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Proceedings of the Colorado River Basin Science and Resource Management Symposium

November 18–20, 2008, Scottsdale, Arizona

Coming Together: Coordination of Science and Restoration Activities for the Colorado River Ecosystem

Scientific Investigations Report 2010–5135

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

Cover photographs—

Front. **Colorado River in Grand Canyon** (*courtesy of the Water Education Foundation*)

Back (clockwise). **Children with fish** (*San Juan River Basin Recovery Implementation Program*)
Hoover Dam (*Bureau of Reclamation*)
Humpback chub (*George Andrejko, Arizona Game and Fish Department*)
Raft and rafters (*Jeff Sorensen, Arizona Game and Fish Department*)

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Coming Together: Coordination of Science and Restoration Activities for the Colorado River Ecosystem

Edited by Theodore S. Melis, John F. Hamill, Glenn E. Bennett, Lewis G. Coggins, Jr., Paul E. Grams, Theodore A. Kennedy, Dennis M. Kubly, and Barbara E. Ralston

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Executive Summary: Future Challenges for Science and Resource Management of the Colorado River

By John F. Hamill[1]

Introduction

Since the 1980s, four major science and restoration programs have been developed for the Colorado River Basin to address primarily the conservation of native fish and other wildlife pursuant to the Endangered Species Act (ESA). The programs are listed below in the order in which they were established.

- Recovery Implementation Program for Endangered Fish Species in the Upper Colorado River Basin (commonly called the Upper Colorado River Endangered Fish Recovery Program) (1988)
- San Juan River Basin Recovery Implementation Program (1992)
- Glen Canyon Dam Adaptive Management Program (1997)
- Lower Colorado River Multi-Species Conservation Program (2005)

Today, these four programs, the efforts of which span the length of the Colorado River, have an increasingly important influence on water management and resource conservation in the basin. The four efforts involve scores of State, Federal, and local agencies; Native American Tribes; and diverse stakeholder representatives. The programs have many commonalities, including similar and overlapping goals and objectives; comparable resources and threats to those resources; and common monitoring, research, and restoration strategies. In spite of their commonalities, until recently there had been no formal opportunity for information exchange among the programs. To address this situation, the U.S. Geological Survey (USGS) worked in coordination with the four programs and numerous Federal and State agencies to organize the first Colorado River Basin Science and Resource Management Symposium, which took place in Scottsdale, AZ, in November 2008. The symposium's primary purpose was to promote an exchange of information on research and management activities related to the restoration and conservation of the Colorado River and its major tributaries.


Jeff Sorenson, Arizona Game and Fish Department

The Colorado River from Deer Creek overlook in Grand Canyon National Park, Arizona. Four collaborative management programs span the length of the Colorado River. Working in different parts of the basin, each program seeks to conserve or restore species listed under the Endangered Species Act and meet water and hydropower demands.

A total of 283 managers, scientists, and stakeholders attended the 3-day symposium, which included 87 presentations and 27 posters. The symposium featured plenary talks by experts on a variety of topics, including overviews of the four restoration programs, water-management actions aimed at restoring native fish habitat, climate change, assessments of the status of native and nonnative fish populations, and Native

[1] U.S. Geological Survey, Southwest Biological Science Center, Grand Canyon Monitoring and Research Center, Flagstaff, AZ 86001.

American perspectives. Intermixed with plenary talks were four concurrent technical sessions that addressed the following important topics:

1. Effects of dam and reservoir operations on downstream physical and biological resources
2. Native fish propagation and genetic management and associated challenges in co-managing native and nonnative fish in the Colorado River
3. Monitoring program design, case studies, and links to management
4. Riparian system restoration, monitoring, and exotic species control efforts

In her opening remarks, Kameran Onley, then U.S. Department of Interior's Acting Assistant Secretary for Water and Science, encouraged better coordination and information sharing among the various recovery and restoration programs. She recounted the history of water management in the basin and emphasized the complex challenge of balancing competing societal needs such as water delivery, hydropower generation, and natural resource protection. Ms. Onley also underscored the importance of independent scientific research as a critical ingredient in the decisionmaking process. In closing, she asked the "USGS to provide recommendations on how science and restoration efforts could be enhanced collectively through better basinwide cooperation and integration." Today, Ms. Onley's request still seems relevant as the Obama Administration considers water, energy, and environmental priorities for the Colorado River Basin.

Anne Phillips, USGS

Eight hydroelectric generation units make up the the powerplant at Glen Canyon Dam. The Department of the Interior balances competing societal needs for water, power, and environmental protection.

San Juan River Basin Recovery Implementation Program

Operated by the Navajo Nation, the fish passage at the Public Service Company of New Mexico Weir in the San Juan Basin provides educational opportunities for local students.

It is difficult to distill a 3-day conference to a few pages of an executive summary, so the following is an attempt to highlight the most compelling issues and themes that emerged from this first symposium. These highlights are drawn not only from the papers that follow (a third of the papers presented at the symposium), but also from symposium presentations that did not result in papers.

Ms. Onley's opening remarks were followed by overviews of each of the four Colorado River Basin restoration programs, which were provided by program leaders. All four programs focus on meeting ESA compliance requirements and, in the case of the Glen Canyon Dam Adaptive Management Program, the 1992 Grand Canyon Protection Act (GCPA). All four programs are designed to conserve or restore endangered species and mitigate the impacts of existing and new water-development and hydropower projects. Each program has implemented an impressive list of actions to conserve native fish, including extensive efforts to control nonnative fish that compete with or prey upon native fish. Other efforts include the construction of fish ladders to expand the range of native fish, the installation of fish screens on irrigation diversions, the acquisition of flood-plain habitats, and the restoration of several thousand acres of riparian and marsh habitat. Hundreds of thousands of native fish have been raised in hatcheries and isolated predator-free ponds and stocked in various locations throughout the basin. Some documented evidence of survival and recruitment of the hatchery fish exists, although overall survival rates for hatchery fish generally are very low.

Water resources also are being managed by the programs in order to benefit native fish. The San Juan River Basin Recovery Implementation and Upper Colorado River Endangered Fish Recovery Programs are regulating flows from a variety of Federal reservoirs to more closely mimic a natural hydrograph (reservoir releases are increased to maximize the spring peak). The hypothesis is that a natural flow regime is best suited to native fish recovery. For example, spring releases from Flaming Gorge Dam are timed with high flows from the Yampa River to maximize peak flows in the Green

River near Jensen, UT. Similar flow-management strategies are being employed at the Aspinall Unit—Blue Mesa, Morrow Point, and Crystal Reservoirs—to improve habitat for native fish found in the Gunnison River. Efforts are underway to enhance base flows in the Yampa River and the “15-mile reach,” a segment stretching east of Grand Junction for 15 miles, of the Colorado River with water stored in several upstream reservoirs (for example, Ruedi Reservoir).

Flows from Glen Canyon Dam are being managed to benefit downstream natural, cultural, and recreational resources. The annual release volumes from Glen Canyon Dam are determined by upper Colorado River Basin hydrology and systemwide water storage in combination with downstream water delivery requirements directly tied to the “Law of the River” and the requirements of the 2007 Colorado River Interim Guidelines for Lower Basin Shortages and the Coordinated Operation of Lake Powell and Lake Mead Final Environmental Impact Statement. Monthly and daily flows are designed to generate hydropower at times of peak demand, although diurnal variations have been attenuated since the early 1990s to minimize downstream environmental impacts in Glen Canyon National Recreation Area and Grand Canyon National Park. In addition, since 1996, a series of experimental high flows have been released from Glen Canyon Dam as part of an adaptive strategy intended to restore sandbars in Grand Canyon. The Glen Canyon Dam Adaptive Management Program has also conducted several stable flow tests to benefit humpback chub (*Gila cypha*) and promote a better understanding of how different flow regimes will contribute to meeting program goals.

Populations of native Colorado River fish have responded variably to this extensive suite of recovery actions, although none of the populations have achieved established recovery or restoration goals. While it is difficult to get a complete picture of the population status of native fish on the basis of information presented at the symposium, Colorado pikeminnow (*Ptychocheilus lucius*) have decreased in the Green River Basin and increased in the upper Colorado River. According to the U.S. Fish and Wildlife Service, humpback chub populations have declined in the Yampa River and in the upper Colorado River (Black Rocks and Westwater Canyon). After more than a decade of decline, adult (age 4+) humpback chub in Grand Canyon have increased by about 50 percent since 2001. Populations of razorback suckers (*Xyrauchen texanus*) are being maintained in the lower basin reservoirs and the Green and San Juan Rivers through active stocking programs, and limited natural reproduction and recruitment is evident in some locations.

Assessing the effectiveness of individual recovery or conservation actions is a common challenge for all four of the restoration programs. The implementation of multiple recovery actions in combination with natural ecosystem variability and the long period of time needed to document successful recruitment of native fish species make it difficult to evaluate the success of any individual experiment or management action.


Upper Colorado River Endangered Fish Recovery Program

A biologist holds an adult Colorado pikeminnow (*Ptychocheilus lucius*), an endangered species. Recently, the number of adult fish captured in the upper Colorado River Basin increased from 440 in 1992 to 890 in 2005.

Monitoring is one of the consistent features of science necessary to assess progress in river restoration programs. When coupled with experiments or management actions that purposefully introduce change to the system, monitoring is critical to the assessment of cause and effect relations. This assessment of cause and effect is an important part of the learning process to determine what works and what does not in achieving the restoration objectives of a given program. The importance of monitoring cannot be overstated, yet historically it has not been included consistently in restoration programs. Additionally, when monitoring has been completed, it has often been done qualitatively or anecdotally and not sustained for a sufficient time or intensity to adequately track resource conditions. Several papers were presented on monitoring programs used to track the status of bats, endangered fish, and campsites used by river runners.

Climate Change Impacts

Brad Udall, director of the University of Colorado at Boulder’s Western Water Assessment, spoke about the influence of climate change on the water supply in the Southwestern United States and made one of the symposium’s most compelling presentations. The mean warming of the Southwest is likely to exceed the global mean. In fact, Udall noted that temperatures in the lower Colorado River Basin have increased 2 degrees Fahrenheit (°F) (1.1 degrees Celsius, °C) from 1970 to 2005, which may be the most rapid rate of temperature change for any region in the United States. As the result of higher temperatures, the upper Colorado River Basin will have less precipitation falling as snow, increased evaporative loss, and an earlier peak spring snowmelt. Based on the analysis of multiple models, the scientific evidence suggests that warmer temperatures will reduce the streamflow of the Colorado River. The flow of the river could be reduced by 6 to 45 percent according to the various model projections. Climate change represents a significant challenge for water-resource management in the West because warming may create substantial water-supply shortages in the Colorado River Basin as the region adds population. In contrast, flows and water temperatures in Grand Canyon are linked to the reservoir elevation of Lake Powell. Decreased inflows and increased evaporation from Lake Powell could lead to releases from the warm epilimnion and result in water temperatures in Grand Canyon approaching 30 °C, temperatures similar to pre-dam conditions (William Vernieu, U.S. Geological Survey, oral commun., 2008).

The recent basinwide drought (2000–2007) had markedly different impacts on native fish populations in unregulated sections of the upper Colorado River Basin relative to the regulated section of Grand Canyon. In the Yampa River, the recent drought has been associated with a large increase in nonnative fish populations and a concomitant decrease in native fish populations. From 2000 to 2007, annual peak discharge and base flow in the Yampa River was significantly reduced, and water temperatures were significantly higher. Very low summer base flows may have reduced habitat volume, increasing the potential for competition and predation by nonnative species. Humpback chub declined in the Yampa River during the recent drought. In contrast, the humpback chub population in Grand Canyon increased during the recent drought. From 2000 to 2007, release volumes from Glen Canyon Dam declined to the minimum allowed by law. During this period, rainbow trout (*Oncorhynchus mykiss*) populations declined by 50 percent, and humpback chub populations increased. Water temperatures during this period of low reservoir elevations were as much as 5 °C higher than the 40-year average because withdrawal structures were drawing warm water close to the surface of Lake Powell. Warmwater releases may have allowed for faster growth rates of humpback chub, and reductions in the population of predaceous rainbow trout may have tipped the system in favor of native fish.

Terry Fulp and others (this volume) reported that the Bureau of Reclamation has an active research and development program to evaluate the impacts of climate change on water supplies, water delivery, and power operations in the basin. However, so far there has been no parallel effort to evaluate the likely impacts of prolonged drought and climate change on water quality or the natural and recreation resources in the Colorado River Basin.

Andrew “Frick” Pernick, Bureau of Reclamation

An aerial view of Lake Powell taken in 2004. The white “bathtub ring” indicates how much the water level dropped as the result of a drought that began in 2000.

The Ongoing Threat of Invasive Species

The ongoing threat from the more than 60 nonnative species present in the Colorado River represents one of the most serious challenges to achieving the native fish goals of each of the four restoration programs. A large body of researchers concludes that the establishment of nonnative fish in the Southwest is the primary cause of the deteriorating status of native fish in the region and prevents their recovery (see Clarkson and Marsh, this volume). However, each of the restoration programs is attempting to promote the recovery of native fish while maintaining politically and economically important nonnative sport fisheries.

Numerous papers were presented that document how nonnative fish threaten the long-term sustainability of native fish populations throughout the Colorado River Basin. Kevin Bestgen of Colorado State University and Angela Kantola of the U.S. Fish and Wildlife Service reported significant declines in the endangered humpback chub in the Yampa River associated with dramatic increases in smallmouth bass (*Micropterus dolomieu*) populations in that same river. Michael Yard and others (U.S. Geological Survey, oral commun., 2008) reported that rainbow and brown trout (*Salmo trutta*) prey on endangered humpback chub in Grand Canyon and estimated that more than 20,000 chub would have been consumed by the trout removed as the result of their study. Lewis Coggins and Michael Yard (this volume) reported success in reducing rainbow trout populations in experimental reaches of the Colorado River in Grand Canyon by using intensive electrofishing during a 4-year period.

Robert Clarkson and Paul Marsh (this volume) concluded that segregating native and nonnative fish is the only viable tactic to conserve and recover imperiled warmwater native species in the Gila River Basin in Arizona. They described several projects involving the construction of instream barriers to prevent upstream fish migrations in conjunction with chemical eradication of nonnative fish that were effective at restoring native fish on several small streams. Unfortunately, the authors noted that this type of approach is not technically

Nonnative fish like the northern pike (*Esox lucius*), a voracious predator, are a threat to native fish populations throughout the Colorado River Basin.

or politically feasible in large drainage networks that also support nonnative sport fisheries.

A new invasive species, the quagga mussel (*Dreissena bugensis*) was found in Lake Mead in January 2007 and had spread to more than 30 Colorado River lakes and reservoirs by the end of 2008 (Nalepa, this volume). Quagga mussels are filter feeders, and when they attain high densities in an ecosystem they can dramatically alter water quality and food web structure, including reducing fish populations. Quagga mussels are not expected to attain high densities in riverine sections of the Colorado River Basin (Nalepa, this volume), but they are expected to attain high densities in reservoirs of the Colorado River Basin where important sport fisheries may be affected. Quagga mussels may impact downriver ecosystems by changing the water quality (that is, dissolved nutrients, phytoplankton, zooplankton) of water released from these reservoirs.

Other Resource Issues of Interest

John Schmidt (this volume), a geoscientist with long experience working throughout the basin, surveyed the highly varied range of geomorphic responses that have occurred following dam construction in reaches of the Colorado River and its tributaries, and noted that some reaches have developed significant sediment deficits while other reaches have experienced surpluses. His plea was for decisionmakers to think more strategically and at a more regional scale about the various restoration (or as he phrased it "rehabilitation") program objectives currently being pursued—at substantial cost and with varied successes—and consider in a more integrated way how costs and benefits might be reasonably and efficiently balanced. He asked two compelling questions:

1. What environmental management goals ought to be established for each part of the basin?

2. Should decisions about goals be made at a segment scale by local stakeholders or at a watershed scale by regional or national interests?

Schmidt's assessment suggested that there may be more "bang for the buck" by focusing rehabilitation efforts on the less perturbed parts of the upper basin but noted that currently most of the funding is being directed at efforts below Lees Ferry (Glen Canyon Dam Adaptive Management and Lower Colorado River Multi-Species Conservation Programs). As Schmidt pointed out, there is no regional process for the Colorado River Basin by which the goals of each rehabilitation program are compared nor is there consideration of the tradeoffs between rehabilitation efforts and the level of recovery.

Christopher Konrad's presentation (this volume) provided an overview of several site-based river restoration projects outside of the Colorado River Basin that are currently being evaluated by The Nature Conservancy in collaboration with the USGS. Konrad's presentation offered some perspectives and hope for moving from site-based to basin-scale river conservation on the basis of lessons from several projects he evaluated. One of Konrad's main observations and conclusions is that integrating dam operations with other types of river management, such as flood-plain land use and water quality throughout a basin, can better conserve river ecosystems and align conservation with human welfare. He acknowledged that basin-scale coordination is difficult, controversial, and time consuming to implement. He concluded that integrated management depends on an alliance of stakeholders with shared ecological goals who are willing to work together rather than simply to comply with the regulatory requirements applicable to their individual site.

In his talk titled "Changing the Law-Science Paradigm for Colorado River Restoration," University of Utah law professor Robert Adler questioned whether it is possible to meet the economic goals of water law and development and the environmental goals of the Endangered Species and Grand Canyon Protection Acts fully and simultaneously (Adler, this volume). He acknowledged that one possibility is that more time is needed to study and fine tune restoration programs until success is achieved. Another more sobering possibility is that the current "law-science paradigm" seeks impossible results. In other words, it is impossible to achieve the goals of each of the programs within the existing legal frameworks. Adler challenged the audience to consider a full range of possible alternatives to the existing "law-science paradigm" that underlies each of the current programs. One of his suggested alternatives included the idea for shifting dependence on large reservoirs for water storage to a variety of off-channel options, such as storing more of the river's flow in aquifers where underground storage might be available.

The barriers to effective Native American participation in Federal restoration programs were also discussed on the basis of the experience of Tribal participants active in the Glen Canyon Dam Adaptive Management Program (Dongoske and others, this volume). Kurt Dongoske, who represents the Zuni Tribe, and his co-presenters, members of the Hualapai and Southern Paiute Tribes, argued that heavy reliance on Western science has the unintended effect of disenfranchising participating Native Americans. The authors concluded that within the Western science perspective, Native American perspectives of the ecosystem are delegitimized and marginalized in favor of scientific knowledge. Additionally, cultural differences in communication and differences in educational backgrounds between Tribal representatives and other stakeholders act as barriers to Tribal participation. For example, the sometimes argumentative nature of the exchanges that take place during meetings is uncomfortable for Tribal representatives and limits their participation. The authors assert that to achieve a program that integrates Native American perspectives, program leaders must embrace a paradigm shift that places traditional knowledge of ecosystems on an equal footing with Western science. The development of a stronger social science component of the Glen Canyon Dam Adaptive Management Program would be a first step toward this paradigm shift.

Kirk Emerson (this volume) wrapped up the symposium with her summary talk on "The Promise and Peril of Collaboration in the Colorado River Basin," addressing the potential values of collaboration and the difficult challenges associated with maintaining vital collaborative partnerships. One of the challenges highlighted was the peril of institutionalism for longstanding programs, which includes process fatigue and weakened commitment. Ms. Emerson noted that the jury is still out on large-scale ecosystem restoration programs, but concluded that adaptive management approaches are essential because there are no other alternatives for dealing with complex natural systems and the management challenges they face. Emerson urged the new Obama Administration to embrace the principles of environmental conflict resolution codified in a 2005 policy memorandum issued by the Office and Management and Budget and the Council of Environmental Quality.

Conclusion

The preceding discussion highlights the broader and perhaps more provocative topics that were discussed during the first Colorado River Science and Resource Management Symposium. In conclusion, it seems appropriate to return to the request from Ms. Onley to provide some thoughts on how science and restoration efforts might be enhanced collectively through better basinwide cooperation and integration.

