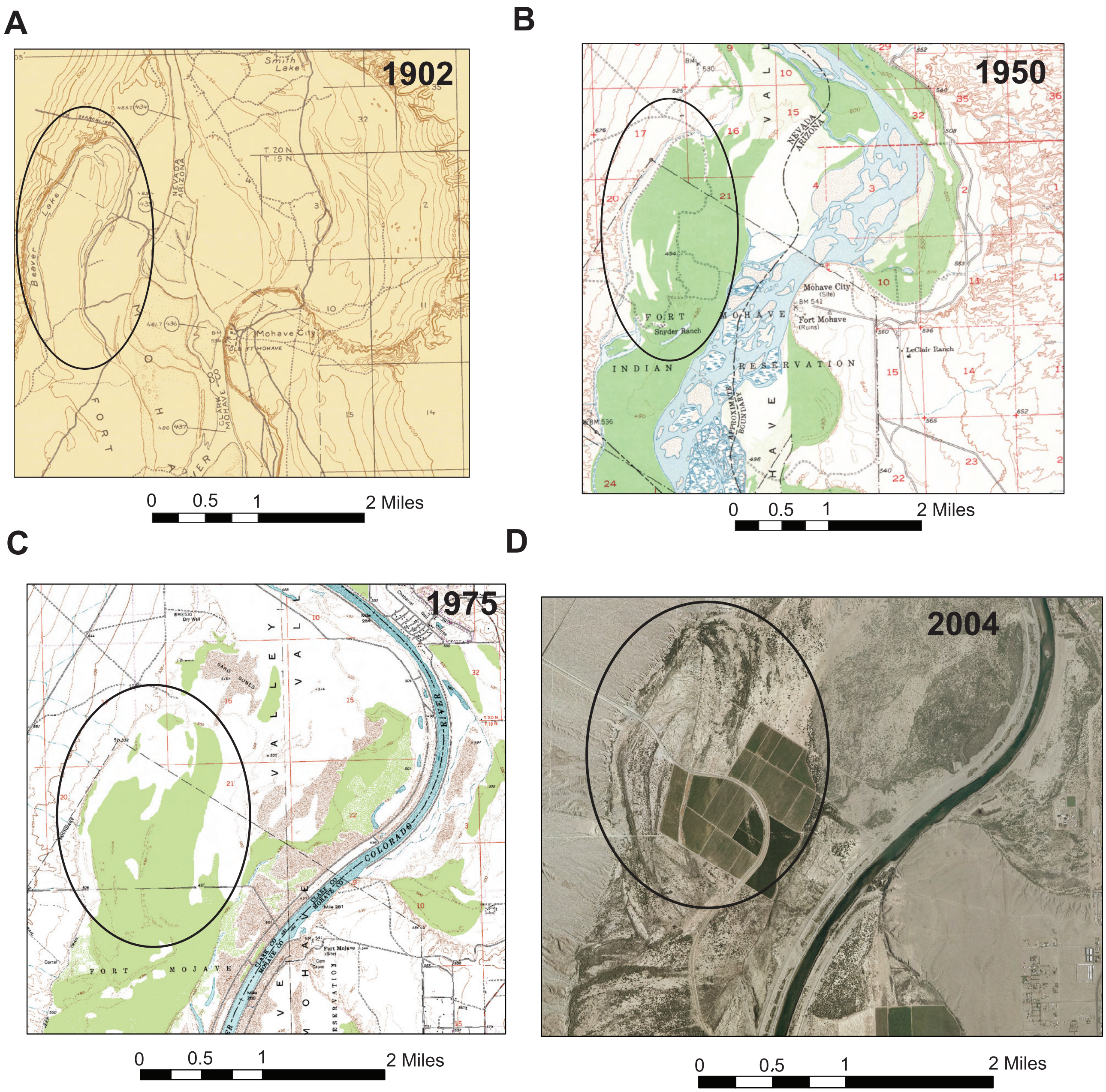

Figure 2. Beaver Lake, in Mohave Valley, as shown in three different topographic maps from (A) 1902, (B) 1950, and (C) 1975, and (D) in a recent (2004) digital orthophotograph.

As shown in a 1902 map, Beaver Lake appeared to be a well-established lake with a surface area of 122 acres (fig. 2A). Ohmart and others (1975) state that the river shifted to the east side of the valley after a large flood in 1905, isolating the lake from the river by 3.5 miles. The lake appears smaller on a 1926 map, but retained its general shape (not shown). By 1950, however, Beaver Lake was not labeled on the topographic map (fig. 2B). By 1975, following the closure of Davis Dam and subsequent channelization of the river, evidence for a lake in the area of Beaver Lake had disappeared (fig. 2C). On the orthophotograph from 2004, the former lake appears as a slight topographic low in the flood-plain, occupied by stands of mesquite and tamarisk (fig. 2D).

Maps and historical records of Beaver Lake, and of other lakes, demonstrate that the lifespan of typical lakes in the natural river system was relatively short, on the order of several decades. Before dam construction, sediment laden floods of the Colorado River would spill onto the flood plains, carrying sand, silt, and clay that rapidly filled depressions on the flood plain. In addition, lake destruction would have been enhanced by high evaporation rates and by frequent movement of the main channel, which could obliterate lakes or isolate them from groundwater recharge.

Factors Influencing the Placement and Design of Backwater Restoration Projects

Channel Stability

Over geologic time, the Colorado River has undergone major cyclical fluctuations in bed elevation. During the Pliocene epoch (approximately 5.3 to 1.8 million years ago), early in the river's history, one or more major aggradational cycles filled the valley with as much as 300 meters (m) of predominantly coarse-grained river-laid sediments (e.g., Longwell, 1936). During the Pleistocene epoch ("the ice ages," approximately 1.8 million to 10,000 years ago), the river filled its valley at least once and likely twice with hundreds of feet of sand, silt, and clay, and there is evidence from the subsurface that the river aggraded significantly during the Holocene (10,000 years ago to present). The remains of Pleistocene and Pliocene aggradational events are widely preserved as fluvial deposits on the margins of the valley above the modern flood plain and form the surfaces and terraces on which much of the urbanization along the Colorado River is concentrated. Following each episode of valley filling, the river incised back through this fill (degraded), resulting in the excavation and downstream transport of much of this fill material. These aggradation/degradation cycles in the Colorado River have been instigated by multiple geologic and climatic mechanisms; the cycles have been attributed to tectonic activity, lake spillover, watershed climate changes, natural impoundments, sea level change, and the carving of Grand Canyon. Aggradation or degradation has also been caused in places along the Colorado River by human intervention, such as dam construction, dredging, bank armoring, and the building of artificial levees. In general, the channel aggrades and degrades in response to perturbations that alter the balance between the supply and transport capacity of the bed material load (generally coarse sand and gravel-sized sediment that can be deposited in the channel bed).