From a coordination perspective, the hope was that the exchange of information that occurred at the 2008 symposium would improve the effectiveness of the programs both individually and collectively. Responses to the conference generally were very positive. The general conclusion was that the symposium provided an excellent forum for information exchange among individuals working on similar issues in different parts of the basin. As this document was being completed, preliminary plans to sponsor a second symposium in the fall of 2011 or winter of 2012 were underway as a means of promoting additional basinwide coordination and cooperation. The intent of the various program sponsors at the next symposium is to expand the scope and address environmental issues associated with the Colorado River in Mexico.

Determining the appropriate level of integration among the restoration programs is a more complicated question. All four programs have evolved independently, which probably has contributed to their current successes and broad agency and stakeholder support. In addition, the large geographic scope of the basin and the diversity of stakeholders warrant maintaining several distinct programs. As such, a suggestion to merge the current programs is not one of the outcomes of the first symposium. It is worth noting, however, that the combined annual cost of the four programs is about $40 million per year and is projected to be nearly $1 billion over the expected lives of the programs. The cost of the four programs, along with several significant basinwide challenges that transcend program boundaries such as climate change and invasive species, suggests that it is time to consider developing a broader framework to guide the overall effort. Although merging the four programs is not suggested, some form of an overarching framework and independent science organization would be useful to

- establish some fundamental science practices to guide overall restoration efforts throughout the basin,
- conduct regional-scale analyses and assessments of the status of important resources,
- establish indices of ecosystem health and develop the necessary database to monitor those indices, and
- serve as a clearing house for reports and information on the best available management practices.

Such a framework also would facilitate the kind of basinwide assessments that were advocated by Konrad and promote a more effective balance between environmental and water-supply objectives. An overarching framework also would allow for setting basinwide priorities and conducting basinwide tradeoff analyses to ensure limited funds are spent on the highest priority resources with the best potential for restoration, as advocated by Schmidt.

Some may argue that such a proposal goes beyond the compliance requirements of the ESA or GCPA, and that may be true; however, such steps may also lead in a direction toward what is needed—a more sustainable and effective science-based conservation effort throughout the Colorado River Basin. Examples exist where the current restoration programs have exceeded the minimum compliance requirements to head off future problems. Most notably, the goals of two of the upper basin recovery programs go beyond meeting basic Section-7 ESA requirements and seek instead to achieve full recovery of the endangered fish. The Lower Colorado River Multi-Species Conservation Program has an objective of avoiding the listing of a variety of candidate and sensitive species. This same kind of forward-looking, broader-scale approach is now needed to ensure a more integrated, adaptable overall effort. With nine national park units and several national wildlife refuges in the area and large numbers of threatened and endangered or sensitive species dependent on the Colorado River, the importance of maintaining a healthy Colorado River ecosystem is unlikely to go away. As Emerson reminded us in her presentation, meeting the environmental challenges in the Colorado River Basin in the face of increasing water demands and decreasing water supplies will stress the existing restoration programs and demand new approaches. A long-term commitment to rely on consistent monitoring and sound science will be one of the keys to an effective, sustainable conservation effort throughout the basin.

Acknowledgments

Many people and agencies deserve thanks. First, our many sponsors who are noted on the cover of these proceedings: Bureau of Reclamation, National Park Service, U.S. Fish and Wildlife Service, U.S. Geological Survey, the Colorado River Fish and Wildlife Council, and the four programs noted above. These agencies and organizations generously provided funding and staff assistance, but most importantly they encouraged and supported their staffs' attendance at the symposium. Also, the scores of presenters who worked very hard to prepare excellent talks, posters, and manuscripts deserve special recognition, because without their collective support this conference simply would not have happened. The many session chairs from various agencies did an excellent job of recruiting speakers and poster presentations and then reviewing manuscripts for publication in these proceedings. Special thanks go to the members of the Program Steering Committee who helped develop the themes for the conference and identify prospective speakers. The outreach and communication staff from the various sponsor agencies did an excellent job of promoting the symposium. Finally, I would like to acknowledge the invaluable support from Water Education Foundation, especially Rita Sudman and Sue McClurg, who handled the overall planning, registration, and logistics for the event and made all our lives so much easier.

Special acknowledgment is owed to Theodore S. Melis, Deputy Chief, and Lara Schmit, Communications and Outreach Coordinator from USGS's Grand Canyon Monitoring and Research Center, who did most of the heavy lifting involved with coordinating this event and overseeing preparation of the symposium proceedings, including the independent peer-review process. Special thanks also go to the members of the Technical Committee who organized and chaired the technical sessions convened during the symposium and later contributed editorial oversight in the publication of these proceedings: Theodore S. Melis, Paul E. Grams, Theodore A. Kennedy, Lewis G. Coggins, Jr., Dennis M. Kubly, Barbara E. Ralston, and Glenn E. Bennett. The dedicated support of the session chairs and their commitment to detail were key to making this first symposium of its kind a success.

Paul Alley, USGS

A long-term commitment to rely on consistent monitoring and sound science will be one of the keys to an effective, sustainable conservation effort throughout the basin.

Plenary Sessions

Colorado River Basin Science and Resource Management Symposium

November 18–20, 2008, Scottsdale, Arizona

Remarks prepared for delivery by

Kameran A. Onley,[1] Acting Assistant Secretary for Water and Science,

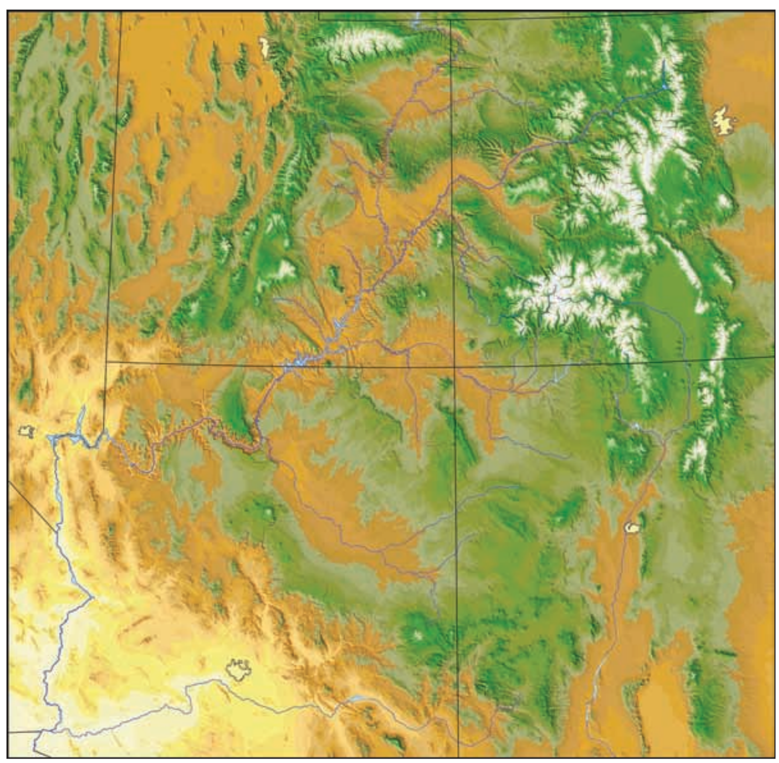
U.S. Department of the Interior

It is a pleasure and quite exciting to be here, at what is the first conference designed to share information among the various environmental programs underway here in the Colorado River Basin. I would like to commend the organizers of this conference: The goal of better coordination of scientific information across programs in the Colorado River Basin is a valuable one, though we should not minimize the difficulty and barriers to achieving better information sharing and integration.

Over the past century, there have obviously been incredible changes here in the Colorado River Basin. We have tamed the Colorado River, tapped its hydropower potential, irrigated the Southwest's vast agricultural lands, and provided water to the major urban areas of the West: Denver, Las Vegas, Phoenix and Tucson, Los Angeles and San Diego. We manage water supplies to meet our water-quantity and water-quality obligations to Mexico under the 1944 treaty and its implementing agreements. We have also protected some of the most magnificent landscapes on Earth: from the headwaters of the Colorado and Green Rivers to Mexico, the Colorado flows through and along unique landscapes, the Black Canyon, Glen and Grand Canyons, Lake Powell and Lake Mead recreation areas, and refuges.

Additionally, the ecological value of the river and its importance to Native American Tribes have gained recognition in recent decades. Today, the Colorado River Basin is intensively managed by the U.S. Department of the Interior (DOI) in partnership with Tribes, States, and many other stakeholders to meet a variety of social, cultural, and ecological demands.

This symposium is specifically aimed at promoting the exchange of information on research and management activities related to the restoration/conservation of the Colorado River. We probably could spend a bit of time discussing and debating whether these efforts are best described as environmental protection, environmental conservation, or



The Colorado River Basin stretches from its headwaters in the Rocky Mountains to the Gulf of Mexico. In the United States, four collaborative management programs—each working in a different portion of the Colorado River Basin—have developed over the past 20 years largely to respond to concerns about endangered species. Shaded relief map created by Barry Middleton, USGS Southwest Geographic Science Team, Flagstaff, Arizona.

[1] Director of U.S. Marine Policy, The Nature Conservancy, Arlington, VA 22203.

perhaps “environmental restoration,” as is used in the title for this conference. Inherent in these various descriptions are statements about goals, values, and objectives.

We have seen different programs and initiatives in the basin—each with its own history, stakeholders, and approaches. From my perspective, gaining a better understanding of the elements that unite these programs and ensuring that accurate, timely scientific information is shared among these programs may be the single most important element that will distinguish between success and failure in coming years and decades—though I do not want to minimize the challenge of coming to agreement or consensus on what success looks like.

Over the past 20 years we have seen incremental development of environmental programs from the headwaters of the basin to the Mexican border. Obviously, many of these efforts have been driven by concerns regarding endangered species:

- Established in 1988, the Upper Colorado River Endangered Fish Recovery Program is a partnership of public and private organizations working to recover these endangered species while allowing continued and future water development.

- Established in 1992 with the signing of the cooperative agreement, the San Juan Recovery Implementation Program is designed to help recover the Colorado pikeminnow (*Ptychocheilus lucius*) and the razorback sucker (*Xyrauchen texanus*) while allowing water development to continue in the San Juan River Basin.

- The Glen Canyon Dam Adaptive Management Program was established in 1997 to assist the Department to meet the goals and objectives of the Grand Canyon Protection Act.

- And most recently in 2005, the Department formally established the Lower Colorado River Multi-Species Conservation Program, a 50-year, nearly $1 billion investment to enhance habitat along the lower Colorado River to both conserve species that are currently endangered and threatened and to help reduce the potential for further additional listings in the future.

Our expectation is that the effectiveness of programs individually and collectively will be enhanced by the information that is provided and the relationships that emerge from this symposium.

So, we now have these programs—each working in a portion of the Colorado River Basin—and one of the fundamental questions and challenges we face is the integration and coordination of the scientific information that will help guide the course of these efforts. In spite of the commonalities among the programs, until now there has been no formal opportunity for information exchange among programs. This symposium is specifically aimed at promoting the exchange of information on research and management activities related to the restoration/conservation of the Colorado River in the United States.

Some of the most significant challenges that these programs face transcend program boundaries. A recent example of a transboundary issue is the quagga mussel (*Dreissena bugensis*) invasion; the mussel is an invasive species that was found in Lake Mead in early 2007 and has spread throughout many portions of the basin and the West. Also of grave concern is the spread of zebra mussels (*Dreissena polymorpha*). Both invasive organisms threaten native species and water-supply systems. Climate change is predicted to have a profound impact on water supplies and water quality and significantly alter ecological processes.

Restoration and recovery strategies need to anticipate and adapt to these basinwide challenges and what is working today may not work under tomorrow’s climate regime and biological environment. Trying to determine whether proposed goals can be achieved in the face of predicted hydrologic changes that may come from both climate change and continued consumptive uses is a significant challenge.

These programs are also linked by goals that require recovery throughout the basin. Under the current recovery goals, achieving demographic criteria and minimizing and removing threats (in order to meet down-listing and delisting requirements) are expected to be accomplished through these various programs.

Our expectation is that the effectiveness of programs individually and collectively will be enhanced by the information that is provided and the relationships that emerge from this symposium. Perhaps future symposia will be expanded to include cross-border issues within Mexico at the Colorado River delta and will include more involvement from international partners.

The Difficulty of Coordination and Integration

Anyone who has worked on large-scale ecosystem efforts knows the challenges that come with working across agency, political, and policy boundaries. Any number of fundamental questions and complications are evident. How do the various programs gather, evaluate, and publish scientific information? How are the conflicting protocols, objectives, and procedures—and statutory missions—to be addressed among the agencies? How do we integrate the peer-review of emerging science into public processes such as National Environmental Policy Act (NEPA) studies and Endangered Species Act (ESA) consultation? How do we ensure continued participation by experts while integrating new researchers and new methods into research efforts?

I would ask each of you to think about the challenges of information sharing just within your own organization and then expand that difficulty across the areas that will be discussed over the next 3 days. Think of it: coordination within offices and within agencies is quite a challenge. Take that task and broaden the goal to achieve improved information sharing between researchers, universities, agencies, States, Tribes, and the broader members of the interested public. Quite a challenge. Then, on top of all of those inherent organizational challenges—add the destabilizing complexity of global climate change and the effects that are anticipated for this most arid part of our Nation. It is clear that we all have a stake in improved coordination and effective information sharing.

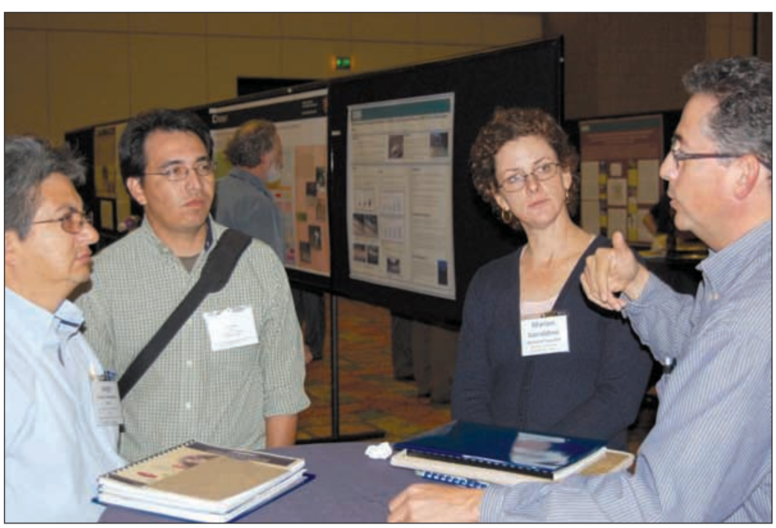
Many fields of scientific study face the same challenge of integration and coordination. In emerging areas of nanotechnology and biotechnology research, we have seen institutes formed between government agencies, universities, and private corporations to achieve better efficiencies and effective research. Some of these institutes are physical—some are virtual—but a key objective is always improved information sharing.

Here in the Colorado River Basin, we cannot simply form a Colorado River Institute and assume that the coordination we need will emerge. Instead, we will need more efforts such as this conference, continued investment in research and monitoring, and continued flexibility through adaptive management to take advantage of the scientific advances in ecosystem understanding. As we go forward, I believe that the need for independent science research from the U.S. Geological Survey (USGS) and others will remain essential. Scientific efforts need to operate separately from management actions and political influence. At the same time, we must encourage an integrative science approach to understanding and managing entire watersheds or ecosystems. Mutual respect for the scientific process—we hope—will lead to increased cooperation among diverse—at times, competing—groups of stakeholders. We have seen in a number of settings the importance of information from independent scientific research to facilitate sound policy and decisionmaking (e.g., the USGS role in polar bear research).

Outcomes of this Conference

To ensure that results of this symposium are factored into DOI management of the Colorado River Basin, I have asked the USGS to provide recommendations to me on how science and restoration efforts could be enhanced collectively through better basinwide cooperation and integration. In coming days, I will ensure that these recommendations are passed along to President-elect Obama's transition team for its consideration.

As you all are well aware, the Secretary of the Interior has a unique connection to the Colorado River—based on the unique history of the development of this basin. The Secretary has a very difficult task of balancing competing societal needs within the Colorado River Basin (a good example is water delivery, hydropower generation, and natural resource protection). In the talks that follow, the agenda will focus on how an adaptive management approach is being used to integrate science, stakeholder concerns, and water and resource management decisions, and how we can more effectively use the scientific knowledge across program lines.



Symposium participants at the Colorado River Basin Science and Resource Management Symposium, which took place November 18–20, 2008, in Scottsdale, Arizona.

Closing Observations

John Wesley Powell is certainly one of the towering figures in Colorado River Basin history. He is known as a one-armed Union veteran of the Civil War, who survived his 1869 expedition down the Green and Colorado Rivers. He later became the second Director of the USGS. Powell was known for his attempts to categorize and integrate new information—to create scientific order from new facts. Late in a life driven by scientific curiosity and exploration, he made a number of political proposals that were informed by his western explorations and Colorado River Basin experiences. One proposal—or recommendation—that he made in 1889 was to organize some of the new Western States along hydrographic basins—rather than arbitrary political lines. Powell's view was that organizing political boundaries by watersheds would allow for economic unity—and productivity—within basins. Conflict, litigation, and other costly inefficiencies would be lessened as the decisionmaking in upstream and downstream areas of a basin were integrated. Science and reason—integrated into political governance. While his advocacy on this point did not succeed, I think his observations are still quite compelling.

Efforts such as this conference—cooperative efforts to advance scientific coordination within this watershed—the Colorado River Basin—are entirely consistent with Powell's goals to advance scientific understanding and to improve societal decisionmaking. I thank you for your efforts and applaud your goal of better coordination and information sharing among the programs in the basin.

Overview of the Colorado River Basin Collaborative Management Programs

By David Campbell,[1] Scott Durst,[1] Angela T. Kantola,[2] Dennis M. Kubly,[3] Robert T. Muth,[4] John Swett,[5] and Sharon Whitmore[1]

Abstract

Today, four collaborative management programs stretch the length of the Colorado River. Each of the four programs seeks to conserve or restore species listed under the Endangered Species Act, particularly endangered fish, while continuing to meet water and hydropower demands. The Recovery Implementation Program for Endangered Fish Species in the Upper Colorado River Basin was initiated in 1988 and was the first Colorado River collaborative management program. The San Juan River Basin Recovery Implementation Program was established in 1992 and was followed by the Glen Canyon Dam Adaptive Management Program in 1997 and the Lower Colorado River Multi-Species Conservation Program in 2005.

All of the Colorado River collaborative programs involve multiple stakeholders, which, depending on the program, can include representatives of Federal and State resource management agencies, Colorado River Basin States, Native American Tribes, environmental groups, recreation interests, water-development proponents, and energy and power users. The programs coalesced not only because the natural systems they were dealing with were complex, as were the needs of the species they were seeking to recover, but also because no one party could resolve the challenges independently or win a lasting victory through legal or legislative action.

The four program descriptions presented here include information on program history and goals, geographic scope, participants, resources of concern, activities, and progress. The programs discussed here are at different stages of development, which is reflected in the following descriptions.

[1] San Juan River Basin Recovery Implementation Program, U.S. Fish and Wildlife Service, New Mexico Ecological Services Field Office, Albuquerque, NM 87113–1001.

[2] Upper Colorado River Endangered Fish Recovery Program, U.S. Fish and Wildlife Service, PO Box 25486, Denver Federal Center, Lakewood, CO 80225.

[3] Upper Colorado Region, Bureau of Reclamation, 125 South State Street, Salt Lake City, UT 84138.

[4] U.S. Fish and Wildlife Service, Bozeman Fish Technology Center, 4050 Bridger Canyon Road, Bozeman, MT 59715.

[5] Lower Colorado River Multi-Species Conservation Program, Bureau of Reclamation, PO Box 61470, Boulder City, NV 89006.

Introduction

The Colorado River provides water for more than 27 million people in the United States and more than 3.5 million acres of agricultural land (U.S. Department of the Interior, 2007). A vast system of dams and reservoirs is in place to manage the river's valuable waters; there are 22 major storage reservoirs in the Colorado River Basin and 8 major out-of-basin diversions (Pontius, 1997).