Long-term trends of aggradation or degradation exert a primary geologic control on the formation and evolution of flood-plain lakes, in both the natural and engineered river. Aggradational conditions of the Colorado River favor both rapid formation and rapid destruction of flood-plain lakes. During aggradation, a surplus of the bed material load is deposited in point bars and other channel features, causing frequent channel shifting and, therefore, frequent lake formation. Aggradation of the channel bed is also likely to be accompanied by high suspended-sediment concentrations, leading to high rates of lake infilling. Radiocarbon dates on wood fragments encountered in cores drilled beneath the modern flood plain show that the LCR aggraded in early Holocene, since the most recent deglaciation and subsequent sea level rise (Metzger and others, 1973; D. Malmon and K. Howard, unpub. data, 2008). The short lifespan of flood-plain lakes in the river before major human intervention may partially be the result of this Holocene aggradational episode.

By contrast, long-term river degradation reduces the frequency of channel shifting. A deficit in the bed material load leads to channel narrowing and bed coarsening, resulting in an entrenched, single-thread channel. The geologic record contains evidence of bed coarsening during the degradational phases of such cycles during the Pleistocene (e.g., Longwell, 1936). Historical degradation of the Colorado River below Hoover, Davis, and Parker Dams was also accompanied by channel narrowing and coarsening of the bed texture (Williams and Wolman, 1984); this degradation has been widely cited as a textbook example of river response to sediment deficit caused by dams. Degradation and associated channel narrowing would tend to inhibit channel shifting and reduce the rate of lake formation. In addition, because the river controls the regional groundwater table, downcutting can lower the water table across the entire flood plain, leading to the stranding and dewatering of lakes.

The engineered Colorado River has some reaches that are aggrading and others that are degrading, with important consequences for both natural and engineered flood-plain lakes. In degrading reaches, lakes are likely to be dewatered. For example, in the vicinity of Needles, CA, and the northern part of the Havasu National Wildlife Refuge, gradual channel degradation has occurred in response to channelization locally between 1949 and 1953 (Malmon and others, 2009), requiring the installation and operation of large pumps to move water from the river to the flood-plain surface to retain the prescribed water level within Topock Marsh in Havasu National Wildlife Refuge (J. Earle, refuge manager, oral commun., 2008). The disappearance of Three Fingers Lake, near Blythe, CA, has also been attributed to relocation and lowering of the river channel (Ohmart and others, 1975).

In other places, aggradation of the modern river is occurring where sand-sized sediment is depositing, such as at the heads of reservoirs. Long-term aggradation may lead to channel shifting and rising water tables, which could impact present and future backwaters. The closure of Parker Dam in 1938 led to rapid sedimentation at the head of its Lake Havasu Reservoir, causing rapid channel shifting and flooding as far north as Needles (50 miles upstream from the dam). Aggradation of the main channel upstream from the reservoir led to the creation of Topock Marsh, resulting in the flooding of several previously isolated lakes in lower Mohave Valley (Ohmart and others, 1975). Localized aggradation and degradation are also occurring in specific reaches other than above reservoirs, in response to past river engineering by the Bureau of Reclamation.

Engineered flood-plain lakes or backwater restoration projects are meant to last as long or longer than those that formed in the natural river system, so they will be subject to the same long-term (decadal) influences that affected lakes in the natural system, including aggradation or degradation of the main channel in the vicinity of the lake. Lakes built in aggrading reaches may risk being flooded over time, while those in degrading reaches may risk being dewatered. To avoid or anticipate future maintenance costs for future dredging (in aggrading reaches) or the installation of pumps and inlet structures, canals, and dikes (in degrading reaches), the long-term stability of the channel in the vicinity of proposed backwater restoration sites should be examined. Determination of whether a reach is aggrading or degrading can be accomplished with field observations and by comparing modern data with historical records. Useful types of historical records include repeat aerial and ground photographs, old maps, bridge as-built surveys, records of river stage at low flow at nearby gaging stations, or surveyed cross sections made for past hydrologic modeling projects along the LCR. In addition, the Web site of the Colorado River Front Work and Levee System (*http://www.usbr.gov/projects/Project.jsp?proj_Name=Colorado%20River%20Front%20Work%20and%20Levee%20System*) provides a historical discussion of aggrading and degrading reaches along the LCR.

Material Properties

The distribution of sand- and silt-sized sediment on the flood plain is a second important geologic influence on flood-plain lakes. The exchange of water through sandy substrate is many orders of magnitude more rapid than through clayey and silty sediment, making the distribution of sediment grain size on the flood plain an important control on water availability and quality. Subsurface water flow between flood-plain lakes and the main channel is essential to prevent deoxygenation, salinization, temperature rise, and contaminant accumulation (Ohmart and others, 1975). In the LCR area, evaporation rates are high, and evaporated water must be replaced with groundwater influx to support ecosystems. Inadequate groundwater flux may prevent lakes from maintaining the appropriate water depth, temperature, salinity, and oxygen levels, making them useless to some species of wildlife.

The pre-engineered Colorado River sorted its sediment load by particle size and deposited relatively coarse sediment (sand and gravel) in channel settings and finer grained sediment (fine sand, silt, and clay) on flood plains. Natural flood-plain lakes, which occupied abandoned channels of the river, likely formed in sand-rich substrate. As lakes were filled in with finer grained sediment during overbank floods, a change in grain size lead to a progressive reduction in the hydrologic connection of the lakes with the river-controlled water table. We speculate that this mechanism may have contributed significantly to the desiccation of natural lakes.

Engineered flood-plain lakes must maintain a subsurface hydrologic connection with the main channel of the Colorado River or be watered through a system of pumps and canals. Thus, it would be advantageous, in terms of water, energy, and infrastructure costs, to locate engineered lakes in locations where subsurface water exchange will be enhanced. A rule of thumb could be to ensure that restoration backwaters are connected to the main channel by way of one or more contiguous pathways of sand-dominated sediment with adequate hydraulic conductivity, so that groundwater influxes may compensate for evaporation from the lake surface (fig. 3A).

Such pathways can be identified relatively easily through geologic and soils mapping of the flood plain in the vicinity of proposed backwater restoration sites. Sediment deposited in the former channel of the Colorado River tends to be dominated by sand, whereas sediment deposited in the flood plain contains a higher fraction of silt- and clay-sized sediment. Historical investigations of river channel changes on repeat sets of aerial photographs and maps can provide guidance on the distribution of deposits in the modern flood plain and be supplemented by strategic field sampling of flood-plain sediments. For example, a recently completed map of the flood plain near Needles delineated the approximate distribution of channel deposits and overbank deposits through analysis of channel positions in six sets of historical maps and aerial photographs (Malmon and others, 2009) (fig. 3B). Geophysical techniques, such as ground-penetrating radar, may also be useful for identifying and mapping irregularly shaped sand bodies in the subsurface of the flood plain.

Lake Sizes and Shapes

The distribution of lake sizes and shapes influence water depth, temperature, and other parameters that determine habitat quality. The native fish species targeted by backwater creation projects have evolved within a system of lakes with a particular combination of sizes and shapes. If one of the goals of the LCR MSCP is to create isolated backwaters that mimic the habitat types (including patch size, surface area, depth, temperature, perimeter-to-area ratio, etc.) found in the undisturbed river system, it may be desirable to allocate backwater habitat in a way that imitates the distribution of lake sizes and shapes in the natural river.

Detailed plane-table survey maps from 1902 (U.S. Geological Survey, 1927) portray baseline conditions for the distribution of lake sizes and shapes in the natural river system. We digitized all bodies of water isolated from the main channel in the 1902 maps and compiled a digital database of lakes in the natural system (fig. 4). A total of 145 lakes were mapped in the Mohave Valley and the Parker/Blythe/Cibola Valleys (fig. 1) (located in reaches 4 and 3 of the LCR MSCP, respectively). Within these two valleys, isolated or disconnected backwaters occupied 962 acres of the valley floor—a mean lake area of 6.6 acres. However, the mean is not necessarily a good indicator of the patch size of typical lake habitat in the natural system. By far, most of the mapped lakes were smaller than 3 acres (fig. 4A; note that lakes in the “0–3 acre” bin had an average area of 0.5 acres). However, most of the total lake area was within larger lakes (fig. 4B, C); lakes larger than 100 acres contained 58 percent of the total lake area, and lakes larger than 30 acres contained 71 percent of the total lake area. The natural distribution of lake surface area was bimodal (fig. 4C), and most of the area of isolated lake area was in

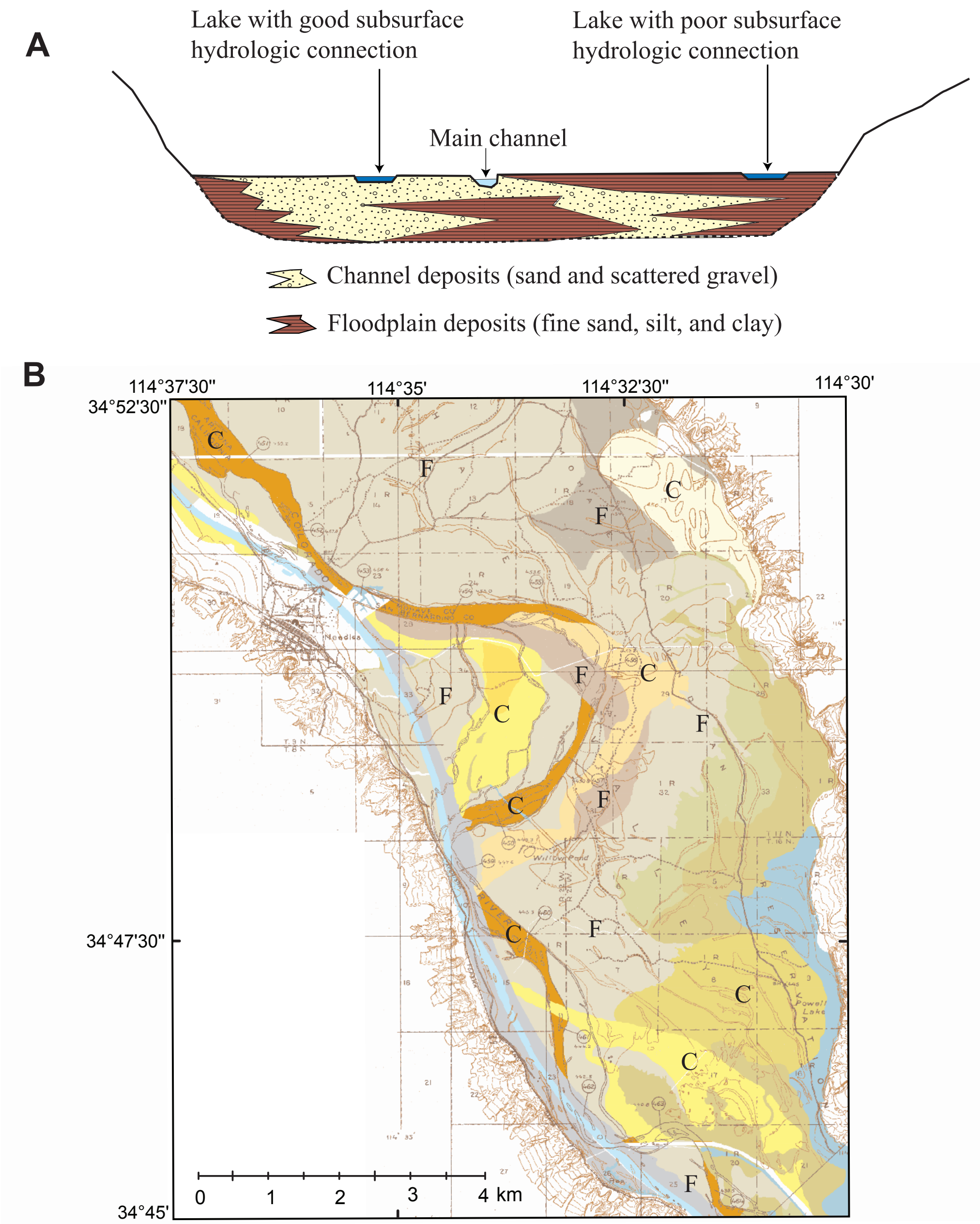

Figure 3. (A) Schematic cross section of coarse and fine sediments in the lower Colorado River valley, showing a location for a lake having adequate subsurface hydrologic connection with main channel by way of a continuous sand body, and a location with a poor subsurface hydrologic connection. (B) Mapped distribution of coarse and fine-grained sediment in the valley floor within the Needles 7.5' quadrangle. Shades of yellow and orange labeled "C" indicate relatively coarse-grained, sand-rich channel deposits. Green and brown shades labeled "F" indicate inferred flood-plain deposits (silt- and clay-rich deposits) (from Malmom and others, 2009). Mapping is based on historical documentation of channel position, and only qualitatively field-checked; such a map could be refined with field sampling and particle size analyses for the purpose of siting backwater restoration sites.

small lakes and large lakes; only 11 percent of the total lake area was in intermediate-sized lakes (between 6 and 20 acres).