Conflict attached itself early to Colorado River water and its management. In 1922, the seven Colorado River Basin States signed the Colorado River Compact, which Congress ratified the same year, allocating the Colorado River's water resources among the seven basin States. The compact divides the river basin into two parts: the upper division (Colorado, New Mexico, Utah, and Wyoming) and the lower division (Arizona, California, and Nevada). The compact allowed for the development of water resources by the Federal government and made possible widespread irrigation. However, Arizona refused to ratify the agreement until 1944 and disputed the water allotments until the United States Supreme Court upheld the allocations in 1963.

The construction of dams in the Colorado River Basin altered the historical flow and temperature patterns of the river, which has affected the habitat and reproductive success of native fish. However, early European settlers altered the Colorado River's fish community well before the construction of mainstem dams through the introduction of nonnative fish. For more than 100 years, nonnative fish—from sports fish to escapees from aquaria—have been intentionally and unintentionally stocked in the Colorado River (Mueller and Marsh, 2002). Nonnative species are potential predators of and competitors with native species. Today, because of the range of nonnative species found in the Colorado River, nonnative fish may negatively interact with native species under virtually any temperature regime and in any habitat (Gloss and Coggins, 2005).

Four species of Colorado River fish are currently listed as endangered under the Endangered Species Act (ESA): Colorado pikeminnow (*Ptychocheilus lucius*), razorback sucker (*Xyrauchen texanus*), bonytail (*Gila elegans*), and humpback chub (*Gila cypha*). The Colorado pikeminnow and humpback chub were both added to the Federal list of endangered species in 1967, while the bonytail and razorback sucker were listed in 1980 and 1991, respectively.

Efforts to protect declining native fish under Section 7 of the ESA resulted in entrenched conflicts. For example, in the upper Colorado River Basin, the Colorado River Water Conservation District filed suit against the U.S. Fish and Wildlife Service (Service) in the late 1970s, challenging the listing of the Colorado pikeminnow and the humpback chub. Because the Service had taken action pursuant to the ESA that would have prevented more water development along the river, the river district accused the agency of damaging property rights and hindering economic development. In 1983, water developers challenged the scientific basis for agency-proposed minimum streamflow standards.

It became clear by the early 1980s that conflicts between resource protection and resource development in the upper Colorado River Basin were unlikely to be resolved through litigation or legislative action. The parties recognized that an adversarial approach was "unlikely to result in progress toward recovery of the listed species and could lend a measure of uncertainty to future water resource development in the upper basin" (U.S. Fish and Wildlife Service, 1987, p. 1–6). As a result, the parties sought to accommodate their competing demands through discussion and negotiation under the auspices of the Upper Colorado River Basin Coordinating Committee, which was formally established in 1984 (U.S. Fish and Wildlife Service, 1987). The Coordinating Committee and its various subcommittees included the Service, Bureau of Reclamation (Reclamation), and the States of Colorado, Wyoming, and Utah, and also representatives of water users, proponents of water development, and conservation organizations.

Through discussion, the members of the Coordinating Committee determined that both the biological needs of the endangered species and the hydrology of the upper basin were "exceedingly complex," requiring a systematic approach to achieve native fish conservation and continued water development in the upper basin (U.S. Fish and Wildlife Service, 1987). In the end, the group concluded that a comprehensive program was needed to implement the broad array of measures necessary to "not only preserve the listed species but to ensure their full recovery and eventual delisting" (U.S. Fish and Wildlife Service, 1987, p. 1–6). Thus, the Recovery Implementation Program for Endangered Fish Species in the Upper Colorado River Basin (also known as the Upper Colorado River Endangered Fish Recovery Program; hereafter, UCRRP)—the first Colorado River collaborative management program—was initiated in 1988. The San Juan River Basin Recovery Implementation Program (SJRIP) was established in 1992 and was followed by the Glen Canyon Dam Adaptive Management Program (GCDAMP) in 1997

Colorado pikeminnow (*Ptychocheilus lucius*)

Razorback sucker (*Xyrauchen texanus*)

Bonytail (*Gila elegans*)

Humpback chub (*Gila cypha*)

George Andrejko, Arizona Game and Fish Department

The four collaborative management programs that focus their efforts on the Colorado River seek to restore species listed under the Endangered Species Act (ESA), particularly endangered fish. The four species of Colorado River fish currently listed as endangered under the ESA are shown above.

and the Lower Colorado River Multi-Species Conservation Program (LCR MSCP) in 2005.

The four collaborative management programs that today span the length of the Colorado River share many of the same antecedents. All four programs were created to conserve or restore species listed as endangered under the ESA, particularly endangered fish, while continuing to meet water storage, delivery, and development needs and hydropower demands. In the case of the GCDAMP, the Grand Canyon Protection Act (GCPA) gives the program's efforts a broader scope in seeking to ensure the long-term sustainability of natural, cultural, and recreation resources found downstream from Glen Canyon Dam in Glen Canyon National Recreation Area and Grand Canyon National Park. All of the programs involve multiple stakeholders which, depending on the program, can include representatives of Federal and State resource management agencies, Colorado River Basin States, Native American Tribes, environmental groups, recreation interests, water development interests, and energy and power users. The programs coalesced not only because the natural systems they were dealing with were complex, as were the needs of the species they were seeking to recover, but also because no one party could resolve the challenges independently or win a lasting victory through legal or legislative action.

Each of the four Colorado River Basin collaborative management programs is described briefly below. The four program descriptions are organized by their location, starting in the uppermost Colorado River Basin and moving downstream, and include information on program history and goals, geographic scope, participants, resources of concern, activities, and progress. Because the four programs came into existence at different times, ranging from 5 to 20 years ago, they are at different stages of development, which is reflected in the following descriptions.

Colorado River Basin Programs

Upper Colorado River Endangered Fish Recovery Program (UCRRP)

Program History

The UCRRP, also known as the Recovery Implementation Program for Endangered Fish Species in the Upper Colorado River Basin, was formally established in January 1988 through a cooperative agreement signed by the Secretary of the Interior; the Governors of Colorado, Wyoming, and Utah; and the Administrator of the Department of Energy's Western Area Power Administration. Water users and environmental organizations signed supporting resolutions. The 1988 agreement provided for a 15-year term for the UCRRP, which was later extended to 2013 and then to 2023. The cooperative agreement grew out of a 3-year process that culminated in a 1987 framework document for the program (U.S. Fish and Wildlife Service, 1987).

Conflicts between the ESA and water development drove the need for the UCRRP. In the 1980s, the Service determined that additional depletion of water from the upper basin would constitute jeopardy to the continued existence of endangered fish. In 1983, the Service proposed minimum streamflows for all habitats occupied by endangered fish in the upper basin (pre-1960 flow levels) and required replacement of depletions on a one-for-one basis. This requirement could have stopped water development in the upper basin, put limits on the use of existing water supplies, and conflicted with existing Federal and State laws that allocate water, resulting in direct conflict among States, water users, Federal agencies, power customers, and environmental organizations.

In order to avoid a head-on collision, the parties sought to accommodate their competing demands through discussion and negotiation under the auspices of the Upper Colorado River Basin Coordinating Committee, which was formally established in 1984 (U.S. Fish and Wildlife Service, 1987). The group concluded that a comprehensive program was needed, and the UCRRP was initiated in 1988 (Wydoski and Hamill, 1991).

Program Goal

The goal of the UCRRP is to recover four endangered fish species—Colorado pikeminnow, humpback chub, bonytail, and razorback sucker—while providing for new water development to proceed in the upper Colorado River Basin.

Geographic Scope

The geographic scope of the UCRRP is the Colorado River Basin upstream from Glen Canyon Dam, excluding the San Juan River subbasin (fig. 1). The focus of the program's attention is the Colorado River and its tributaries in Colorado, Utah, and Wyoming, with the exception of the San Juan River.

Program Participants

The UCRRP is a 10-member partnership among the States of Colorado, Utah, and Wyoming; the Service; Reclamation; National Park Service; Western Area Power Administration; Colorado River Energy Distributors Association; environmental organizations; and water users.

Program Structure and Budget

The UCRRP has five principal elements: (1) habitat management through the provision of instream flows; (2) nonflow habitat development and maintenance; (3) management of nonnative species and sport fishing; (4) native fish stocking; and (5) research, data management, and monitoring. The UCRRP's Recovery Action Plan, a long-range operational plan, is consistent with the 2002 Recovery Goals (U.S. Fish and Wildlife Service, 2002a–d), contains all the actions believed necessary to recover the fish in the upper basin, and is updated annually. Using an adaptive management approach to develop and implement management actions, the UCRRP is able to continually evaluate and revise recovery actions as new information from research and monitoring becomes available and to adapt to changing factors such as the recent years of prolonged drought across the West and proliferation of nonnative fish species.

Coordination and collaboration among UCRRP stakeholders are keys to the UCRRP's success. Each partner fully participates in developing and implementing management actions that will achieve the recovery goals and lead toward delisting of the endangered fish. The UCRRP has three committee levels: a policy-level Implementation Committee; a Management Committee; and three technical committees (Biology, Water Acquisition, and Information and Education). The UCRRP's director and staff coordinate the recovery efforts and serve all of the committees.

The UCRRP's annual budget for fiscal year 2009 was $9.5 million.

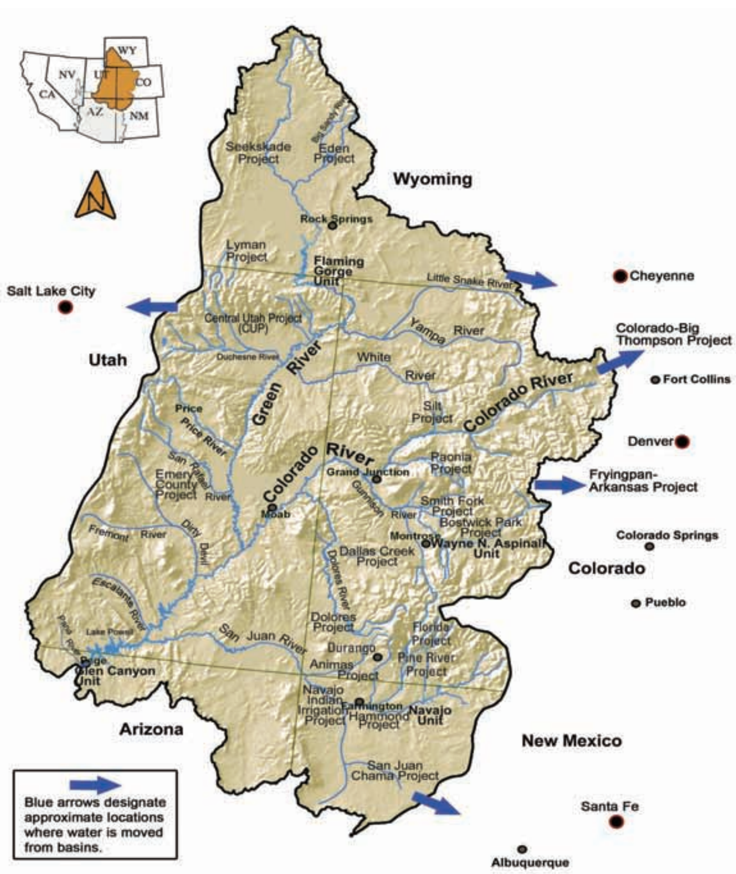


Figure 1. The area served by the Upper Colorado River Endangered Fish Recovery Program. The region includes the Colorado River Basin upstream from Glen Canyon Dam, with the exception of the San Juan River subbasin.

Program Activities

Habitat Management

Habitat management actions focus on identification and provision of instream flows necessary to achieve recovery of the endangered fish. Recovery program partners cooperatively manage water resources to benefit the endangered fish and their habitats in accordance with State water law, individual water rights, interstate compacts, and Federal authorizing legislation. Management is accomplished through a variety of means, including leases and contracts for water supplies, coordinated water releases from upstream reservoirs, participation in reservoir enlargements, efficiency improvements to irrigation systems to reduce water diversions, and re-operation of Federal dams and reservoirs. These water-management actions not only benefit the endangered fish, but also benefit recreational, municipal, and agricultural water users as well.

Operations of five principal reservoirs in Colorado are coordinated to voluntarily release water to enhance Colorado River spring peak flows and improve fish habitat without affecting those reservoirs' yields (fig. 2). Most of these reservoirs also contribute water for late-summer, base-flow augmentation. Construction of seven check structures in the Grand Valley Project Canal System in western Colorado in 2002 has reduced water diversions by 10 to 16 percent. These check structures regulate canal deliveries to meet irrigation demands and help reduce river diversions to keep more water in the river for fish.

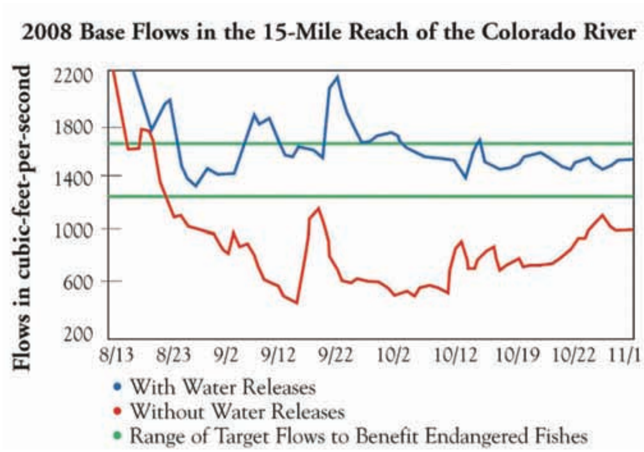


Figure 2. Additional water from upstream reservoirs in Colorado for the "15-mile reach," a segment stretching east of Grand Junction for 15 miles, of the Colorado River in 2008. Averaging 56,000 acre-feet per year since 2000, flows from reservoirs enhance late-summer and fall base flows for endangered fish. The amount of water released in 2008 was the greatest to date, totaling 114,255 acre-feet (Upper Colorado River Endangered Fish Recovery Program /San Juan River Basin Recovery Implementation Program, 2009).

The UCRRP partnered with the Colorado River Water Conservation District and other State and local agencies on a 13,000 acre-foot enlargement of Elkhead Reservoir completed in 2006. The enlarged reservoir provides up to 5,000 acre-feet of permanent water and 2,000 acre-feet of leased water for augmentation of base flows in the Yampa River and about 5,000 acre-feet of water for future growth in Moffat County, CO. The project also creates an improved recreational amenity for the residents of Routt and Moffat Counties and serves as a repository for nonnative sportfish removed from the Yampa River.

Federal reservoirs also provide water for the endangered fish. The Bureau of Reclamation operates Flaming Gorge Dam on the Green River, UT, according to a Record of Decision signed on February 16, 2006, to assist in recovery of the endangered fish (Bureau of Reclamation, 2006). The Aspinall Unit on the Gunnison River in western Colorado is made up of three Federal reservoirs—Blue Mesa, Morrow Point, and Crystal. A draft Environmental Impact Statement on re-operation of Aspinall Unit dams on the Gunnison River to assist in recovery of the endangered fish was released in February 2009 (U.S. Department of Interior, 2009), with a Record of Decision anticipated by 2010.

Habitat Development

The UCRRP restores and maintains habitat for endangered fish by constructing and operating fish passages at diversion dams, constructing and operating fish screens in diversion dam canals to keep fish from entering and becoming trapped, and acquiring and restoring flood-plain habitat to serve primarily as fish nursery areas. All habitat restoration actions are monitored by the UCRRP to evaluate their effectiveness, and management changes are implemented on the basis of evaluation results to further improve habitat conditions.

Fish passages and screens are completed and operational at the Redlands Water and Power Company, Grand Valley Irrigation Company, and Grand Valley Project diversions near Grand Junction in western Colorado, and a passage also is complete at Price-Stubb irrigation diversion. The fish passages provide endangered fish with unimpeded access to about 340 miles of designated critical habitat in the Colorado and Gunnison Rivers. At the Redlands Water and Power Company and the Grand Valley Project Canal System, the passage structures are selective in that when fish reach a holding area at the top, they are removed and sorted, and only native fish are allowed to pass through. Construction of a screen at the Tusher Wash diversion on the Green River is scheduled to begin in 2010.

Flood-plain habitats are being made accessible to all life stages of endangered fish by breaching or removing natural or manmade levees to connect the sites to the river during spring runoff. Restored river habitat also improves sources of food and shelter for other fish, plant, and animal species. The UCRRP has acquired 1,600 acres in Colorado and Utah (19 properties), of which 600 acres (four properties) have been restored. The UCRRP also has restored a total of 2,100 acres owned by the Bureau of Land Management, City of Grand Junction, Colorado Division of Wildlife, Colorado State Parks, or the Service.

Nonnative Fish and Sportfishing

Predation or competition by nonnative fish species is a serious threat to the endangered fish and poses the biggest obstacle to recovery and the greatest long-range management challenge for the UCRRP. Fourteen species or subspecies of native fish occurred historically in the upper basin. Over the past 100 years, more than 50 nonnative fish species have been introduced into the upper basin and now dominate many


Upper Colorado River Endangered Fish Recovery Program

About 2,700 acres of restored flood-plain habitat in the upper Colorado River Basin are managed for all life stages of endangered fish.

fish communities. Currently, northern pike (*Esox lucius*), smallmouth bass (*Micropterus dolomieu*), and other sunfish, including the largemouth bass (*Micropterus salmoides*), are the most problematic nonnative fish and are the principal target species for management.

Management actions of the UCRRP to reduce the abundance of nonnative fish and their impacts to endangered fish recognize the dual responsibilities of State and Federal wildlife agencies to conserve native fish species while providing sportfishing opportunities. Nonnative fish management actions include mechanically removing nonnative fish from rivers, restricting the stocking of nonnative fish, screening of off-river ponds and reservoirs to prevent escapement of fish to rivers, identifying chronic sources of nonnative fish to rivers, changing State bag and possession limits on warmwater sportfish to increase angler harvest, and monitoring the responses of nonnative and native fish to management actions. Where feasible, sportfish removed from rivers are translocated to local off-channel ponds or reservoirs to provide fishing opportunities. Research, monitoring, and adaptive management are used to identify, evaluate, and revise management strategies. Annual workshops are held to further review results of field activities and develop appropriate modifications to the nonnative fish management strategies.

Endangered Fish Propagation and Stocking

Five hatchery facilities produce bonytail and razorback sucker necessary to meet the UCRRP's annual and long-range stocking targets. Broodstocks and propagation of young are managed to maximize the genetic diversity of stocked fish to increase the likelihood that stocked fish can cope with local habitat conditions in the wild. An integrated stocking plan was finalized in 2003 to expedite reestablishment or enhancement of naturally self-sustaining populations and achieve the demographic criteria of the recovery goals (Nesler and others, 2003). Roughly 30,000 razorback suckers and 16,000 bonytails are stocked in the upper Colorado River and Green River systems each year. Survival, growth, and reproduction of stocked fish are monitored to evaluate and improve stocking strategies.

Upper Colorado River Endangered Fish Recovery Program

Some 30,000 hatchery-raised bonytail (*Gila elegans*) are stocked each year in the upper Colorado and Green River systems to reestablish and enhance naturally occurring populations of the fish.

Research, Monitoring, and Data Management

The UCRRP's early emphasis was on research to gather basic life-history information about the endangered fish and determine actions needed for recovery. Research and monitoring now generate information on reproduction, growth, and survival of endangered fish in the wild, and data management systems serve as repositories and analytical tools for that information. Data are used to evaluate and adjust management actions and recovery strategies through adaptive management. The UCRRP uses estimates of the abundance of endangered fish to monitor progress toward achieving the recovery goals.

Progress Toward Program Goals

Nonnative Fish

Over the past 10 years, progress has been made in reducing the abundance of some of the target nonnative fish species in certain rivers of the upper Colorado River Basin. However, a great deal of work remains to identify the methods and levels of management needed to minimize the threat of nonnative fish predation or competition and achieve and maintain recovery of the endangered fish (table 1).