The natural river system contained both large and small lakes, and each size class provided a particular set of biotic conditions and habitat type. To mimic the distribution of lake sizes in the natural river system, a relatively large number of small lakes and a small number of large lakes would be required. This distribution could potentially be modeled after the distribution shown in figure 4.

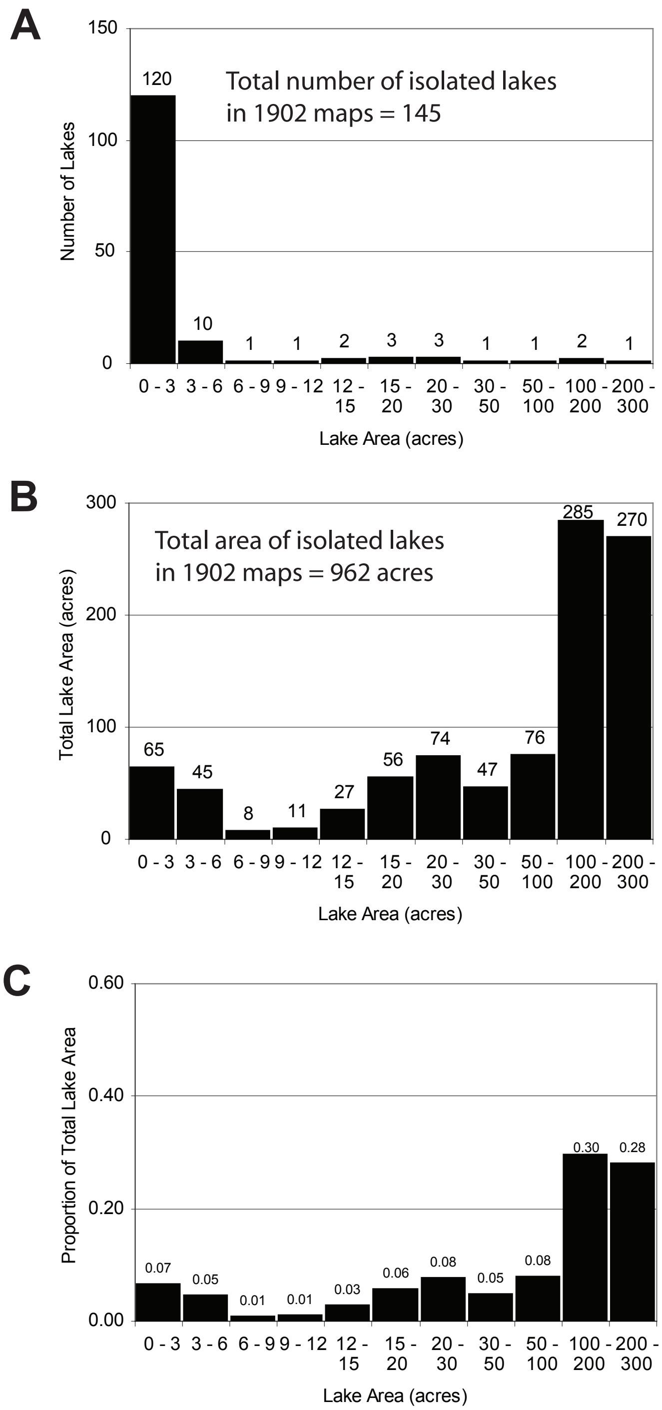

Figure 4. Statistics for isolated lake area in the pre-engineered Colorado River (Mohave, Parker, Blythe, and Cibola Valleys) from digitizing lakes on 1902 plane-table survey maps of the lower Colorado River. (A) Number of lakes in the natural system. (B) Total lake area within lakes of different size. (C) Proportion of total lake area in different size classes.

Deltaic Sedimentation and the Formation of Valley-Mouth Lakes

Lakes form at tributary mouths when aggradation in the main channel of the Colorado River outpaces aggradation in tributaries and creates barrier dams (fig. 5). Lakes formed in this way are common in the geologic record, not only along the Colorado River but also along the Columbia, Mississippi, and other rivers. Valley-mouth lakes are also common in the modern river at areas of rapid aggradation, including at the upstream ends of reservoirs. In reservoirs, this damming mechanism may be enhanced by wave action. As recognized by the LCR MSCP, though not explicitly stated, small bodies of standing water dammed at the mouths of tributaries adjacent to and upstream from reservoirs may present a restoration opportunity for backwater projects. For example, sedimentation at the delta of Imperial Reservoir (in reaches 4 and 5 of the LCR MSCP) created hundreds of lakes of this type, some of which are currently being used as backwater project sites (Lower Colorado River Multi-Species Conservation Program, 2007). Similar lakes are also being used to rear native fish in Lake Mohave. In Lake Havasu, barrier bars are currently forming across tributary mouths north of Lake Havasu City. North of Blankenship Bend (fig. 1), many such lakes are hydrologically isolated from the reservoir. North of River Island, in Topock Gorge, isolated lakes have formed at the mouths of nearly every small tributary. Such features may persist as lakes for several decades, but they will eventually be filled in by deltaic sedimentation, beginning upstream and advancing downstream. In the meantime, the lakes may continue to be isolated from the reservoirs, making them habitable by native fish. In addition, they are likely to have reasonably good subsurface hydraulic connections, owing to the sandy substrate of the barrier bars and a close proximity to the high water table.

Figure 5. Example of an isolated body of water formed at the mouth of a small side valley in Topock Gorge near the upstream end of Lake Havasu. The lake formed as a result of sedimentation at the head of the reservoir, blocking the tributary mouth.

These lakes may provide good opportunities for creating small isolated water bodies (“backwater habitat”) in the LCR valley for the next several decades, but they have significant limitations. It must be recognized that these lakes will be ephemeral features and, in the absence of dredging, will most likely be filled with sand within several decades, eliminating their potential for providing habitat. Furthermore, their habitat suitability may not match that of natural flood-plain lakes. For example, valley-mouth lakes have very different shapes than natural flood-plain lakes, and they may be too close to sources of nonnative fish stock to prevent predator species from being introduced. In addition, while these lakes may replicate the small lake habitat that existed in the natural system, none of these lakes are large enough to create the ecological conditions that existed in large abandoned channels of the Colorado River (fig. 3B).