Endangered Fish

Wild populations of Colorado pikeminnow and humpback chub occur in the upper Colorado and Green River systems. These populations have been studied since the 1960s, and population dynamics and responses to management actions have been evaluated since the early 1980s. Hatchery-produced, stocked fish form the foundation for the reestablishment of naturally self-sustaining populations of bonytail and razorback sucker in the upper Colorado and Green River systems. Significant changes in the status of the four endangered fish generally are not detected on a year-to-year basis. Closed-population, multiple mark-recapture estimators for tracking population trends are being used (where possible) in the upper Colorado and Green River systems to derive population point estimates for wild Colorado pikeminnow and humpback chub.

Recovery goals for the endangered fish identify site-specific management actions to minimize or remove threats and establish criteria for naturally self-sustaining populations. A key requirement of the population criteria is no net loss of fish over established monitoring periods. Downward trends in some wild populations of Colorado pikeminnow and humpback chub have been observed during dry weather and low river runoff conditions since 1999. Biologists hypothesize that these declines may be a result of reduced recruitment that can be largely attributed to increases in certain problematic

Table 1. The Upper Colorado River Endangered Fish Recovery Program's efforts to reduce nonnative fish abundance (Upper Colorado River Endangered Fish Recovery Program and San Juan River Basin Recovery Implementation Program, 2009).

River	Species	History and current status
Colorado (112 miles)[a]	Smallmouth bass	• Increases in abundance first observed in 2003; removal began in 2004. • Abundance declined during 2006–2008; more removal passes added in 2007 to increase captures. • Largemouth bass are an emerging problem; catch of young fish has steadily increased since 2004.
Green (198 miles)[a]	Smallmouth bass	• Increases in abundance first observed in 2003; removal began in 2004. • Adult abundance declined over 50 percent throughout much of the Green River during 2004–2006. • Increased efforts in 2007 (continued in 2008) removed as much as 90 percent of the estimated adult population in certain high-concentration areas.
	Northern pike	• Since removal began in 2001, abundance has decreased by more than 90 percent.
Yampa (94 miles)[a]	Smallmouth bass	• Increases in abundance first observed in 2003; removal began in 2004. • Results through 2007 indicated the adult population was declining; however, substantial reproduction occurred in 2006 and 2007. • Average flows in 2008 in the Yampa, Green, and Colorado Rivers appear to have negatively affected reproduction.
	Northern pike	• Abundance steadily increased during the 1980s and 1990s; removal began in 1999. • Removal through 2007 shifted the size to smaller individuals; in 2008, the overall abundance in critical habitat was near its lowest level.

[a] River miles where work occurred in 2008.

nonnative fish and habitat changes associated with the recent drought (Upper Colorado River Endangered Fish Recovery Program and San Juan River Basin Recovery Implementation Program, 2009). The recovery programs are actively implementing and adaptively evaluating management actions to reduce these threats and reverse the downward population trends to achieve and maintain self-sustaining populations. Meanwhile, progress is being made to reestablish specific populations through stocking.

Following are summaries of the currently available information on the status of each species related to the demographic criteria of the recovery goals for the upper Colorado River Basin.

Colorado Pikeminnow

There are two wild Colorado pikeminnow population centers, one in the upper Colorado River system and one in the Green River system, consisting of separate spawning stocks of which juveniles and adults mix. This exchange of fish sets up a population network or metapopulation, with the Green River system being the largest. Abundance of adults in the Green River system declined from 3,100 to 2,300 between 2001 and 2003 (Upper Colorado River Endangered Fish Recovery Program and San Juan River Basin Recovery Implementation Program, 2009). Reproduction in 2006 was strong, and biologists reported a sixfold increase in the number of young-of-year (less than 1-year-old) Colorado pikeminnow captured in the Green River in the summer of 2009 compared to the average catch rate during the previous 18 years (Upper Colorado River Endangered Fish Recovery Program, 2010). Abundance of adults in the upper Colorado River system increased from about 440 in 1992 to 890 in 2005 (fig. 3) (Upper Colorado River Endangered Fish Recovery Program and San Juan River Basin Recovery Implementation Program, 2009).

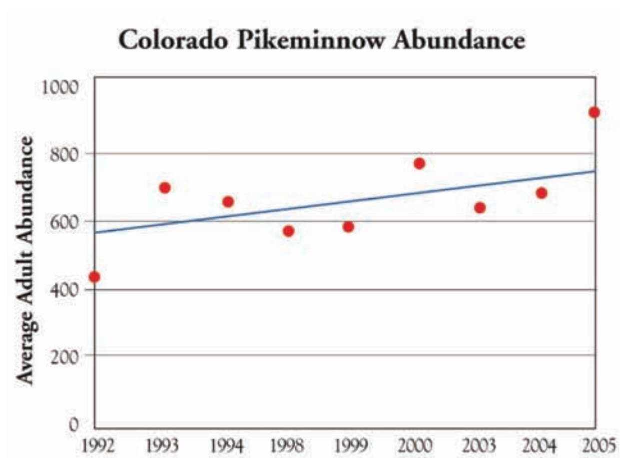


Figure 3. Estimated average abundance of adult Colorado pikeminnow in the upper Colorado River system from 1992 to 2005 (Upper Colorado River Endangered Fish Recovery Program and San Juan River Basin Recovery Implementation Program, 2009).

Humpback Chub

Five humpback chub wild populations inhabit canyon-bound river reaches of the Colorado, Green, and Yampa Rivers. The most current estimates of abundance of these populations indicate downward trends associated with increased abundance of nonnative fish during dry weather and low river runoff conditions since 1999. About 3,000 adults occur in Black Rocks and Westwater Canyons on the Colorado River (Upper Colorado River Endangered Fish Recovery Program and San Juan River Basin Recovery Implementation Program, 2009). Together, these populations have been identified as one core population. About 1,000 adults occur in Desolation/Gray Canyons on the Green River, and this population has been identified as a second core population. Populations in Yampa Canyon on the Yampa River and in Cataract Canyon on the Colorado River are small (as they were historically), each consisting of up to a few hundred adults.

Razorback Sucker

The razorback sucker was historically abundant in most warmwater rivers of the Colorado River Basin, but their numbers decreased dramatically beginning in the mid 1970s. Fewer than 100 wild adult razorback suckers are estimated to still occur in the Green River system, and wild populations are considered extirpated from the upper Colorado River system. Scientists recaptured 2,550 stocked razorback suckers from the Colorado, Gunnison, and Green Rivers from 2000 to 2005. Stocked razorback suckers are moving between the Colorado, Gunnison, and Green Rivers, suggesting that a network of populations (or metapopulation) similar to the Colorado pikeminnow situation may eventually be formed. Razorback suckers stocked in the Colorado and Green Rivers have been recaptured in reproductive condition, and captures of larvae in the Green, Gunnison, and Colorado Rivers demonstrate successful reproduction. Numbers of razorback sucker larvae collected from the Green River in 2007 were the highest ever recorded (fig. 4) (Upper Colorado River Endangered Fish Recovery Program and San Juan River Basin Recovery Implementation Program, 2009). Survival of larvae through the first year is evidenced by captures of juveniles in the Green and Gunnison Rivers.

Bonytail

The bonytail is the rarest of the four endangered Colorado River fish and probably the farthest from recovery. Before stocking began, the species had essentially disappeared in the upper basin and little was known about its biology. A key aspect to bonytail recovery is research and monitoring of stocked fish to determine the life history and habitat requirements of the species and ways to modify the stocking plan to improve the survival of stocked fish. Stocking efforts have been expanded to place fish into flood-plain wetlands to enhance their growth and survival. Stocked bonytails are being recaptured in several locations and habitats throughout the Green and upper Colorado Rivers. About 200 stocked bonytails were recaptured in 2004 and 2005, all within 1 year after stocking.

Water Use and Development

The UCRRP serves as a vehicle for compliance with Section 7 of the ESA for water development and management activities by participants, including the Federal government. Under the UCRRP's "Section 7 Agreement," accomplishments of the UCRRP serve as the reasonable and prudent alternative to jeopardy and adverse modification of critical habitat from water project depletion impacts. Each year, the Service evaluates whether progress in implementing recovery actions is sufficient for the UCRRP to continue to serve as the reasonable and prudent alternative. The UCRRP is responsible for providing flows that the Service determines are essential to recovery; therefore, responsibilities to offset water project depletion impacts do not fall on individual projects or their proponents. The UCRRP provides ESA compliance for more than 1,600 water projects depleting more than 2 million acre-feet of water per year. Most of these depletions were occurring before the UCRRP's inception in 1988, with only 12 percent of this amount from new depletions.

Collaboration

The UCRRP has been effective at implementing actions designed to recover endangered fish species while working in concert with interstate water compacts and State water and wildlife laws. UCRRP participants recognize that consensus-based collaboration is better than unproductive confrontation and that they can accomplish far more working together than would ever be possible working alone. The value of the collaborative approach undertaken by the UCRRP has been recognized by Congress through bipartisan support of

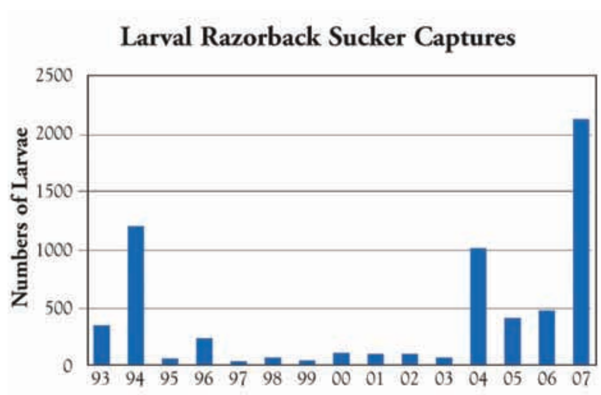


Figure 4. Captures of razorback sucker larvae for 1993 to 2007 in the middle Green River (Upper Colorado River Endangered Fish Recovery Program and San Juan River Basin Recovery Implementation Program, 2009).

appropriations and authorizing legislation: (1) Public Law 106–392 (Oct. 30, 2000, 114 Stat. 1602) specified the Federal and non-Federal cost-sharing arrangements, (2) Public Law 107–375 (Dec. 19, 2002, 116 Stat. 3113) extended the period to complete capital construction to 2008, and (3) Public Law 109–183 (Mar. 20, 2006, 120 Stat. 290) authorized an additional $15 million for capital construction and extended the construction period to 2010.

The UCRRP is considered by many to be a national model of how to recover endangered species in the face of development conflict. Whatever success has been realized is due not to the leadership of just one or two people, but to the synergy of effort and dedication of all its participants. Much like an ecosystem, each participant plays a vital role.

A partnership approach is the only viable means to achieve recovery because each stakeholder's cooperation is needed to accomplish the many and formidable actions required to recover the endangered fish. Although drought and expanding nonnative fish populations have resulted in some recent setbacks, UCRRP partners remain optimistic that they can continue to determine and implement the necessary management actions to ultimately achieve recovery.

For more information contact:

Upper Colorado River Endangered Fish
Recovery Program
U.S. Fish and Wildlife Service
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Lakewood, CO 80225
http://www.fws.gov/ColoradoRiverrecovery/

Upper Colorado River Endangered Fish Recovery Program

San Juan River Basin Recovery Implementation Program (SJRIP)

Program History

In the early 1980s, ESA compliance related to two major projects, the Animas-La Plata Project and Navajo Indian Irrigation Project, lead to the formation of the SJRIP. In the early 1990s, the Service determined that the current and cumulative adverse impacts associated with these water development projects were creating conditions that jeopardized the continued existence of Colorado pikeminnow and the razorback sucker within the San Juan River Basin. The impacts of these projects focused on water depletion but also included water-quality degradation, contamination from irrigation returns, scouring and sedimentation of the river channel, and changes to the water temperature of the river. The Service recognized that for water development to continue and for the endangered fish populations to be protected and recovered in the San Juan River Basin, a program or plan was needed for stakeholders to work cooperatively to meet both needs. To avoid jeopardy to the listed species from the Animas-La Plata Project, a reasonable and prudent alternative was agreed to in 1991; it included the development of a fish recovery program in the San Juan River Basin. A cooperative agreement established the SJRIP in 1992.

Program Goals

The specific goals of the SJRIP are to (1) conserve populations of Colorado pikeminnow and razorback sucker in the San Juan Basin consistent with the recovery goals established under the ESA and (2) proceed with water development in the San Juan Basin in compliance with Federal and State laws, interstate compacts, Supreme Court decrees, and Federal trust responsibilities to the Southern Ute Indian Tribe, Ute Mountain Ute Indian Tribe, the Jicarilla Apache Nation, and the Navajo Nation. It is anticipated that actions undertaken by the SJRIP to recover the listed species will also provide benefits to other native fish in the basin (table 2).

Geographic Scope

The geographic scope of the SJRIP is the San Juan River (fig. 5). From its origins in the San Juan Mountains of Colorado, the San Juan River flows approximately 31 miles to the New Mexico border, 190 miles westward through New Mexico to the Four Corners area, and another 136 miles through Utah to Lake Powell.

Table 2. Native fish of the San Juan River Basin (San Juan River Basin Recovery Implementation Program, 2006).

Species	Status
Bluehead sucker (*Catostomus discobolus*)	Abundant, generally distributed and typically numerous
Bonytail (*Gila elegans*)	Endangered, United States
Colorado pikeminnow (*Ptychocheilus lucius*)	Endangered, United States
Colorado River cutthroat trout (*Oncorhynchus clarki pleuriticus*)	Protected, Colorado
Flannelmouth sucker (*Catostomus latipinnis*)	Abundant, generally distributed and typically numerous
Mottled sculpin (*Cottus bairdii*)	Rare, not generally distributed and never numerous
Razorback sucker (*Xyrauchen texanus*)	Endangered, United States
Roundtail chub (*Gila robusta*)	Protected, New Mexico
Speckled dace (*Rhinichtys osculus*)	Common, generally distributed but typically not numerous

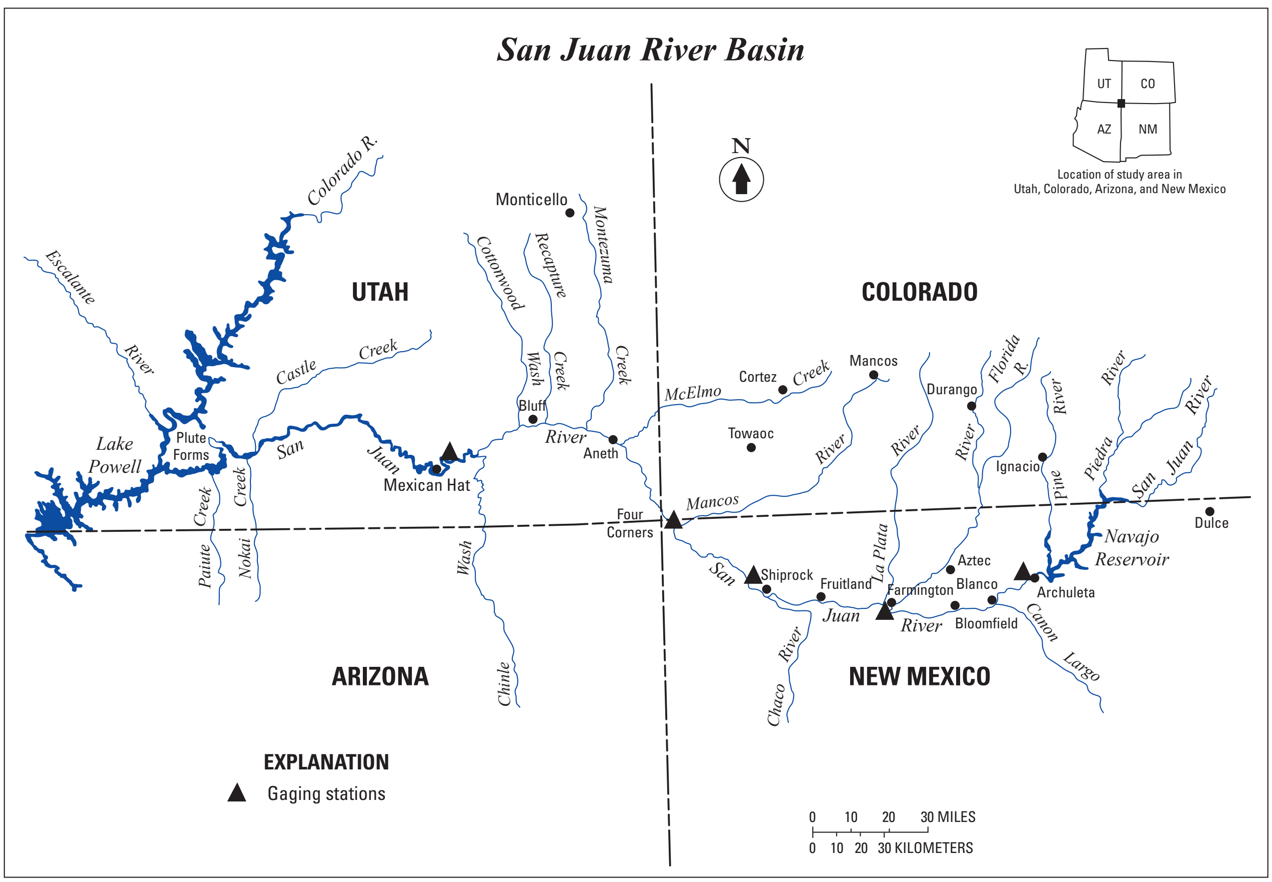

Figure 5. The San Juan River and its tributaries in Arizona, Colorado, New Mexico, and Utah.

Program Participants

- Jicarilla Apache Nation
- Navajo Nation
- Southern Ute Indian Tribe
- Ute Mountain Ute Indian Tribe
- State of Colorado
- State of New Mexico
- Bureau of Indian Affairs
- Bureau of Land Management
- Bureau of Reclamation
- U.S. Fish and Wildlife Service
- Water development interests in Colorado and New Mexico
- Conservation interests

Program Structure and Budget

The SJRIP developed a long-range plan to serve as the research, monitoring, and implementation document for recovery activities (San Juan River Basin Recovery Implementation Program, 2009). The long-range plan specifies the logical progression and priority for implementing recovery actions within the San Juan River Basin that are expected to result in recovery of the San Juan River populations of Colorado pikeminnow and razorback sucker based on the research and evaluation information provided from past studies. This plan along with other SJRIP documents provides the foundation for scheduling, budgeting, and implementing research, monitoring, and capital projects and other recovery activities.

Genetically diverse Colorado pikeminnow (*Ptychocheilus lucius*) and razorback sucker (*Xyrauchen texanus*) produced at facilities like the Dexter National Fish Hatchery and Technology Center, which is pictured here, are used to stock the San Juan River.

Three committees—Coordination, Biology, and Hydrology—were established to carry out the SJRIP. The purpose of the Coordination Committee is to assure that the goals of the SJRIP are achieved in a timely manner. It establishes SJRIP policies, direction, procedures, and organization; approves annual work plans and budgets; and performs conflict resolution. Each participant in the SJRIP has the right to one voting representative on the Coordination Committee. Coordination Committee members appoint representatives to the Biology and Hydrology Committees.

The SJRIP's annual budget for fiscal year 2009 was $2.4 million.

Program Activities

Recovery for the Colorado pikeminnow and razorback sucker is based on the reduction or removal of threats and the improvement of the status of each species during the time it is federally listed. The recovery goals for these two endangered fish include site-specific management actions and tasks and describe objective measurable downlisting and delisting criteria (U.S. Fish and Wildlife Service, 2002b, 2002d). The recovery plans list demographic criteria that describe numbers of populations and individuals (adults and juveniles) that are required before downlisting and delisting can be considered. Recovery elements include the following areas:

Protection, Management, and Augmentation of Habitat. This element identifies important river reaches and habitats for different life stages of the endangered fish and makes appropriate habitat improvements, including providing flows in the San Juan River and passage around migration barriers to provide suitable habitat to support recovered fish populations.

Water-Quality Protection and Enhancement. This element identifies and monitors water-quality conditions and takes actions to diminish or eliminate identified water-quality problems that limit recovery.

Interactions Between Native and Nonnative Fish Species. This element identifies problematic nonnative fish species and implements actions to reduce negative interactions between the endangered fish species and nonnative fish species.