Summary and Conclusions

Flood-plain lakes along the Colorado River provide critical habitat for many endemic and now endangered fish, bird, and other species. In the pre-engineered Colorado River, isolated lakes formed frequently as a result of rapid channel shifting. Lakes were destroyed by subsequent channel shifting, dewatering, and overbank sedimentation, which occurred following turbid floods. Therefore, lakes in the natural system had short life spans, likely on the order of decades. Damming and confinement of the Colorado River eliminated the mechanisms by which these features were formed and destroyed. Current efforts to “restore” backwater habitat by building and maintaining isolated lakes along the river corridor can benefit from considering the most important geologic factors that control the function of these systems.

A primary factor controlling the longevity of lakes, in both the natural and engineered river system, is the stability of the main channel in the vicinity of the lake. Long-term channel aggradation favors channel shifting and a rising water table, possibly resulting in lake infilling and the establishment of surface-water connections between the lakes and main channel, allowing the introduction of predator fish species to engineered backwaters. Long-term degradation of the main channel may cause lakes to be dewatered, requiring the installation and maintenance of pumps and canals. It would be most efficient to place projects in reaches that are neither aggrading nor degrading over several decades.

Another important factor controlling lake function is the distribution of sand, silt, and clay between the lake and the main channel, because these materials control the flow of groundwater between the main channel and the lake. Sandy deposits allow subsurface exchange of water between the lake; silt- and clay-rich deposits inhibit water exchange. Where possible, lakes should be sited in locations where a subsurface hydrologic connection can be maintained by way of a contiguous sand body. These bodies commonly trace the recent courses of the main river channel and can be readily identified by geologic and soils mapping guided by historical aerial photograph interpretation.

The sizes and shapes of lakes influence their hydrologic and ecologic function by controlling evaporation rates, water depth, water temperature, and patch size. Natural lakes had a characteristic distribution of sizes and shapes, determined by geologic processes, to which species that use the lakes have adapted. Thus, one restoration goal may be to mimic the distribution of lake sizes and shapes of the natural river system. The areas of 145 isolated flood-plain lakes in a detailed set of maps surveyed in 1902–03 show that the natural system contained many small lakes and relatively few large lakes. More than one-half of the area of isolated lakes in the natural system was contained within several lakes each having a surface area greater than 100 acres.

Deltaic sedimentation near the upstream ends of reservoirs commonly blocks the mouths of tributary valleys, creating off-channel lakes that may be temporarily habitable by native fish. Some of these valley-mouth lakes are being adapted for backwater restoration in the headwaters of Imperial Reservoir, and such lakes also occur in Lake Havasu north of Blankenship Bend. These small lakes are close to but hydrologically isolated from the predatory nonnative fish in the main reservoir and may continue to maintain water levels for several decades because of a high water table and sand-rich substrate. However, these sediment-dammed, valley-mouth lakes will, in the absence of dredging, fill in with sand, silt, and clay as deltaic sedimentation progresses, a process that occurs over a time scale of decades.

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Ecosystem Restoration—Alamo Lake and the Bill Williams River

By William E. Werner[1, 2]

Abstract

Alamo Dam was completed in 1968 on the Bill Williams River, a tributary to the Colorado River, for flood control, water conservation, and recreation. Riparian woodland habitats, particularly cottonwood- (*Populus fremontii*) willow (*Salix gooddingii*) gallery forest, found on the Bill Williams River are believed to be a relic of habitats once common along the lower Colorado River. In 1990, a multiagency steering committee-lead effort was initiated to develop a consensus recommendation among resource agencies on improvements to operation of the dam to benefit a suite of resources, including fish, wildlife, and their habitat both upstream and downstream from the dam. That process culminated with a Record of Decision on an Environmental Impact Statement in 1999 and a revised water control manual in 2003. Since then a rechartered steering committee has worked to gather data and develop models to support adaptive management of the system. Products include a digital terrain model, measurements made during high flow (sediment, turbidity, and water-surface elevations), a hydraulic model (HEC-RAS), an ecosystem functions model (HEC-EFM), and biologic monitoring to link flows to ecological responses.

Introduction

The Bill Williams River (fig. 1), in west-central Arizona, is a tributary to the Colorado River with confluence about 0.75 kilometers (km; 0.5 mile (mi)) above Parker Dam, which forms Lake Havasu. The Bill Williams River itself begins at the confluence of the Santa Maria and Big Sandy Rivers about 12 km (7.5 mi) upstream from Alamo Dam, which forms Alamo Lake. Following historic Bill Williams River floods in February 1890, February 1891, and February 1937, which resulted in flooding in developed valleys along the lower Colorado River, Congress authorized construction of Alamo Dam in 1944. Planning was conducted in the early 1960s, and construction was completed in 1969. Additional Congressionally authorized purposes of Alamo Dam included water conservation and recreation. Precipitation in the watershed ranges from approximately 45 centimeters (cm; 18 inches (in.)) in the headwaters to 22 cm (9 in.) at Alamo Dam (National Climatic Data Center station Alamo Dam 6ESE) to 13 cm (5 in.) at Parker, AZ, near the Colorado River confluence (National Climatic Data Center station Parker 6NE). Alamo Dam itself is a rolled earthfill type structure 84 meters (m; 275 feet (ft)) in height. The reservoir Alamo Dam impounds has a capacity of 122,768 hectare meters (ha-m; 995,300 acre-feet (acre-ft)), about nine times mean annual inflow, with 616 ha-m (5,000 acre-ft) allocated to recreation; 28,370 ha-m (230,000 acre-ft) allocated to water conservation; and 75,041 ha-m (608,369 acre-ft) allocated to flood control (U.S. Army Corps of Engineers, 2003).