Protection of Genetic Integrity and Management and Augmentation of Populations. This element ensures that the SJRIP's augmentation protocols maintain genetically diverse fish species while producing new generations of Colorado pikeminnow and razorback sucker to stock the river system.

Monitoring and Data Management. This element evaluates the status and trends of the endangered fish species, and of other native and nonnative species, and measures progress toward achieving recovery goals.

Progress Toward Program Goals

Flow Recommendations

The Animas-La Plata Project includes several measures intended to offset or minimize negative impacts on the fish community, which are based on 7 years of research to determine the endangered fish habitat needs and to operate Navajo Dam to mimic a natural hydrograph for the life of the dam. In 1991, experimental flow releases from Navajo Dam were initiated for the recovery of the two endangered fish species in the San Juan River. Since then, the reservoir has been operated to mimic a natural hydrograph with high spring peak releases and low base-flow releases.

Based on information from the experimental flow period and the 7-year study completed in 1999, the Biology Committee developed quantitative flow recommendations for the San Juan River below the Animas River confluence (Holden, 1999). The flow recommendations consist of (1) spring snowmelt period peak-flow rates, durations, and recurrence intervals to provide for creation and maintenance of spawning and rearing habitat on the basis of flow statistics for the San Juan River at Four Corners and (2) target base flows in the San Juan River to provide low-velocity habitats for rearing during the summer, fall, and winter months as measured by a combination of gages at Farmington, Shiprock, Four Corners, and Bluff. The flow recommendations were adopted by the Coordination Committee and are being implemented by specific operations decision criteria for Navajo Dam. These operating rules provide sufficient releases of water at times, quantities, and durations necessary to meet the flow recommendations while maintaining the authorized purposes of the Navajo Unit.

Removing Barriers and Preventing Entrainment

Five diversion structures were identified in the Program Evaluation Report between river mile (RM) 180 and RM 140 that were reported to be potential barriers to fish movement, particularly upstream movement (Holden, 2000). From upstream to downstream, the identified diversions were Fruitland Diversion (RM 178.5), Public Service Company of New Mexico Weir (PNM Weir; also known as the San Juan Generating Station; RM 166.6), Arizona Public Service Company Weir (APS Weir; also known as Four Corners Generating Station; RM 163.3), Hogback Diversion (RM 158.6), and Cudei Diversion (RM 142.0). Upon further investigation, the Fruitland Diversion did not appear to be an impediment to fish passage (Stamp and others, 2005). Cudei Diversion, Hogback Diversion, and APS Weir were deemed to be passable by fish at some flows, but upstream movement was restricted by PNM Weir, especially for nonnative fish (Ryden, 2000). The Biology Committee recommended that the SJRIP work with the Bureau of Reclamation to explore alternatives that could improve fish passage at the APS Weir (U.S. Fish and Wildlife Service, 2006). In 2002, the SJRIP combined Hogback and Cudei Diversions and constructed a nonselective fish passage at the Hogback Diversion to restore access to 36 miles of critical habitat. The SJRIP completed a selective fish passage around the PNM Weir in 2003 that allows native fish to continue upstream while removing nonnative fish from the San Juan River. Fish use of the PNM passage is monitored by the Navajo Nation Department of Fish and Wildlife in monthly reports. Currently, all identified impediments to fish movement have been removed with the exception of APS Weir and Fruitland Diversion. The SJRIP continues to track fish movement up and downstream from these diversions to evaluate the number and frequency of fish that negotiate these barriers and will pursue a potential passage at APS Weir and Fruitland Diversion if it is warranted.

In addition to blocking upstream movement of adult fish, diversions may also impact endangered fish recruitment by entraining eggs and larvae. In 2004 and 2005, numerous native and nonnative fish, including more than 200 Colorado pikeminnow, were detected in irrigation canals along the San Juan River but were most numerous in the Hogback Diversion Canal (Renfro and others, 2006). The SJRIP will begin construction of a fish weir at Hogback Diversion in 2010. Methods are being implemented to ensure that endangered fish do not become entrained in these structures by shifting the timing of stocking events to occur after the active irrigation season and evaluating the need to screen the intakes to these facilities to keep fish from entering the canals (Renfro and others, 2006). The SJRIP continues to evaluate the need for fish screens or deflection weirs at other diversion and out-take structures along the San Juan River.

Nonnative Fish Removal

The introduction of nonnative species has been a major factor contributing to the extinction of many North American freshwater fish because of predation, competition, and hybridization (Miller and others, 1989). The SJRIP began limited mechanical removal of nonnative fish in 1997, and intensive removal of nonnative fish by way of raft electrofishing has occurred in the upper and lower portions of the San Juan River since 2001 and 2002, respectively (Davis and others, 2009; Elverud, 2009). Beginning in 2006, management efforts were expanded to remove nonnative fish from a greater proportion of critical habitat by including the reach from Shiprock, NM, to Mexican Hat, UT. Nonnative control efforts have focused on removing channel catfish (*Ictalurus punctatus*) and common carp (*Cyprinus carpio*) from the San Juan River. Although river-wide capture rates of channel catfish have remained relatively constant following the initiation of intensive nonnative removal efforts, catfish do appear to be responding to removal efforts and have shifted their distribution to sections of the river that have not been included in this long-term removal effort (Ryden, 2009). Capture rates of common carp have declined through time over the entire river (Davis and others, 2009; Elverud, 2009). With continued river-wide removal efforts there is hope that numbers of these

T. Ross Reeve, Bureau of Reclamation

Nonnative fish removal efforts have reduced the abundance of adult channel catfish (*Ictalurus punctatus*) in high-priority upper and lower sections of the San Juan River where catfish numbers were highest.

nonnative predators and competitors will decline. Endangered fish population response cannot yet be linked to nonnative removal efforts, but it is expected that these efforts will promote the survival of native fish as the amount of predation and competition between native and nonnative fish is reduced. However, there does not appear to be a clear response of common native sucker species to nonnative fish removal efforts (Davis and others, 2009).

Stocking and Augmentation

Of all the management actions to recover Colorado pikeminnow and razorback sucker in the San Juan River, stocking/augmentation with hatchery-produced fish has probably led to the largest population response of the endangered fish because of its direct impact on increasing endangered fish numbers. The SJRIP developed formal augmentation plans for razorback sucker and Colorado pikeminnow in 1997 and 2002, respectively (Ryden, 1997, 2003). Colorado pikeminnow are reared at Dexter National Fish Hatchery and Technology Center (Dexter) to satisfy the SJRIP's annual stocking objectives of 300,000 young-of-year and 3,000 juvenile pikeminnow. Razorback sucker reared at Uvalde National Fish Hatchery (Uvalde) are stocked in the San Juan River, and razorbacks reared at Dexter are stocked in Navajo Agricultural Products Industry (NAPI) grow-out ponds in the spring and harvested in the fall to supplement the number of fish stocked from Uvalde. The program's stocking objective for razorback sucker is 11,400 fish from Uvalde, and the 10,500 razorbacks stocked at NAPI ponds are supplemental to the 11,400 stocking target. With an expected return rate of 40 to 60 percent at NAPI ponds, an additional 4,200 to 6,300 supplemental razorback suckers are anticipated to be stocked into the river. Because both species are long-lived it will take many years to determine if these stocking activities are successful.

Coordination with Other Recovery Efforts

Activities conducted under the SJRIP are closely coordinated with the UCRRP. The programs share outreach, education, and research efforts and co-fund hatchery production efforts for razorback sucker and bonytail at Uvalde National Fish Hatchery. Coordination among recovery efforts throughout the basin could effectively reduce overlap and duplication of recovery, outreach, and research activities and improve the overall effectiveness of each program.

For more information contact:

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Albuquerque, NM 87113–1001
http://www.fws.gov/southwest/sjrip

Glen Canyon Dam Adaptive Management Program (GCDAMP)

Program History

The GCDAMP was established in 1997 as an outcome of the 1996 Record of Decision on the Operation of Glen Canyon Dam Final Environmental Impact Statement. Like many other environmental programs, the GCDAMP was the outgrowth of a long history of conflict surrounding the effects of Glen Canyon Dam operations on downstream resources in Glen Canyon National Recreation Area and Grand Canyon National Park. Glen Canyon Dam lies about 16 miles above the boundary between the upper and lower Colorado River Basin, or the “Compact Point.” This point is the boundary for water deliveries from the upper to the lower basin. So, although many of the effects of Glen Canyon Dam occur in the lower basin, the GCDAMP is treated as an upper basin program because the dam is physically located there. In this case, geopolitical boundaries and ecological boundaries do not coincide.

Controversy over the effects of dam operations motivated the Commissioner of Reclamation to initiate a science program in 1982 to examine the effects of dam operations on downstream resources. In 1989, in response to the findings of the science program, Secretary of the Interior Manual Lujan, Jr., ordered an Environmental Impact Statement (EIS) on the operation of Glen Canyon Dam and, to further protect downstream resources, in 1991 adopted interim operating criteria that restricted dam operations.

While the EIS was underway, Congress passed the 1992 Grand Canyon Protection Act (GCPA), which required the Secretary of the Interior to “operate Glen Canyon Dam…and exercise other authorities under existing law in such a manner as to protect, mitigate adverse impacts to, and improve the values for which Grand Canyon National Park and Glen Canyon National Recreation Area were established, including, but not limited to natural and cultural resources and visitor use” (Sec. 1802 (a) of Public Law 102–575, Oct. 30, 1992). The act also required the Secretary to undertake this requirement “in a manner fully consistent with and subject to the [body of laws] that govern allocation, appropriation, development, and exportation of the waters of the Colorado River Basin” (Sec. 1802 (b) of Public Law 102–575, Oct. 30, 1992).

The Secretary of the Interior clearly was faced with a dilemma. Congress required operation of the dam to protect and improve park resources while fulfilling all water delivery and development purposes at a time when, admittedly, there was insufficient knowledge of how to operate the dam to achieve the required objectives. To proceed in the face of uncertainty, the Secretary decided to implement the preferred alternative outlined in the 1995 EIS, which included an adaptive management program having two major principles: (1) increased and recurrent stakeholder involvement through a Federal Advisory Committee and (2) a strong commitment to a scientific foundation for recommendations through a research and monitoring program.

Program Goals

According to the 1995 Final EIS, the “purpose of the AMP [Adaptive Management Program] would be to develop modifications to Glen Canyon Dam Operations and to exercise other authorities under existing laws as provided in the GCPA to protect, mitigate adverse impacts to, and improve the values for which the Glen Canyon National Recreation Area and Grand Canyon National Park were established” (U.S. Department of the Interior, 1995, p. 34).

Geographic Scope

The GCDAMP focuses on a study area that encompasses the Colorado River corridor from the forebay of Glen Canyon Dam to the western boundary of Grand Canyon National Park. The study area includes the approximately 15 river miles of the river from the dam to Lees Ferry within Glen Canyon National Recreation Area and the entire 277 river miles of the river below Lees Ferry and within Grand Canyon National Park. In total, the study area includes some 293 river miles of the Colorado River.

Program Participants

Tribes
Hopi Tribe
Hualapai Tribe
Navajo Nation
Pueblo of Zuni
San Juan Southern Paiute Tribe
Southern Paiute Consortium

State and Federal Cooperating Agencies
Arizona Game and Fish Department
Bureau of Indian Affairs
Bureau of Reclamation
National Park Service
U.S. Department of Energy, Western Area Power Administration
U.S. Fish and Wildlife Service

Colorado River Basin States
Arizona: Arizona Department of Water Resources
California: Colorado River Board of California
Colorado: Colorado Water Conservation Board
Nevada: Colorado Water Commission of Nevada
New Mexico: New Mexico Office of the State Engineer
Utah: Water Resources Agency
Wyoming: State Engineer’s Office

Nongovernmental Groups
Grand Canyon Trust
Grand Canyon Wildlands Council
Federation of Fly Fishers/Northern Arizona Flycasters
Grand Canyon River Guides
Colorado River Energy Distributors Association
Utah Associated Municipal Power Systems

Program Structure and Budget

The GCDAMP is facilitated by the Adaptive Management Work Group (AMWG), which is organized as a Federal advisory committee. The Secretary of the Interior appoints the group's 25 members, who include representatives from the entities identified above. The AMWG makes recommendations to the Secretary on dam operations and other actions under the Secretary's authority. Many AMWG recommendations have been for management experiments to better understand the effects of dam operations on natural resources. The GCDAMP is administered by a senior Department of the Interior official who also serves as the chair of AMWG.

The GCDAMP also includes the U.S. Geological Survey's (USGS) Grand Canyon Monitoring and Research Center, the Technical Work Group (TWG), and independent scientific review panels. The TWG is composed of managers from the same 25-member group as the AMWG. Additional scientific expertise is provided by a standing group of science advisors and ad hoc external scientists who review proposals and provide reviews of research and monitoring protocols. Recently, the Secretary of the Interior added a Policy Group, composed of senior officials that oversee Departmental agencies, to ensure intradepartmental communication and coordination at the national level (fig. 6; Norton, 2006).

As the program's name implies, adaptive management guides the efforts of the GCDAMP. Murray and Marmorek (2004, p. 1) succinctly define adaptive management as "…a rigorous approach to environmental management designed to explicitly address and reduce uncertainty regarding the most effective on-the-ground actions for achieving management goals and objectives." The important point is that adaptive management is an iterative learning process that recognizes uncertainty and invokes science in decisionmaking. Policies are treated as experiments, and thus, they must be tested.

The GCDAMP's annual budget for fiscal year 2009 was $13.6 million.

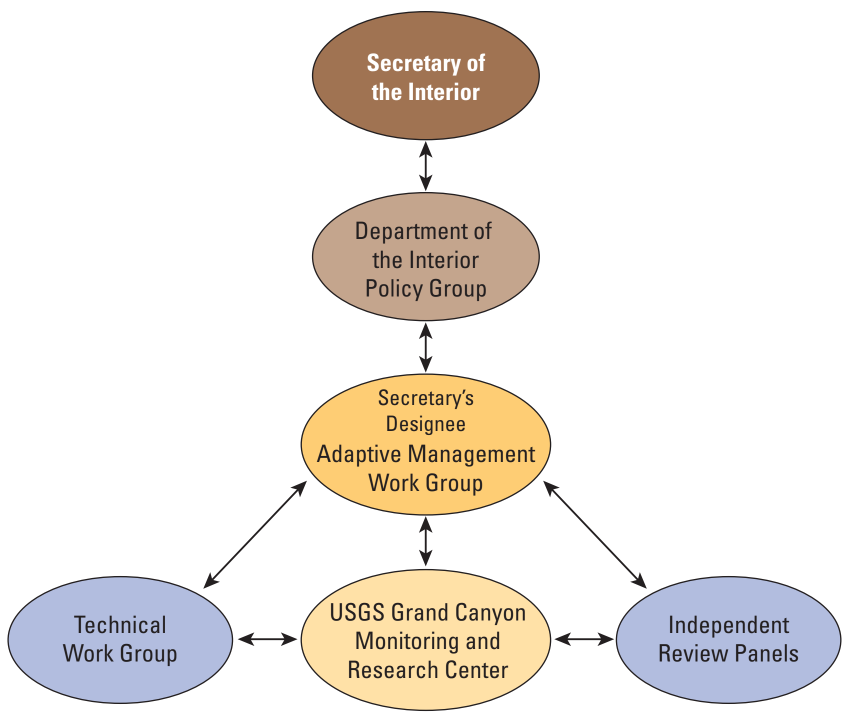

Figure 6. Structure of the Glen Canyon Dam Adaptive Management Program. The Secretary of the Interior appoints the Adaptive Management Work Group's 25 members, who include representatives from Federal and State resource management agencies, the seven Colorado River Basin States, Native American Tribes, environmental groups, recreation interests, and contractors of Federal power from Glen Canyon Dam.

Program Activities

The program undertakes three types of activities: (1) long-term monitoring, (2) research and development, and (3) flow and nonflow experimentation related to the efficacy of a range of management actions. Monitoring involves consistent, long-term repeated measurements using accepted protocols to assess status and trends of key resources, including native and nonnative fish, sediment resources such as sandbars, water quality, aquatic food production, riparian vegetation, recreation, and cultural sites. Research and development activities test specific hypotheses related to key resources and develop and test new technologies and monitoring procedures. Experimentation is used to determine how water releases from Glen Canyon Dam and other potential nonflow management actions might be used to meet resource goals. Because it is the cornerstone of adaptive management, experimentation is discussed in greater detail below.

Experimentation

The GCDAMP is best known for a series of three high-flow experiments, or water releases designed to mimic natural seasonal flooding, conducted in 1996, 2004, and 2008 (table 3). High-flow experimental releases from the dam are designed to maintain Colorado River sandbars, or beaches, by flushing tributary-derived sand from the riverbed up and onto sandbars. The high-flow experiments had multiple objectives, but two were paramount. The first purpose was to rebuild beaches used by campers and river runners, and the second was to rejuvenate and re-create attendant native fish habitats, the backwaters that formed in the lee spaces between the sandbars and the river banks. By building beaches and backwaters along the shores of the main channel, managers and scientists also sought to provide habitats that would be used by young native fish, especially in their first year of life.

Another major experiment occurred in 2000, when low summer steady flows, bordered by powerplant capacity habitat maintenance flows, were released from Glen Canyon Dam. This complex flow experiment was considered a test of concept for the seasonally adjusted steady flow reasonable and prudent alternative issued by the Service in its 1995 biological opinion.

In 2002, an environmental assessment written by Reclamation, the National Park Service, and the USGS increased the range of GCDAMP experimental actions by including

Table 3. Chronology of experiments conducted under the Glen Canyon Dam Adaptive Management Program.

Year	Dam operations	Nonflow actions
1996	Modified low fluctuating flows/beach/habitat-building flow	None
1997	Modified low fluctuating flows/habitat maintenance flow	None
1998	Modified low fluctuating flows	None
1999	Modified low fluctuating flows	None
2000	Modified low fluctuating flows/low summer steady flows/ habitat maintenance flows	None
2001	Modified low fluctuating flows	None
2002	Modified low fluctuating flows	None
2003	Modified low fluctuating flows /nonnative fish suppression flows	Nonnative fish mechanical removal/tributary translocation of endangered humpback chub
2004	Modified low fluctuating flows/beach/habitat-building flow/ nonnative fish suppression flows	Nonnative fish mechanical removal/tributary translocation of endangered humpback chub/habitat conservation for endangered Kanab ambersnail
2005	Modified low fluctuating flows/nonnative fish suppression flows	Nonnative fish mechanical removal/tributary translocation of endangered humpback chub
2006	Modified low fluctuating flows	Nonnative fish mechanical removal
2007	Modified low fluctuating flows	None
2008	Modified low fluctuating flows /beach/habitat-building flow/ Sept.-Oct. steady flows	Tributary translocation of endangered humpback chub/habitat conservation for endangered Kanab ambersnail/nearshore ecology research
2009	Modified low fluctuating flows/Sept.-Oct. steady flows	Nonnative fish removal/tributary translocation of endangered humpback chub/hatchery refuge/nearshore ecology research

T. Ross Reeve, Bureau of Reclamation

Glen Canyon Dam releases high flows of Colorado River water on the night of March 6, 2008. A high-flow experiment was undertaken to determine if water releases designed to mimic natural seasonal flooding could be used to improve a wide range of resources in Glen Canyon National Recreation Area and Grand Canyon National Park.

mechanical removal of nonnative fish in the Colorado River and translocation of endangered humpback chub to an unoccupied reach of the Little Colorado River. The transition to an experiment containing both flow and nonflow actions was important because not all threats to Colorado River resources could be addressed adequately through dam operations. The 2002 environmental assessment also contained triggers that dictated minimum tributary fine sediment inputs necessary to initiate the second experimental high flow. The environmental assessment reduced the period of the high release from 1 week to 60 hours and included increased winter daily dam release fluctuations ranging from 5,000 to 20,000 cubic feet per second as "nonnative fish suppression flows." Because of drought and the associated low tributary sediment inputs, this high release did not occur until November 2004.