The Bill Williams River supports riparian habitat, particularly cottonwood- (*Populus fremontii*) willow (*Salix gooddingii*) gallery forest, (fig. 2), believed to be relic of habitat once found along the lower Colorado River. Following large inflows in 1978, 1979, and 1980, water was held in Alamo Lake because of concurrent Colorado River flooding. During this time, Alamo Lake reached record elevations. To evacuate the water, once capacity in the Colorado River was available, long-duration releases of 60–70 times base flow were made in 1979 and as much as 100 times base flow in 1980, on the basis of the original "Water Control Manual, Alamo Dam and Lake, Colorado River Basin, Bill Williams River, Arizona" (U.S. Army Corps of Engineers, 1973). Prolonged inundation from extended high releases was commonly believed to have resulted in mortality of cottonwood trees, which is a matter of concern considering the existing reduction of areal extent from the pre-dam period reported by Ohmart (1982). Shafroth and others (2002) reviewed riparian vegetation changes associated with Alamo Dam and noted that effects of inundation by high flows may have been localized, but effects of low base flow may have been more widespread. Ohmart (1982) attempted to quantify changes in riparian vegetation from that described in historical accounts through the post Alamo Dam period, estimating a 70-percent reduction between Alamo Dam and the Bill Williams River confluence with the Colorado

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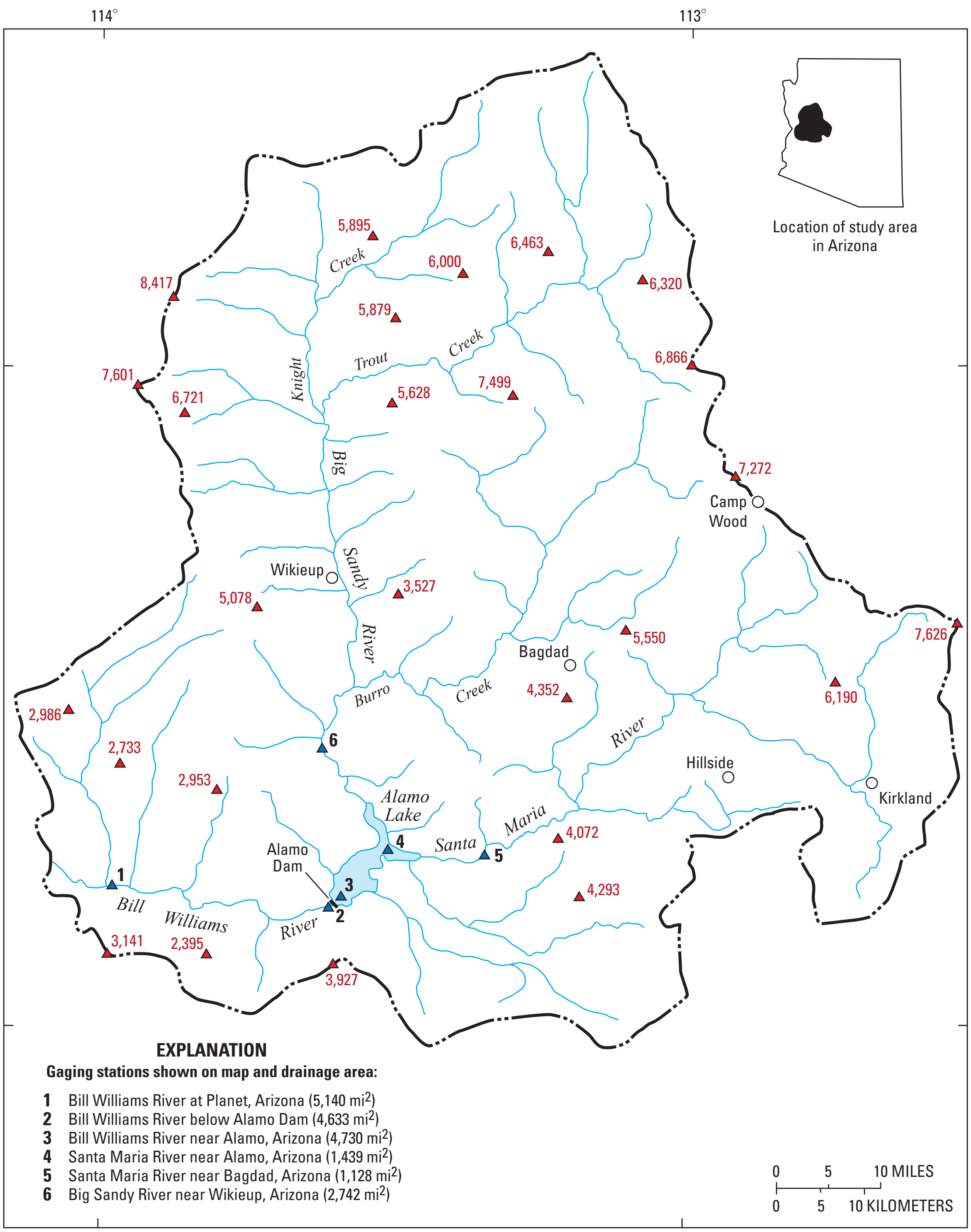

Figure 1. The Bill Williams River Basin (from Shafroth and Beauchamp, 2006).

Figure 2. Riparian vegetation along the Bill Williams River on the Bill Williams River National Wildlife Refuge, Arizona.

River. Classification of vegetation by species apparently was problematic because of the resolution of some of the early photographs. This reduction includes the reach, approximately 6.4 km (4 mi), of the Bill Williams River inundated by what is now Lake Havasu by the construction of Parker Dam on the Colorado River in 1938.

Planning Process

Management for native riparian woodland habitat is a priority for many resource agencies, and agency personnel were concerned about mortality and lack of recruitment in existing stands. Agencies began to focus on fish and wildlife habitat issues associated with Alamo Dam operation although not in a coordinated manner. In 1990, the Arizona Game and Fish Department convened leaders of involved agencies, including Arizona State Parks Department, Arizona Department of Water Resources (as an advisor), Bureau of Land Management, Bureau of Reclamation, U.S. Army Corps of Engineers (hereafter, Corps of Engineers), and U.S. Fish and Wildlife Service. The Arizona Game and Fish Department is involved in two principal ways, with statutory responsibility for protection and management of wildlife throughout Arizona and as manager, for fish and wildlife purposes, of most of the land controlled by the Corps of Engineers at Alamo Lake. The agency leaders agreed to a goal to "carry out a coordinated interagency planning effort to develop an effective water management plan for Alamo Lake and Bill Williams River corridor resources" (Bill Williams River Corridor Technical Committee, 1994) and to a process to develop a consensus recommendation for operation of Alamo Dam. Summarized, this process was to (1) assemble a committee of representatives from each agency—the Bill Williams River Corridor Technical Committee (BWRCTC), (2) identify each agency's resources goals and objectives, (3) formulate alternative reservoir operation plans that best meet collective goals, (4) analyze/evaluate alternative reservoir operation plans, (5) collectively select the reservoir operation plan that best meets all agency resource objectives while acknowledging the importance of other agency objectives, and (6) submit the recommended operation plan.