In 2008, Reclamation proposed a 5-year (2008 to 2012) experimental plan containing a high-flow experiment, steady flows during each September and October, and a diverse set of conservation measures that included nonnative fish removal in the Colorado River and its tributaries, translocation of endangered humpback chub, establishment of a hatchery refuge for the endangered fish, continued development of a comprehensive management plan and watershed plan for the endangered chub, evaluation of endangered razorback sucker habitat for potential augmentation, and monitoring of other endangered species. This combination of efforts indicates a further recognition of the likely suite of actions that may be necessary to fully evaluate dam operations and other actions under the authority of the Secretary of the Interior. Other actions considered in the interim have included the construction and operation of a temperature control device to deliver warmer water through the dam and sediment augmentation through a slurry pipeline from Navajo Canyon in Lake Powell (Randle and others, 2007) to one or more locations below Glen Canyon Dam.

Progress Toward Program Goals

In its recently published guidebook on adaptive management, the Department of the Interior identified four measures of success in carrying out adaptive management: (1) stakeholders are actively involved and committed to the process, (2) progress is made toward achieving management objectives, (3) results from monitoring and assessment are used to adjust and improve management decisions, and (4) implementation is consistent with applicable laws (Williams and others, 2007). These metrics should be common to most adaptive management programs and should therefore have widespread utility in such assessments, including that of the GCDAMP.

Stakeholder Involvement and Support

The various GCDAMP members have very different ideas about what decisions the Secretary of the Interior should make to achieve an acceptable balance in dam operations priorities. To understand how different their values and positions are, it is only necessary to realize that the dam provides water and energy to supply the needs of millions of people, but it also sits within a national recreation area and above a national park containing one of the seven natural wonders of the world, Grand Canyon. Yet early acrimony among the members has given way to orderly development of annual budgets and work plans, complete with major experiments that use the dam as a learning tool, all delivered as recommendations to the Secretary. It appears that even people with very different value systems can work cooperatively when the goal is to increase the understanding of how a contested system works.

The primary purpose of the AMWG is to advise the Secretary of the Interior on actions that will assist in achieving the balance of interests identified in the GCPA, but not to manage or make operational decisions for the Secretary. This proximity to the ultimate decisionmaker (that is, the Secretary of the Interior) is one aspect of the GCDAMP that does not occur in many adaptive management programs. It provides a high level of relevancy to the recommendations made by the committee and a clear opportunity for them to understand the extent to which their advice is heeded in decisionmaking.

Monitoring Results Used to Adjust and Improve Management Decisions

A major challenge for the Secretary of the Interior is to balance the Colorado River Storage Project purposes for Glen Canyon Dam with subsequent responsibilities for resource stewardship provided through environmental laws and the GCPA. Any serious attempt to achieve this balance depends on a program of monitoring to determine the responses of system variables to actions taken by the adaptive management program. From its earliest days, the GCDAMP has been engaged in developing and implementing research and monitoring to assess the effects of dam operation on Colorado River resources. Because of the emphasis on active adaptive management, the GCDAMP does not just monitor resources, it also purposefully perturbs the Colorado River ecosystem through experiments and measures the resource responses. Three resources—fine sediments, endangered fish (humpback chub), and hydropower—with perceived divergent objectives exemplify the issues over how the dam is operated.

Fine Sediments

Nearly all the fine sediments that were carried through Grand Canyon before the emplacement of Glen Canyon Dam are now deposited on the bottom of Lake Powell and are unavailable to build beaches in Grand Canyon. Two tributaries below the dam—Paria River and Little Colorado River—now provide much of the fine sediments to the Grand Canyon reach of the Colorado River. Scientists measure the inputs of fine sediment from these tributaries; the concentration and size distribution of the particles as they are carried downstream, deposited, and re-suspended by the Colorado River; and the amount of sediment leaving Grand Canyon to develop a sediment budget. As with the money entering and leaving a bank account, this approach provides an index of whether one is overspending the account. These data combined with topographic surveys of the beaches and bathymetric surveys of the river bottom provide a portrayal of not only whether the remaining fine sediment below the dam is being conserved, but also where it is residing in the river corridor over time.

From dam experiments and attendant monitoring, scientists have determined that the sediment conservation paradigm used to develop EIS alternatives overestimated the residence time of new fine sediment added to the river bottom by downstream tributaries under the preferred alternative operations (Rubin and others, 2002; Melis and others, 2007). This discovery has led to development of minimum tributary sediment input criteria that must be met before a high-flow experiment can be implemented (U.S. Department of the Interior, 2002). Because the river never rests and ensuing clearwater flows released from the dam gradually reclaim the sediment thrown temporarily above its normal flow lines, the principal question for sediment researchers is whether there is a sustainable flow-only dam operation alternative that will rebuild and maintain sandbar habitats over decades. This question is being addressed through a combination of monitoring the effects of research flows and using models to determine if there is enough sand (Wright and others, 2008).

Endangered Humpback Chub

The population of endangered humpback chub in Grand Canyon is estimated through mark and recapture data that are incorporated into an age-structured stock assessment model similar to those used successfully for exploited marine fish (Coggins and others, 2006). All humpback chub of a sufficient size are marked with passive integrated transponders that respond to electronic signals by registering an identifying number. Movement information and change in size and condition are recorded when these same fish are recaptured. Because many individuals of this species reside for parts of the life cycle in the Little Colorado River and Colorado River where conditions for growth, reproduction, and survival differ markedly, it is a major accomplishment to gain such insight into the ecology of this fish.

The first continuous series of annual population estimates for the endangered humpback chub population in Grand Canyon (fig. 7) has been accomplished during the GCDAMP (Coggins, 2008; Coggins and Walters, 2009). A credible series

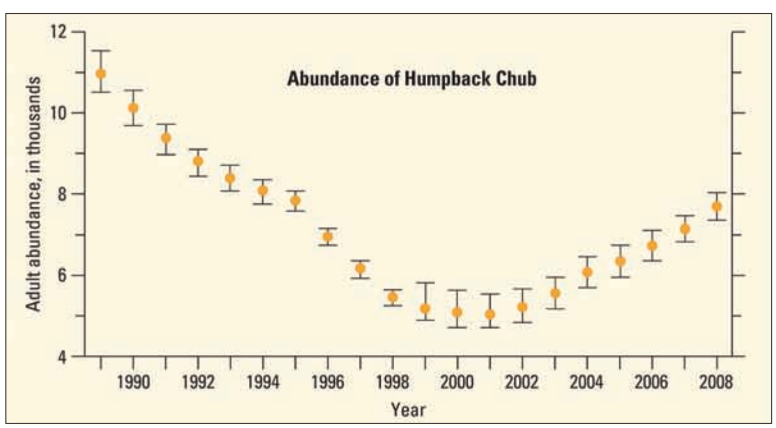

Figure 7. Estimated adult humpback chub abundance in Grand Canyon using age-structured mark recapture model and incorporating uncertainty in assignment of age (Coggins and Walters, 2009).

of estimates was not available in 1995, when the Service reached its determination of jeopardy. Model estimates showed that approximately 7,400 adults were present in the Grand Canyon population in 1995. Adult numbers subsequently fell to a low of about 5,000 in 2001, but by 2008 had rebounded to an estimated 6,000 to 10,000 adults (fig. 7) (Coggins and Walters, 2009). This turnaround, in conjunction with conservation measures for the endangered fish being undertaken by Reclamation through the GCDAMP, has convinced the Service to rescind its earlier jeopardy opinion in favor of a non-jeopardy opinion (U.S. Fish and Wildlife Service, 2008).

Hydropower

Hydropower monitoring data for the GCDAMP were largely collected and held by Reclamation and Western Area Power Administration until 2007. These data are available from the System Control and Data Acquisition system on an hourly time step and are reported daily, weekly, and monthly. The Western Area Power Administration is preparing to provide the hydropower data to the USGS Grand Canyon Monitoring and Research Center to serve through its Web site. Much of the interest in these data has been for their use in retrospective analyses of costs associated with experiments that released water and bypassed the powerplant or reduced the ability to match hydropower demand with hydropower production.

Another use of hydropower production data is to determine whether projections of the 1995 EIS preferred alternative have been borne out. The change in hydropower production under the preferred alternative in the 1995 EIS was projected to be a decrease of 442 megawatts (MW) of capacity in winter and 463 MW in summer (U.S. Department of the Interior, 1995). Economic cost increases of $15.2 to $44.2 million per year were estimated, and the financial costs to utilities were estimated at $89.1 million per year. Attribution of impacts to hydropower, including supplemental purchases, from the GCDAMP experiments is difficult and has not yet been done in a comprehensive manner, although the cost of replacement power for the recent 2008 high-flow test was estimated to be $4.1 million. It is clear from hydropower generation data that there has been a decrease in peaking generation capacity and associated revenue at Glen Canyon Dam since the 1996 Record of Decision and that costs for replacement power must be added. There are, however, a number of confounding factors, not the least of which is the loss of head from declining reservoir elevations during the recent protracted drought, which challenges this analysis (fig. 8). Efforts now underway (Tom Veselka, Argonne National Laboratory, oral commun., October 20, 2008) will soon close this gap and determine the cost to hydropower from resource protection in the adaptive management framework.

Progress Toward Achieving Resource Objectives

The 1995 EIS assessed effects of dam operations on 11 resource categories. In its 2001 strategic plan, the GCDAMP identified 11 resource goals, which are largely directed at these same resources (table 4). Nested under the 11 goals are 56 management objectives for resources or program functions. One shortcoming of most resource objectives is that although they contain metrics to be measured, they do not prescribe well-defined desired future conditions. In 2007, as part of development of a long-term experimental plan,

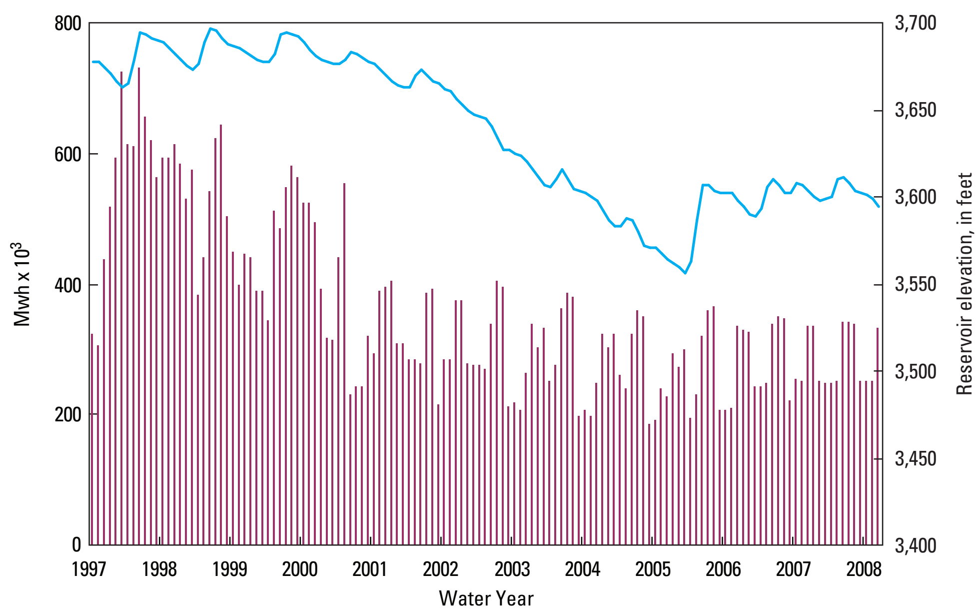

Figure 8. Monthly net generation of hydroelectric energy from Glen Canyon Dam (bars) and Lake Powell reservoir elevation (line) during the period from 1997 to 2008.

Table 4. The resource goals identified by the Glen Canyon Dam Adaptive Management Program (GCDAMP) currently being pursued and a summary of 2009 resource conditions (Glen Canyon Dam Adaptive Management Program, 2001; Hammill, 2009).

[EIS, Environmental Impact Statement]

Resource and GCDAMP goal	1995 EIS prediction	2009 summary
	Natural resources	
Water quality (Goal: Establish water temperature, quality, and flow dynamics to achieve GCDAMP ecosystem goals)		
Water temperature	No effect	Since 2003, downstream water temperatures have increased in response to drought conditions.
Specific conductance (salinity)	No effect	Drought conditions, prevalent since 1999, generally result in increases in specific conductance.
Sediment (sandbars and related physical habitats) (Goal: Maintain or attain levels of sediment storage within the main channel and along shorelines)	Modest improvement	Sandbars erode during periods between high flows. Increases in total sandbar area and volume are only possible when high-flow releases follow large tributary floods that enrich sand supplies in the main channel.
Aquatic food web (Goal: Protect or improve the aquatic food base)	Potential major increase	Increases were apparent in Glen Canyon Dam tailwater reach, but the trend is unclear along downstream reaches. Unlikely that quagga mussels (*Dreissena bugensis*) will become well established in the mainstem Colorado River below Lees Ferry or its tributaries.
Native fish (humpback chub) (Goal: Maintain or attain viable populations of existing native fish)	Potential minor increase	The population of adult humpback chub (*Gila cypha*) decreased between 1989 and 2001; however, adult abundance has increased more than 50 percent since 2001.
Trout (Goal: Maintain a naturally reproducing population of rainbow trout above the Paria River)	Increased growth potential, dependent on stocking	Rainbow trout (*Oncorhynchus mykiss*) numbers have decreased in the Lees Ferry reach.
Riparian vegetation (Goal: Protect or improve the biotic riparian and spring communities)	Modest increase	Native and nonnative woody vegetation continues to expand in the river corridor. Nonnative tamarisk (*Tamarix ramosissima*) is the dominant species, making up 24 percent of vegetation.
Kanab ambersnail (Goal: Maintain or attain viable populations of Kanab ambersnail)	Some incidental take	Snail habitat increased since 1998.
	Cultural resources	
Archeological sites affected (Goal: Preserve, protect, manage, and treat cultural resources)	Moderate degradation (less than 157 sites affected)	Archeological site condition continues to decline because of a combination of factors including erosion, gravity, visitor impacts, and insufficient sediment.
Traditional cultural resources affected (Goal: Preserve, protect, manage, and treat cultural resources)	Increased protection	Tribes have developed protocols for monitoring the condition of cultural resources in accordance with Tribal values.
	Recreation resources	
Whitewater boating camping beaches (average area at normal peak stage) (Goal: Maintain or improve the quality of recreational experiences)	Minor increase	Areas suitable for camping have decreased on average 15 percent per year between 1998 and 2003.

GCDAMP members began to develop a list of desired future conditions. Initial objectives for two resources, humpback chub and fine sediment, put forward by two members with differing views, Western Area Power Administration and the National Park Service, were developed through the Technical Work Group and Grand Canyon Monitoring and Research Center and were submitted to the AMWG with a request for direction to proceed with additional resources. Completion of this endeavor would provide an important feedback loop for monitoring to determine the effectiveness of the GCDAMP in meeting its resource objectives and provide a better foundation for the Secretary to balance project purposes with resource protection.

One of the criticisms of adaptive management, particularly of large programs like the GCDAMP, is that they are expensive. Since the inception of the program in 1997, approximately $92 million have been expended on this effort, with the primary source of funding coming from revenue derived from the generation of hydropower. Views among GCDAMP members, and indeed the public, vary greatly on whether this expenditure will result in desired future resource conditions and an equitable balance among the differing interests. None can dispute, however, that uncertainty is being replaced with knowledge and that adaptive management is providing a more objective basis for consideration of policy change.

Implementation Consistent with Applicable Laws

Until February 2006, GCDAMP members could contend that adaptive management serves as an insulator against legal action. Major experiments were carried out with little resistance, and no lawsuits were threatened or carried out against the program. In that month, however, five environmental groups sued the Secretary of the Interior and Reclamation claiming violations of the GCPA, ESA, and National Environmental Policy Act (NEPA). An out-of-court settlement of the lawsuit, which provided for initiation of NEPA and ESA compliance activities by agreed upon dates, was reached in August 2006.

With the legal waters settled, the GCDAMP moved forward with assistance from the Grand Canyon Monitoring and Research Center in 2007 toward developing a long-term experimental plan intended to cover approximately 10 years of scientific studies beginning in 2008 (U.S. Geological Survey, 2008). Reclamation and 16 cooperating agencies prepared alternative experimental designs from which a preferred alternative in an Environmental Impact Statement would be selected. In September 2007, however, one of the environmental groups in the GCDAMP delivered a notice of intent to sue Reclamation for violations of the ESA and NEPA. A supplemental complaint later added the Service as a defendant. The threatened legal action was taken in December 2007 and is ongoing.

An important conclusion for the process of adaptive management provided by the Glen Canyon Dam example is not whether lawsuits will occur, but whether the process reduces this likelihood. What is most important to learn in the present example is that even in the face of litigation, the GCDAMP persists and is continuing to function. In March 2008, even as litigation was underway, the hollow jet tubes again were opened on Glen Canyon Dam and a third experimental high-flow test took place. Scientists busily gathered more data to be analyzed, synthesized, and integrated into reports and publications. Scientists will present their reports to technical level managers who will convey their impressions of what has been learned to their Federal Advisory Committee counterparts. The AMWG will once again meet and make its recommendations, considering scientific, legal, and policy perspectives, to the Secretary of the Interior. And no doubt the Secretary will, with the advice of his Policy Group, use those recommendations to balance the priorities for which the dam was built with those that have come about through ensuing laws. Achieving that balance will be accomplished with much greater participation and with a much firmer scientific foundation than would have been possible in the days before the GCDAMP—not perfect, perhaps, but a definite move in the potentially fruitful direction of integrating science into policymaking.

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Lower Colorado River Multi-Species Conservation Program (LCR MSCP)

Program History

The LCR MSCP is a partnership of Federal and non-Federal stakeholders, created to respond to the need to balance the use of lower Colorado River water resources and the conservation of native species and their habitats in compliance with the ESA. This program is a long-term (50-year) plan to conserve at least 26 species along the lower Colorado River from Lake Mead to the southerly international boundary with Mexico through implementation of a Habitat Conservation Plan (HCP).

Twenty-six Federal or State-listed candidate and sensitive species and their associated habitats, ranging from aquatic and wetland habitats to riparian and upland areas, are covered in the LCR MSCP. Of the 26 covered species, 6 are currently listed under the Federal ESA. The program addresses the biological needs of mammals, birds, fish, amphibians, and reptiles, as well as invertebrates and plants.

Developed between 1996 and early 2005, implementation of the LCR MSCP began in April 2005 with the signing of a Record of Decision by the Secretary of the Department of the Interior. In December 2004, a Final Environmental Impact Statement for this effort was developed (Lower Colorado River Multi-Species Conservation Program, 2004a), which included a Habitat Conservation Plan (Lower Colorado River Multi-Species Conservation Program, 2004b) and a Biological Assessment (Lower Colorado River Multi-Species Conservation Program, 2004c). The implementation activities are based on adaptive management principles, which allow program conservation measures to be adjusted over time on the basis of monitoring and research. Reclamation, in consultation and partnership with a Steering Committee made up of representatives from the 56 participating entities, is the primary implementing agency for this activity.

Program Goals and Structure

The overall goal of the LCR MSCP is to develop and implement a plan that will

- conserve habitat and work toward the recovery of threatened and endangered species, as well as reduce the likelihood of additional species being listed;
- accommodate present water diversions and power production and optimize opportunities for future water and power development to the extent consistent with the law; and
- provide the basis for incidental take authorization.

Reclamation is the lead implementing agency for the LCR MSCP. Partner involvement occurs primarily through the LCR MSCP Steering Committee, currently representing 56 entities, including water and power users, Federal land-management agencies, State wildlife agencies, and other interested parties.

The LCR MSCP provides ESA compliance for covered actions undertaken by Federal agencies under Section 7 and by non-Federal partners under Section 10 of the act. Non-Federal partners have received incidental take authorization under Section 10(a) (1) (B). The program also allows California agencies to meet their obligations under California State law for the California Endangered Species Act (CESA).

Geographic Scope

The LCR MSCP area extends over 400 miles of the lower Colorado River from Lake Mead to the international boundary with Mexico, and includes Lakes Mead, Mohave, and Havasu, as well as the historic 100-year flood plain along the mainstem of the lower Colorado River (fig. 9).

Program Participants

Steering Committee Members:
- Federal Participants:
  - Bureau of Reclamation
  - U.S. Fish and Wildlife Service
  - National Park Service
  - Bureau of Land Management
  - Bureau of Indian Affairs
  - Western Area Power Administration
- Arizona Participants:
  - Arizona Department of Water Resources
  - Arizona Electric Power Cooperative, Inc.
  - Arizona Game and Fish Department
  - Arizona Power Authority
  - Central Arizona Water Conservation District
  - Cibola Valley Irrigation and Drainage District
  - City of Bullhead City
  - City of Lake Havasu City
  - City of Mesa
  - City of Somerton
  - City of Yuma
  - Electrical District No. 3, Pinal County, Arizona
  - Golden Shores Water Conservation District
  - Mohave County Water Authority
  - Mohave Valley Irrigation and Drainage District
  - Mohave Water Conservation District
  - North Gila Valley Irrigation and Drainage District
  - Town of Fredonia
  - Town of Thatcher
  - Town of Wickenburg
  - Salt River Project Agricultural Improvement and Power District

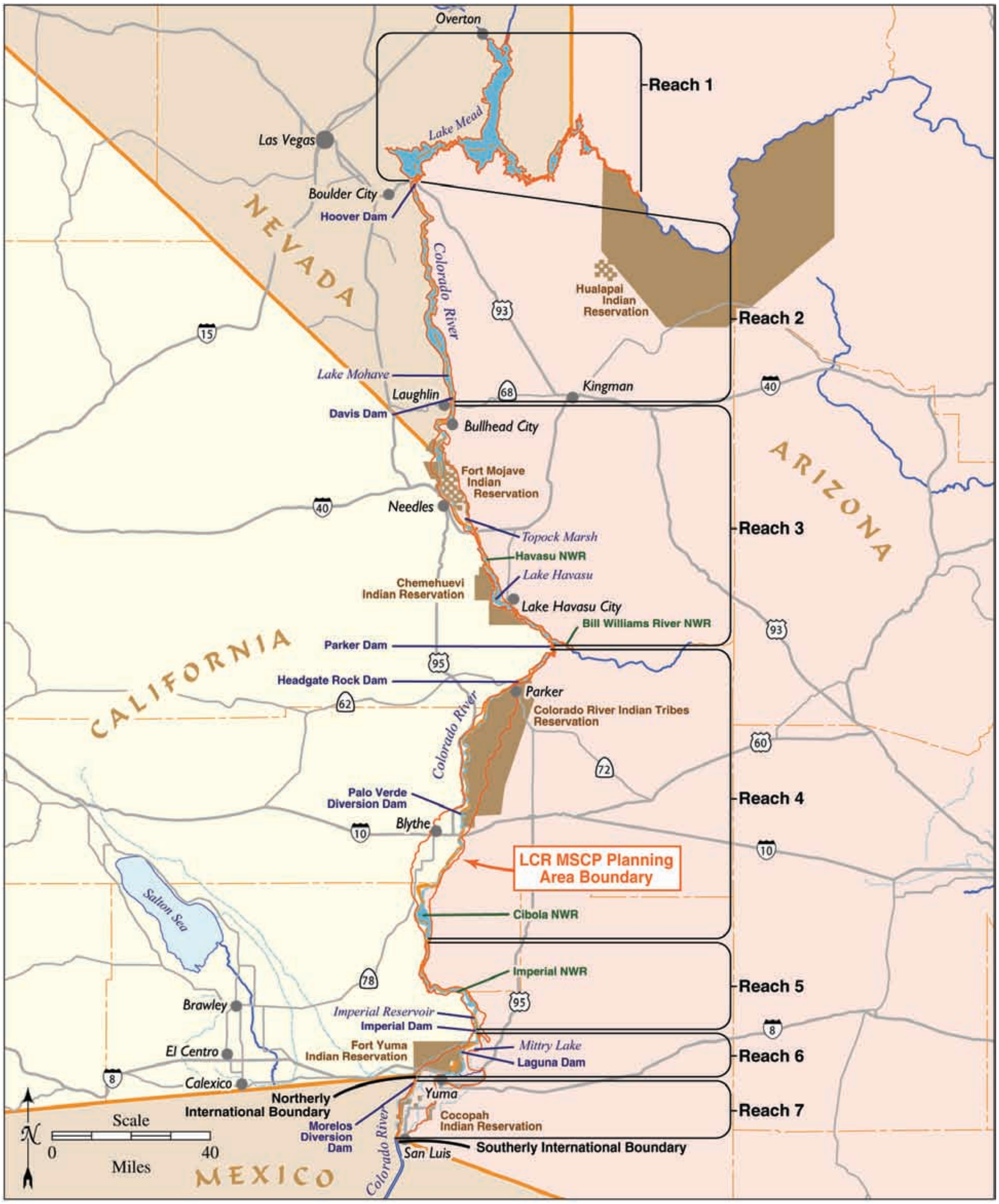

Figure 9. Lower Colorado River Multi-Species Conservation Program planning area and river reaches.

Arizona Participants (continued):
- Unit "B" Irrigation and Drainage District
- Wellton-Mohawk Irrigation and Drainage District
- Yuma County Water Users' Association
- Yuma Irrigation District
- Yuma Mesa Irrigation and Drainage District

California Participants:
- California Department of Fish and Game
- City of Needles
- Coachella Valley Water District
- Colorado River Board of California
- Bard Water District
- Imperial Irrigation District
- Los Angeles Department of Water and Power
- Palo Verde Irrigation District
- San Diego County Water Authority
- Southern California Edison Company
- Southern California Public Power Authority
- The Metropolitan Water District of Southern California

Nevada Participants:
- Colorado River Commission of Nevada
- Nevada Department of Wildlife
- Southern Nevada Water Authority
- Colorado River Commission Power Users
- Basic Water Company

Native American Participants:
- Hualapai Tribe
- Colorado River Indian Tribes
- The Cocopah Indian Tribe

Conservation Participants:
- Ducks Unlimited
- Lower Colorado River RC&D Area, Inc.

Other interested parties:
- QuadState County Government Coalition
- Desert Wildlife Unlimited

Program Structure and Budget

The HCP outlines general and species-specific measures to conserve species and their habitats. Chief components of the plan include:

- native fish augmentation
- species research
- species and ecosystem monitoring
- conservation area development
- existing habitat protection
- adaptive management

Twenty-six species are covered under the LCR MSCP through conservation measure implementation, including 4 native fish, 12 birds, 4 mammals, 2 reptiles, 1 amphibian, 1 insect, and 2 plants. In addition, conservation measures have been established for five evaluation species, including three mammals and two amphibians. These evaluation species were not covered by the program because life-history information available during plan development was not sufficient to determine whether covered actions would affect them or to develop effective specific conservation measures.

Total LCR MSCP costs are estimated at $626 million over 50 years, in 2003 dollars indexed annually to inflation. The LCR MSCP's annual budget for fiscal year 2009 was $15.8 million. The Department of the Interior will provide 50 percent of the program's estimated cost, and California, Nevada, and Arizona will jointly provide the other 50 percent.

Program Activities

Program activities have two main thrusts. The native fish augmentation program is designed to increase populations of several native fish species in the Colorado River, including the razorback sucker and bonytail. Habitat is also being created for species such as the southwestern willow flycatcher (*Empidonax trailii extimus*), the yellow-billed cuckoo (*Cuccyzus americanus occidentalis*), and the Yuma clapper rail (*Rallus longirostris yumanensis*). Conservation measures require 660,000 razorback suckers and 620,000 bonytail be released in the mainstem Colorado River downstream of Davis Dam over the life of the program. The LCR MSCP will create at least 8,100 acres of new riparian, marsh, and backwater habitats.

Progress Toward Program Goals

Since 2005, approximately 107,000 razorback suckers and bonytail have been stocked into Lake Mohave and the Colorado River below Davis Dam. Research and monitoring activities are ongoing in an effort to determine the success of this program.

During the first 3 years of LCR MSCP implementation, approximately 3,300 acres and 15,000 acre-feet have been secured for potential habitat creation. Several large habitat creation projects have been initiated since 2006, including two sites near Blythe, CA. Approximately 600 acres have been established during the first 3 years of program implementation, including 450 acres of cottonwood-willow. Approximately 92 acres of marsh and backwater habitats have been constructed at Imperial National Wildlife Refuge, near Yuma, AZ, to provide habitat for fish and marsh bird species.

Future Challenges

Since the LCR MSCP is a 50-year program, with reconsultation with the Service likely at the end of the current program, adaptive management will be an important component to ensure appropriate adaptation to changes in water and power demands, water priorities, water availability, and other unexpected changes in conditions.

Conservation area development requires the mutual commitment of LCR MSCP and the landowner or land manager prior to the initiation of any habitat creation project. This commitment ensures the availability of land and water at each site through the life of the program. Since native riparian habitat being created at many sites will require active management throughout the 50 years, this commitment is essential.

Research and monitoring will continue to be important components of the LCR MSCP over the life of the program so that potential issues are identified in time to plan and implement effective management actions.

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Changing the Law-Science Paradigm for Colorado River Restoration

By Robert W. Adler[1]

Abstract

Legal mandates and scientific realities conflict when existing legal principles do not match the realities and limits of current science. Those conflicts can be addressed if scientists communicate the limits of existing scientific capabilities and if the legal system responds accordingly. One example of that phenomenon was the Federal statutory response to the limits of science in toxic tort litigation. Scientists charged with restoration of Colorado River ecosystems work within an ambiguous and limiting legal framework. Federal statutes and other legal authorities governing Colorado River management often present conflicting goals and requirements and assume that existing patterns of water and energy use are inviolate. Colorado River restoration efforts also face physical impediments because of historical development along the river, alterations in hydrology, and other factors that are difficult or impossible to reverse. One key role of scientists in the legal and policy process is to communicate those limitations clearly. Based on that information, the legal and legislative communities could alter the existing law-science paradigm governing restoration programs. A broader concept of environmental restoration would seek replacements for some of the key resources currently drawn from the river (such as water and energy) and that currently limit restoration efficacy.

Introduction

Scientists working to restore complex ecosystems express frustration when asked to give definitive answers to complex issues in the face of uncertainty, or to answer questions that cannot be answered given current knowledge or methods, or to answer questions that cannot be answered by science alone—what nuclear physicist Alvin Weinberg referred to as "trans-science" (Weinberg, 1972). In the context of minimum viable population estimates for species, conservation biologist Michael Soulé wrote: "[T]he quest for a simple bottom line is ... a question for a phantom by an untrained mind" (Soulé, 1986).

In part, other authors have suggested that this frustration reflects a "culture clash" between law and science. Sheila Jasanoff explained that "[S]cience seeks truth, while the law does justice; science is descriptive, but the law is prescriptive; science emphasizes progress, while the law emphasizes process" (Jasanoff, 1995). David Faigman wrote: "Science explores what is; the law dictates what ought to be. Science builds on experience; the law rests on it. Science welcomes innovation, creativity, and challenges to the status quo; the law cherishes the status quo" (Faigman, 1999). Lawyers often rely on enforceable legal rules and presumptions to generate stability and certainty in the face of factual uncertainty so that individuals and businesses can make decisions and invest resources with some degree of security (Adler, 2003). Scientists see the world as complex, changing, and uncertain. They test theories against the best available information, articulate hypotheses that best fit that existing knowledge, and revise those theories as better information becomes available. There is no absolute "truth" or finality.

Finality versus "Truth" in Private Litigation

The legal perspective makes sense when applied to situations in which certainty and finality are more important than the ultimate truth, and traditional common law doctrines generally have adopted that approach. A good example is a commercial transaction such as a sales contract. The contract identifies factors such as what is being sold, the price, the delivery date, and who is paying for the shipping. Even the simplest of commercial transactions, however, involve risk. The market price may change between the contract date and the transaction date, in which case one party wins and the other loses. If there is a supply shortage, the seller may not be able to deliver the goods. There may be a risk of loss or damage in transit. Parties manage those risks through educated guesses, but understand that certainty is impossible. If they

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guess wrong, “a deal is a deal.” Although the result may be harsh, business cannot proceed without the certainty provided by the law of contacts. Stability is more important than getting the “right” answer. Moreover, as between two private entities in an arm’s length transaction, society has less interest in who “wins” than in the fairness of the process.

A more disturbing example, however, might involve a toxic tort lawsuit (personal injury case arising from exposure to toxic substances). Two parents allege that their child has a terrible birth defect because of the mother’s exposure to a toxic chemical during pregnancy. The child lives with considerable pain, serious learning disabilities, and large ongoing medical and special education expenses. The relevant science is uncertain, making the case difficult to prove. There is little epidemiological evidence. Other factors may have caused or contributed to the birth defect. Toxicological evidence is debatable because of uncertainty about extrapolations from higher dose laboratory animal exposures to lower dose human exposures. Scientists might study the problem, develop one or more hypotheses, structure experiments to test those hypotheses and collect more data, and revisit the hypotheses as more knowledge and understanding are gained.

Traditional common law tort doctrines and related rules of civil litigation address the problem much differently. A “statute of limitations” requires the claim to be filed within a fixed time after the injury is discovered. The case can be dismissed for lack of diligent prosecution, meaning the plaintiff cannot wait years for additional scientific proof to be developed. The law assigns the “burden of proof” to the plaintiff in civil lawsuits, requiring them to prove their case by a “preponderance of the evidence,” meaning that it is *more likely than not* that the material facts are true (including the “fact” that the toxic exposure caused the birth defect). This standard guarantees that a large number of cases will be decided incorrectly. Scientists may not find that result surprising because, as between competing theories, the scientific process seeks the one that best explains available data and other factors with the understanding that the ultimate truth may be elusive. The key difference, however, is the legal doctrine of *res judicata* (“things adjudicated”). Once a case is decided, after a fair process and all available appeals are exhausted or waived, the matter is closed. Typically, the matter cannot be reopened *even if subsequent information suggests that the result was wrong*.

This process can lead to seemingly unfair outcomes. If the defendant loses, it may pay millions of dollars in damages for an injury caused by some other factor, or combination of factors, or bear the full costs of a harm for which it was only partially responsible. If the plaintiffs in the example above lose, the parents may not be able to afford proper medical care or remedial education. The rationale for this result is that the value of finality trumps the search for truth. Statutes of limitations allow people and businesses to move on without a perpetual shadow of liability, and resources devoted to insurance can be invested in other ways. A different standard of proof would favor one party in civil litigation, in which society cares more about a fair dispute resolution process than in the ultimate outcome.[2] *Res judicata* is often the bitterest pill for non-lawyers to swallow. Even if a new study released a year later provides much stronger evidence of causation (or lack thereof), the case cannot be reopened. The value of finality allows people to get on with their lives free of the permanent risk of uncertain liability.

Shifting the Law-Science Paradigm in the Public Law Context

The standard law-science paradigm is less helpful in describing the evolving interaction between law, science, and policy in public decisions and processes involving many more interests and the public at large. Key examples include decisions and processes about ecosystem restoration and conservation biology. In those realms, the law-science paradigm is shifting in various ways, especially as statutory approaches have supplemented or replaced common law approaches. One example, familiar to scientists involved in large-scale ecosystem restoration, is adaptive management, in which decisions are not viewed as “final” but rather as hypotheses to be tested and revised on the basis of structured iterations of management experiments, data collection, and feedback. This process has been described in similar terms by experts in science (Walters and Holling, 1990), policy (Lee, 1993), and law (Keiter, 2003).

A second example, and the main thesis here, is that where legal mandates and scientific realities present irreconcilable conflicts, which are useful in response to the search for a new law-science paradigm, just as scientists develop new paradigms to address irreconcilable conflicts between existing theories and new data. A good example is the public law (statutory) response to the toxic tort dilemma, in which it is difficult to meet traditional legal standards of causation because of uncertainty in the sciences of toxicology and epidemiology and the presence of confounding variables that may have caused or contributed to the injury. We did not abandon the law of toxic torts, and private remedies remain for plaintiffs who can meet the applicable burden of proof and other requirements. However, Congress elected to address the problem of exposure to toxic substances at a different level and from a different perspective by adopting regulatory statutes to prevent exposures to *potentially* dangerous substances rather than waiting for proof of harm. The Federal Clean Water Act prohibits the discharge of any pollutant from a point source without a permit, and without applying the

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[2] In criminal law cases, society does articulate a strong preference by imposing a much stricter standard of proof in which the government must prove its case “beyond a reasonable doubt.” This standard reflects the societal preference that it is better to let a guilty person go free than to deprive an innocent person of life or liberty.

best available treatment technology, absent any showing of harm or causation (Clean Water Act §301, 33 U.S.C. §1331). Courts ruled that Congress' regulatory approach was to target endangerment rather than demonstrated harm (Reserve Mining Co. v. Environmental Protection Agency, 514 F.2d 492, 8th Cir. 1975). Similarly, the Federal Superfund statute requires responsible parties to clean up contaminated sites or to compensate cleanup costs incurred by others based on strict liability, that is, without the need to prove causation or harm (Comprehensive Environmental Response, Compensation, and Liability Act, §107, 42 U.S.C. §9607).

This shift in the law applicable to toxic substances provided a better match between what the law requires and what science can reasonably provide. It eliminated the need to prove causation in situations where risk of harm was likely but proof of harm was elusive. It reversed the burden of proof by requiring dischargers to prove compliance with applicable treatment requirements before being allowed to discharge pollutants, rather than requiring injured parties to prove harm in order to prevent exposures from occurring. Most important for Colorado River restoration, the new approach moved upstream to tackle the root cause of the problem—exposure to toxic substances—to avoid perplexing issues of scientific proof and uncertainty that prevailed under the existing law-science paradigm.

Two key precursors helped generate this paradigm shift. The scientific community had to be honest about the limitations of current methods and understanding. This idea is fundamental to normal scientific research, in which good scientists report both the results and limitations of their research. Trials and other legal processes, however, often pit scientific experts against one another and may inhibit the willingness of each party to concede uncertainty in their respective positions. The legal community must be willing to change the legal paradigm applicable to the relevant problem, sometimes requiring a different set of societal choices and priorities. It is the *interaction* between the scientific and legal processes that is critical, however, because the wisdom of those choices turns in part on the quality of the scientific input.

Implications for Colorado River Restoration and Management

The Current Law-Science Paradigm for Colorado River Restoration

The legal regime governing Colorado River restoration is far more complex than can be summarized here. Using the Grand Canyon reach of the river as an example, however, two significant driving factors are Section 7 of the Federal Endangered Species Act (ESA) and the Grand Canyon Protection Act. Each of these statutes illustrates problems with the existing law-science paradigm governing Colorado River restoration and the manner in which science might inform a shift in that paradigm.

Section 7 of the ESA provides, in relevant part, that "… all federal agencies must take such action as is necessary to insure that actions authorized, funded, or carried out by them do not jeopardize the continued existence of" threatened or endangered species (ESA §7, 16 U.S.C. §1536). A key scientific issue suggested by this language is what level of impairment in any given circumstance constitutes "jeopardy" to the listed species. Assuming that a jeopardy determination is made, the secondary question is whether the action—in this case operation of Glen Canyon Dam—may continue based on "reasonable and prudent alternatives" sufficient to avoid jeopardy. Although the Supreme Court ruled early in the history of the ESA that Congress intended Section 7 to be interpreted strictly (Tennessee Valley Authority v. Hill, 437 U.S. 153, 1978), in practice most Section 7 decisions consider whether a balance can be struck between human economic activity supported by the Federal action and "reasonable and prudent alternatives" to mitigate the effects of the action on the listed species.

Congress sought a similar "win-win" balance in more specific legislation governing operation of Glen Canyon Dam. In the Grand Canyon Protection Act (GCPA), Congress directed the Bureau of Reclamation, guided by a multi-interest group advisory committee, to operate the dam "in such a manner as to protect, mitigate adverse impacts to, and improve the values for which Grand Canyon National Park [was] established…," and to implement the Act "in a manner fully consistent with and subject to" the Colorado River Compact and other components of the Law of the River (Grand Canyon Protection Act, Public Law 102–575, October 20, 1992, 106 Stat. 4600).