To begin the planning process, problems, needs, and opportunities were identified for threatened and endangered species, enhanced water-based recreation, restoration and enhancement of Bill Williams River riparian habitats, wildlife habitat in general, and improved fisheries at Alamo Lake and the Bill Williams River. These problems, needs, and opportunities were to be considered in context of Alamo Dam operation for flood control, water conservation, recreation, and inspection and maintenance needs, which are the Corps of Engineers' authorized purposes and requirements. Riparian, fisheries, wildlife, recreation, and reservoir operations technical subcommittees were appointed by agency leaders. The riparian, fisheries, wildlife, and recreation subcommittees were tasked with independently preparing reports identifying, for their resource objective, optimum Alamo Lake elevation and optimum downstream flow regime by month. The products of the fisheries, riparian, recreation, and wildlife technical subcommittees were then integrated with reservoir operations authorities and physical constraints to formulate alternative operation scenarios for the operation of Alamo Dam.

Recommendations to benefit cottonwood and willow trees are based on foundational concepts, summarized by Shafroth and Beauchamp (2006), that in a natural setting, river floodflows remove vegetation and scour and deposit mineral soils within the river's flood plain, thus creating seedbeds. Germination and successful establishment can occur when seeds lodge on those flood-scoured or deposited surfaces, provided that post-flood water table decline is at a rate slower than tree seedling root growth. Common factors in alternative reservoir operation plans developed by the BWRCTC (1994) included: (1) riparian habitat streamflow requirements, with consideration of seasonal base flow, would support established vegetation below Alamo Dam; (2) floodflows would be released in a more natural manner; (3) the rate of change of the elevation of Alamo Lake would be limited during the largemouth bass (*Micropterus salmoides*) spawning season; (4) drawdown to perform required dam inspection and maintenance would be factored in; (5) and adaptive management would enable improvements based on monitoring.

With operation scenarios described in terms of optimum Alamo Lake elevations and optimum downstream releases from Alamo Dam, performance of the scenarios was modeled and evaluated using the Corps of Engineers HEC-5 computer program (U.S. Army Corps of Engineers, 1982). This program tracks streamflow, evaporation, diversions, and reservoir storage, using conservation of mass in a large spreadsheet type program. Daily flow data from 1928 to 1993 for the gage site on the Bill Williams River below Alamo Lake was used in the simulation. In the simulations, inflow to and evaporation and releases from Alamo Lake, evapotranspiration from the riparian woodland vegetation, pumping from the Planet Ranch aquifer, and discharge to Lake Havasu were calculated on a daily time step under each of the alternative operating schemes developed (fig. 3). A naturalized flow series for the gage below the Alamo Dam was created using pre-dam gage data and upstream gage and inflow data on the basis of change in lake stage into Alamo Lake. Evaporation was based on pan evaporation at Alamo Dam. Evapotranspiration was estimated from the areal extent riparian vegetation downstream from Alamo Dam by using evapotranspiration rates for the lower Colorado River. Information on groundwater/surface-water interaction was based on work by Rivers West, Inc. (1990). Details on modeling are included in a report of the Bill Williams River Corridor Technical Committee (1994). In other words, an analysis was completed of how the system would perform if the dam were in place and operated under a certain approach under conditions as they were before dam construction (for example, 1939) or any other year during which river flow records were kept.

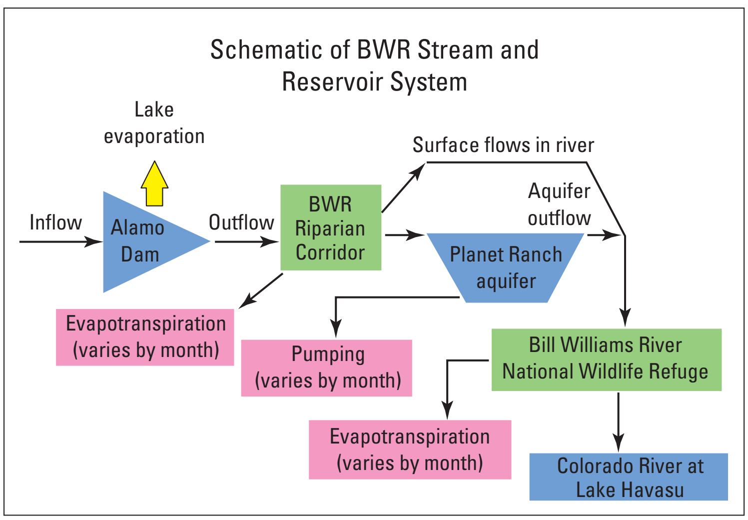

Figure 3. Schematic of Bill Williams River (BWR) stream and reservoir system used in hydrologic modeling.

Evaluation criteria for recreation included percentage of time the water surface in Alamo Lake would be within the operating range of existing boat ramps. Criteria for water conservation included quantification of the amount of water delivered from the Bill Williams River to the lower Colorado River each year and a quantification of evaporation from the surface of Alamo Lake. Criteria for flood control included the number of days water would be held in the flood-control pool portion of Alamo Lake capacity and the maximum percentage of flood control space used. The evaluation criteria for fisheries included percentage of time Alamo Lake would be in an elevation range that maximized the amount of lake less than 6 m (19.7 ft) deep, the optimal range for largemouth bass spawning, and the percentage of time the lake-surface elevation would fluctuate more than 5 cm (2 in.) per day during March through May, factors affecting bass spawning success (Stuber and others, 1982).

Because riparian woodland plants, such as cottonwood and willow, require water throughout the growing season and less water while dormant, evaluation criteria for riparian habitat included percentage of time that there would be sufficient water in Alamo Lake to make a release ≥25 cubic feet per second (ft^3/s) November through January each year, percentage of time there would be sufficient water for releases of ≥40 ft^3/s February through April and in October, and percentage of time there would be sufficient water for releases of ≥50 ft^3/s May through September. For planning and modeling purposes, the growing (or nondormant) season for cottonwood and willow was defined as February through October on the basis of qualitative field observations. Increased water use was assumed during the hotter months of May though September. Determination of dormancy in the field has been problematic with some trees still fully leaved, some leafless, and some budding-out in December and January. To reduce cottonwood mortality from inundation along the Bill Williams River, high-volume releases would be such that the hydrograph followed a more natural pattern, with rapid increase to maximum, then a long tail to reduce the rate of groundwater decline in flood-plain soils. Also, a dry-out period of >30 days would be provided when discharges of >1,000 ft^3/s would be released for 30 days during the growing season or 60 days during the nongrowing season. In addition, maximum Alamo Lake elevation was considered, with a goal of avoiding raising the lake into previously uninundated pool space to avoid enhancing the establishment of nonnative saltcedar (*Tamarix* spp.) in stands of cottonwood and willow as had occurred at maximum lake elevation in the early 1980s.

Following a review of the performance of several operational alternatives, an alternative that established a "target elevation" above which flood releases would be made and below which releases for base flow in the Bill Williams River would vary by month, with consideration of how extended releases would be made, was selected for recommendation by the BWRCTC.

A comparison of the original authorized schedule of releases from Alamo Dam and the schedule under the revised operating plan are shown in figure 4. This figure graphically shows how reservoir pool space is allocated by the Corps of Engineers with (1) a minimum pool from the bottom of the reservoir up to elevation 1,070 ft with primary purposes of recreation; (2) a pool with water conservation as the primary purpose from elevation 1,070 ft to elevation 1,171 ft; (3) and a pool space operated to control downstream floods from elevation 1,171 ft to the spillway crest at elevation 1,235 ft. Included in figure 4 are original maximum releases from Alamo Dam in each portion of the pool space and the revised schedule of releases based on the revised operation plan.

Following completion and member agency endorsement of recommendations of the BWRCTC (Bill Williams River Corridor Technical Committee, 1994), a process was begun

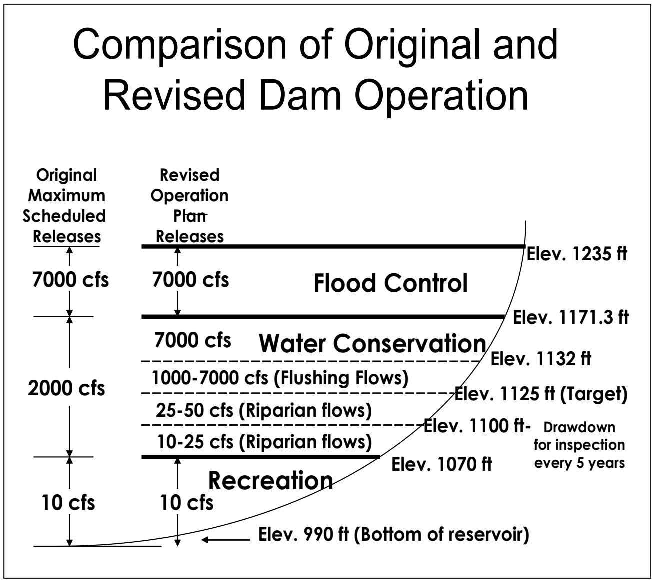

Figure 4. Original and revised dam operation schedules.

to evaluate whether the Corps of Engineers could formally integrate the recommendations into Alamo Dam operations. This process included a feasibility study under the Corps of Engineers authority, a formal Environmental Impact Statement under the National Environmental Policy Act (Public Law 91–190), and a formal biological assessment and biological opinion under the Endangered Species Act (Public Law 93–205). During the course of these studies, the Arizona Game and Fish Department sought inclusion of legislative language to amend the authorized purposes of Alamo Dam, and the purposes were "… modified to authorize the Secretary [of the Army] to operate the Alamo Dam to provide fish and wildlife benefits both upstream and downstream of the Dam. Such operation shall not reduce flood control and recreation benefits provided by the project" (Section 301(b)(1) of the Water Resources Development Act of 1996, Public Law 104–303). Formal adoption of the recommendations of the BWRCTC was completed with revision by the Corps of Engineers of its water control manual (U.S. Army Corps of Engineers, 2003), which provides instruction on operation of Alamo Dam.

Following revision of the water control manual there was renewed interest among stakeholders in developing a monitoring and adaptive management strategy, the need for which was recognized during the planning phase. In 2003, a new Memorandum of Understanding was signed reaffirming the intent of the renamed BWRCSC to communicate and collaborate. At that time the City of Scottsdale (owners of Planet Ranch, the location of historical groundwater pumping) and The Nature Conservancy were added as signatories. In July 2002, a Memorandum of Understanding was signed between The Natural Conservancy and the Corps of Engineers at the national level for the Sustainable Rivers Project with Alamo Dam, one of 26 Corps of Engineers-operated dams across the United States, identified in the program.

In March 2005, the BWRCSC held an ecological flow workshop, an element in The Nature Conservancy's Ecologically Sustainable Water Management (ESWM) process, to review river flows needed to sustain native tree species and ecosystem functions for Alamo Dam and the Bill Williams River. The ESWM process, described by Evelyn and Hautzinger (2006), is a framework for developing a recommendation that meets human needs for water use and can maintain or restore the ecological integrity of river ecosystems. As an element of implementing the ESWM modeling, the non-Federal members of the BWRCSC lobbied for additional Congressional appropriation to the Corps of Engineers to support additional technical work. Products of this effort include hydrologic cross sections of the Bill Williams River between Alamo Dam and Lake Havasu and a digital terrain model. These products enabled development of a HEC-river area simulation (HEC-RAS) model (U.S. Army Corps of Engineers, 1995) that permits detailed modeling of the effects of water releases from Alamo Dam. This model is linked to an ecosystem function model (HEC-EFM) (U.S. Army Corps of Engineers, 2008), which is designed to predict the ecological response of analyzed flow regimes on the Bill Williams River. Through this modeling, for example, analysis of the amount and location of river flood plain to be reworked and wetted by various flood-release scenarios can be performed, linking those processes to establishment of seedbeds and germination events for riparian trees species. Such modeling enables planners to analyze potential operating scenarios at a much greater level of detail than formerly possible and to refine operating criteria through adaptive management.

Results and Discussion

The Corps of Engineers has worked with the BWRCSC to implement recommendations in their report (Bill Williams River Corridor Technical Committee, 1994), beginning with the pattern of release of floodwaters in 1993 and in 1995. The need for monitoring to inform adaptive management was stressed during the planning process, although funding has not been consistently available. There is an ongoing effort to develop a monitoring and research strategy to pursue funding to ensure that data collection occurs to track the performance of management strategies through time and in response to major flow events. The effects of implementation of management strategies since 1993 have been investigated. Factors affecting establishment of woody riparian vegetation in response to annual patterns of streamflow on the Bill Williams River were investigated by Shafroth and others (1998). Riparian vegetation response to altered disturbance and stress regimes on the Bill Williams River were reported by Shafroth and others (2002), including comparison to a reference site upstream on the Santa Maria River. These authors report the years of stand establishment for cottonwood, willow, and saltcedar in 5-year time blocks, including an increase in establishment of cottonwood and willow patches in the 1990–1994 time block from the 1985–1989 time block. Woody riparian vegetation response to different alluvial water-table regimes on the Bill Williams River during the 1995–1997 period was reported by Shafroth and others (2000).

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