Agency officials articulate similar goals in the program documents describing the Section 7 process being used to oversee the Upper Colorado River Endangered Fish Recovery Program and for an "incidental take" permit issued for the Lower Colorado River Multi-Species Conservation Program (LCR MSCP) under Section 10 of the ESA. The upper Colorado program is designed to "recover the endangered fishes while providing for existing and new water development to proceed" (U.S. Fish and Wildlife Service, 2000). Likewise, the LCR MSCP aspires to prevent species extinction but also to "accommodate present water diversions and power production and optimize opportunities for future water and power development, to the extent consistent with the law" (Lower Colorado River Multi-Species Conservation Program, 2004). The science/policy/management dilemma this poses is whether it is really possible to meet the economic goals of water law and development and the environmental goals in the ESA and GCPA, fully and simultaneously.

Status of Existing Restoration Programs

All three major ecosystem restoration programs for the Colorado River have made significant efforts to restore species and habitats in the face of perplexing scientific and management challenges. With varying degrees of on-the-ground success, programs have been designed and implemented to control invasive species of plants and fish; replant native riparian vegetation; reconnect main channels to backwaters and flood plains; restore the level, timing, and temperature of instream flows; restock native fish and take steps to ensure their survival and reproduction; and facilitate movement of fish by installing fish passages and other structures. All are sound strategies and are either useful or essential to ecosystem recovery. However, at least to date, the programs have not succeeded in achieving the defined program goals or satisfying the applicable legal standards in the ESA and the GCPA (Gloss and others, 2005; Adler, 2007). Similar problems face other large aquatic ecosystem programs (Doyle and Drew, 2008).

One possible conclusion, especially given the adaptive management strategy adopted for all three programs, is that more time, study, and learning are necessary to modify restoration efforts until success is achieved. A more sober possibility is that the current law-science paradigm seeks impossible results under the circumstances. At least in some reaches of the Colorado River, perhaps conditions are altered to such a degree that existing restoration efforts alone, conducted within the constraints of current water law and policy, will not be sufficient to meet restoration goals. One candid scientific assessment (Mueller and Marsh, 2002) advised:

> The future is grim for native fish in the Lower Colorado River. Remnant native fish communities continue to decline, except for small refugium populations. Their fate has been sealed by the dependence on the river by 30 million water users in the United States and Mexico. *Societies' dependence on water makes native fish recovery economically and politically unlikely, and perhaps impossible*.

Several sets of anthropogenic conditions impose significant impediments to Colorado River restoration. First, human water diversions and a history of overly optimistic planning assumptions limit the amount of water available for in-stream use and restoration. The commissioners who negotiated the 1922 Colorado River Compact falsely assumed reliable average runoff in the basin sufficient to allocate at least 16.5 million acre feet (maf) of water per year, although they understood the need for significant storage capacity to buffer the impact of low water years (Meyers, 1966). Tree ring histories suggest that average flows in the basin over the past several centuries have been significantly lower than the compact assumptions (Woodhouse and others, 2006), and the hydrological impacts of climate change may reduce future runoff even further (Christensen and Lettenmaier, 2006). Development rates and patterns have been different than predicted at the time of the compact, and the upper basin States have yet to develop their full compact share. The key legal question is whether the upper basin States will get the benefit of the bargain they struck in 1922—i.e., relief from the prior appropriation doctrine of western water law—under which the lion's share of the river would have gone to more rapidly developing California (Adler, 2007). The key question for scientists is the extent to which these hydrological realities will limit the efficacy of restoration efforts.

Second, the basinwide system of dams, diversions, levees, and other physical structures built to facilitate land and water development causes hydrological and physical habitat changes that are difficult to address through minor operational modifications to that infrastructure. Those facilities alter the flow and timing of water as well as other key constituents in the aquatic environment, such as sediment, nutrients, and organic matter, and also change patterns of temperature and water-quality characteristics. One question in Grand Canyon restoration, for example, is whether sediment input below Glen Canyon Dam suffices to support long-term habitat restoration. If not, the only real solutions may be either to decommission the dam or to transport sediment stored in Lake Powell downstream.

Third, development in the river's flood plains and riparian zones, especially along the lower river, impede efforts to restore native vegetation and habitats. Existing restoration pursuant to the LCR MSCP involves labor-intensive, expensive efforts to replant relatively small areas with native plant communities, the long-term efficacy of which is inconsistent and uncertain. Even if many or all of those efforts succeed, insufficient habitat will be restored to make a real difference. Along the approximately 500 river miles below Hoover Dam, reservoirs inundate 210,000 acres of riparian habitat, approximately 300,000–350,000 riparian acres are developed, and only 23,000 acres of native vegetation remain. Against that background of losses, the LCR MSCP establishes a goal of restoring just 8,000 acres of new habitat (Lower Colorado River Multi-Species Conservation Program, 2004; Adler, 2007). Restoring natural flood regimes might be a much more successful and cost-effective strategy to restore ecosystem structure and function over a much larger area. We cannot, however, promote natural flood regimes in developed areas. The "trans-science" issue, which can be informed but not answered by the relevant science, is whether we should spend so much time and money on restoration efforts that are so constrained by existing conditions.

Implications for Colorado River Restoration

These circumstances suggest difficult choices for Colorado River restoration and management. We could accept that some places are irrevocably altered and forego restoration efforts altogether. After all, no one suggests that we try to restore the native ecosystems of Manhattan Island, and it is not prudent to use limited resources where restoration efforts are not likely to succeed. Alternatively, we could adopt more limited restoration goals. For example, in some portions of the watershed we could restore acceptable fishery habitats, but not necessarily for native species. Or, we could explore ways to undo some of the fundamental anthropogenic environmental factors that limit current restoration efforts.

All three choices would require some shift in the law-science paradigm governing Colorado River restoration analogous to the shift that occurred when Congress augmented the common law regime of torts as applied to toxic substances with preventive statutory approaches. The first two choices would require significant amendments to the ESA and other environmental laws that historically have been supported by the public and a retreat from the longstanding belief that it is possible to enjoy economic benefits from the Colorado River without sacrificing its unique ecosystems and species. The third choice—one that maintains those commitments and the integrity of our environmental statutes—would require a paradigm shift similar to the one adopted with respect to toxic pollutants. That shift would entail a significant expansion of the *concept of restoration* to include changes to some of the background conditions that constrain existing restoration efforts. The new statutory approaches to toxic pollutants addressed uncertainty in proving causation after harm occurred by shifting from *post hoc* compensation to prevention. The new approach focused on root causes rather than mitigation of effects. Likewise, broader concepts of restoration would seek to alter root causes that currently impede restoration efforts and scientific uncertainty about how to mitigate those impediments. Three brief examples are presented below, but are not intended to be exclusive.

First, we could revisit various components of the Law of the River, including the Colorado River Compact, *as a restoration strategy* (Adler, 2008). The compact was an ingenious solution to the legal and practical problems the basin States faced in 1922. Like all legal arrangements, however, it can be changed to meet current realities. For example, some have proposed that we move the location of the upper basin States' delivery obligation from Lees Ferry to Hoover Dam (Richard J. Ingebretsen, University of Utah, oral commun., 2007). This move would eliminate the need for two huge storage reservoirs simply for purposes of meeting the compact's artificial delivery obligation. Lake Powell and Lake Mead are now well below capacity, and if long-term reductions in basin runoff are likely because of climate change, this may become normal rather than "drought" conditions. If so, maintaining both reservoirs in active status significantly increases the ratio of evaporative surface area to storage volume, thus reducing water supplies for both human and environmental purposes. From a restoration perspective, taking Glen Canyon Dam out of operation would result in a far longer stretch of free-flowing river through Grand Canyon to Lake Mead.

Second, we could rethink water use and management in the basin *as a restoration strategy*. Water use in the basin is dictated largely by storage capacity and by supply and demand. Given the highly seasonal runoff pattern and the significant variability in annual runoff in the basin, storage is essential for human uses. As discussed above, however, in-stream reservoirs are major impediments to restoration. One solution to this problem would be to shift much of the basin's storage capacity from in-stream storage to a combination of off-channel reservoirs (such as the Sand Hollow Reservoir in Utah) and aquifer storage and recovery (as is being used for the Arizona Water Bank). We do not know whether there is sufficient off-stream storage capacity in the basin to eliminate the need for one or more of the major in-stream reservoirs that currently constrain restoration programs. However, we similarly did not know the potential in-stream reservoir storage capacity in the basin until we sent hydrologists and engineers to investigate in the early 20th century. Aquifer storage and recovery is one component of ongoing efforts to restore the Everglades (Comprehensive Everglades Restoration Plan, 2001). A similar effort may be appropriate here, and we could fund that effort as part of Colorado River restoration programs.

Similarly, water use in the basin depends on supply and demand. One plausible restoration strategy is to purchase water subsidies and to dedicate the saved water back to the river. If applicable science indicates that insufficient water is a limiting factor in restoration efforts, purchasing water may use limited restoration dollars more effectively than some current strategies. Similarly, direct investments in water efficiency, as have occurred in the basinwide salinity program (Bureau of Reclamation, 2005), might result in a more effective use of restoration program resources. One study estimated that over 1 maf of cost-effective water savings are possible in Arizona alone through improved irrigation (Morrison and others, 1996). On the supply side, as desalination technologies and cost-effectiveness improve (National Research Council, 2008), investment in desalination plants in California might constitute an effective restoration strategy, if Colorado River water now diverted to the west coast is dedicated back to the river for restoration.

Third, we could rethink power use and generation *as a restoration strategy*. One benefit of hydroelectric power is that it is clean and does not produce greenhouse gases (GHGs) relative to coal or other fossil fuels. However, other renewable energy sources are available in large amounts in the Southwest. For example, the total solar-generating capacity in the Southwest is estimated to be equal to seven times the current U.S. power demand (National Renewable Energy Laboratory, 2007). Arizona alone has over 2.5 million megawatts of solar-generating capacity (equal to over 1,800 Glen Canyon Dams). Moreover, investments in energy efficiency can significantly and cost-effectively reduce electric power demand in the region. If the in-stream dams that contribute to the Southwest power load also impede the efficacy of restoration programs, restoration program dollars might be spent effectively to reduce demand or to generate power from other renewable sources.

Conclusion

This analysis suggests a potential dilemma for scientists (especially those working for government agencies). Scientists need not formulate changes in law and policy, or even propose them, to facilitate shifts in the law-science paradigm governing the Colorado River (although they are certainly not precluded from doing so). Scientists involved in restoration efforts simply need to provide good, reliable, and candid information about the relative success of restoration efforts and, more importantly, key impediments to success. Available science suggests that, absent elimination of fundamental impediments to restoration, current efforts will have limited success or will fail altogether. In the face of this information, one approach is for scientists and managers simply to do what was assigned and to let someone else worry about other issues and implications. A second approach is to view the role of scientists in a broader sense as providing the information necessary to ensure sound public decisions and investments. Candid scientific assessments of both the strengths and limitations of existing restoration strategies can help legislators, senior regulatory officials, judges, and other decisionmakers to decide whether different or additional strategies are necessary or appropriate. A third approach is to do more than just provide advice and information and to advocate actively for a broader set of actions needed to ensure restoration success. The danger of the first approach is that we might continue to delude ourselves into thinking that science can achieve the impossible. The second and third approaches could facilitate the kinds of shifts in the law-science paradigm that occurred in the toxic tort example, which might facilitate more productive strategies for Colorado River restoration programs.

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A Watershed Perspective of Changes in Streamflow, Sediment Supply, and Geomorphology of the Colorado River

By John C. Schmidt[1]

Abstract

More than a century ago, John Wesley Powell urged westerners to politically organize their region by watersheds, because he believed that the need to allocate the scarce water supply was a critical decision that ought to be shared among each watershed's inhabitants. The modern Colorado River system is a modified watershed linked by a comprehensive network of dams and diversions that are regionally managed for water supply and electricity production. The river rehabilitation programs of the Colorado River and its major tributaries, however, are not managed from a watershed perspective but nevertheless have a regional context. Typically, dams have reduced the magnitude and decreased the duration of floods, increased the magnitude of base flows, and trapped incoming sediment. In response, the post-dam sediment mass balance downstream from each dam has been perturbed into sediment deficit, and these channels typically have been evacuated of sediment. In some cases, evacuation has involved large-scale bed incision, but this has not occurred in debris fan-affected segments where there is abundant coarse bed material. Elsewhere, the post-dam sediment mass balance has been perturbed into sediment surplus, and parts of the upper Colorado River, downstream parts of the Green River, and short segments of the lower river that forms the Arizona-California border have accumulated sediment. The entire network has been subject to channel narrowing that is caused by decreases in the flood regime and invasion of riparian vegetation. These perturbations cause changes in channel size and flood-plain connection that constitute changes in aquatic and riparian habitat that contribute to the endangered status of some species comprising the Colorado River's native fishery. The analysis presented here demonstrates that the magnitude and style of perturbations to different parts of the river system vary widely. Thus, different approaches are required to rehabilitate geomorphic and habitat conditions in different parts of the river network. The primary goal of this paper is to inspire a comprehensive watershed-scale geomorphic and ecological assessment of the relative challenge of rehabilitating the river network.

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Introduction

Despite its modest discharge, the Colorado River is significant in terms of its utilization by human society. The Colorado River's reservoirs are larger in relation to mainstem streamflow than any other large watershed in North America (Hirsch and others, 1990), and diverted streamflow and hydroelectricity are used by more than 30 million people. Some of the dams in the watershed, such as Hoover (Stevens, 1988) and Glen Canyon (Martin, 1989), are nationally famous, as are the political debates that stopped the proposed dams at Echo Park (Harvey, 1994) and Marble Canyon (Pearson, 2002). Approximately 10 percent of the predevelopment streamflow now crosses the international border to Mexico, and most of this flow is diverted for irrigation and does not reach the Gulf of California.

The Colorado River is also significant in terms of its scenery, unique attributes of the riverine ecosystem, and the scientific studies conducted there. The Colorado Plateau portion of the watershed has the densest concentration of federally protected areas within the National Park System. Approximately 7.9 x 10^6 people visited Lake Mead National Recreation Area and its reservoir in 2008 (*http://www.nps.gov/lame/parknews/lake-mead-proves-popular-during-economic-downturn.htm*), and 4.3 x 10^6 people visited Grand Canyon National Park in 2009 (*http://www.nature.nps.gov/stats/viewReport.cfm*). The watershed's unique fishery has the highest degree of endemism of any large basin in North America (Minckley and Deacon, 1991). Many fundamental concepts in geomorphology were developed in concert with exploration of the Colorado Plateau (Powell, 1875; Gilbert, 1876, 1877; Dutton, 1880, 1881, 1882).

Today, more than \$4.0 x 10^7 is spent annually by four Federal-State-private collaborative programs that seek to recover endangered species or improve conditions of the native riverine ecosystem of the Colorado River or its headwater tributaries (table 1). Each of these river rehabilitation programs—the Upper Colorado River Endangered Fish Recovery Program, San Juan River Basin Recovery Implementation Program, Glen Canyon Dam Adaptive Management Program, Lower Colorado River Multi-Species Conservation Program (LCR MSCP)—is focused on a particular part of the channel network. The purpose of this paper is to provide the watershed context of these rehabilitation programs by summarizing an ever-growing body of research describing the present river ecosystem and its historical changes. A quantitative comparison of the relative perturbation of each river segment from its early 20th century condition is also presented and used to evaluate options for watershed-scale rehabilitation.

Table 1. Summary information concerning the large Colorado River rehabilitation programs.

Upper Colorado River Endangered Fish Recovery Program[a]	
Established in 1988	
\$9.5 million/yr (\$199 million; FY1989–FY2009)	
Program statement: recover the Colorado pikeminnow (*Ptychocheilus lucius*), razorback sucker (*Xyrauchen texanus*), humpback chub (*Gila cypha*), and bonytail (*Gila elegans*) “while allowing continued and future water development”	
Program goal: “Endangered Colorado pikeminnow, razorback suckers, bonytail and humpback chub will be considered recovered when there are self-sustaining populations of each fish species and when there is natural habitat to support them.” [specific recovery goals have been defined for each endangered species]	
Program partners:	
State of Colorado	The Nature Conservancy
State of Utah	U.S. Fish and Wildlife Service
State of Wyoming	Utah Water Users Association
Bureau of Reclamation	Western Area Power Administration
Colorado River Energy Distributors Association	Western Resource Advocates
Colorado Water Congress	Wyoming Water Development Association
National Park Service	
San Juan River Basin Recovery Implementation Program[a]	
Established in 1992	
\$2.4 million/yr (\$43 million/yr; FY1992–FY2009)	
Program statement: help recover the Colorado pikeminnow and razorback sucker while allowing water development to continue in the San Juan River Basin	
Program goals:	
(1) conserve populations of Colorado pikeminnow and razorback sucker in the San Juan Basin consistent with recovery goals established under the Endangered Species Act, 16 U.S.C. 1531 et seq.	
(2) proceed with water development in the San Juan Basin in compliance with Federal and State laws, interstate compacts, Supreme Court decrees, and Federal trust responsibilities to the Southern Ute, Ute Mountain Ute, Jicarilla, and the Navajo Tribes.	
Program partners:	
State of Colorado	Bureau of Indian Affairs
State of New Mexico	Bureau of Land Management
Jicarilla Apache Nation	Bureau of Reclamation
Navajo Nation	The Nature Conservancy
Southern Ute Indian Tribe	U.S. Fish and Wildlife Service
Ute Mountain Ute Tribe	Water development interests

Table 1. Summary information concerning the large Colorado River rehabilitation programs.—Continued

Glen Canyon Dam Adaptive Management Program
Established in 1996
$13.8 x 10^6/yr (FY2008)[b]
Program goal: help the Federal government understand the relationship between dam operations and the health of the Colorado River ecosystem downstream from Glen Canyon Dam so that the Federal government can the meet its resource management obligations under the 1992 Grand Canyon Protection Act, the 1995 Glen Canyon Dam Environmental Impact Statement, and the 1996 Record of Decision
Members:
Cooperating agencies:
Arizona Game and Fish Department
Bureau of Indian Affairs
Bureau of Reclamation
Department of Energy
Hopi Tribe
Hualapai Tribe
National Park Service
Navajo Nation
Pueblo of Zuni
San Juan Southern Paiute Tribe
Southern Paiute Consortium
U.S. Fish and Wildlife Service
Environmental groups:
Grand Canyon Trust
Grand Canyon Wildlands Council
Federal power purchase contractors:
Colorado River Energy Distributors Association
Utah Associated Municipal Power Systems
Recreation interests:
Federation of Fly Fishers
Grand Canyon River Guides
Colorado Basin States:
Arizona Department of Water Resources
Colorado Department of Water Resources
Colorado River Board of California
Colorado River Commission of Nevada
New Mexico Interstate Stream Commission
Utah Division of Water Resources
Wyoming

Table 1. Summary information concerning the large Colorado River rehabilitation programs.—Continued

Lower Colorado River Multi-Species Conservation Program[c]

Established in 2005

$15.8 x 10^6/yr (FY2008 actual costs; total authorized program costs are $626 x 10^6 for 50-yr period in 2003 dollars; actual costs to be adjusted for inflation)

Program goals:
- (1) protect the lower Colorado River environment while ensuring the certainty of existing water and power operations,
- (2) address the needs of threatened and endangered wildlife under the Endangered Species Act, and
- (3) reduce the likelihood of listing additional species along the lower Colorado River

Steering committee members:

- Federal participants:
  - Bureau of Reclamation
  - U.S. Fish and Wildlife Service
  - National Park Service
  - Bureau of Land Management
  - Bureau of Indian Affairs
  - Western Area Power Administration
- Arizona participants:
  - Arizona Department of Water Resources
  - Arizona Electric Power Cooperative, Inc.
  - Arizona Game and Fish Department
  - Arizona Power Authority
  - Central Arizona Water Conservation District
  - Cibola Valley Irrigation and Drainage District
  - City of Bullhead City
  - City of Lake Havasu City
  - City of Mesa
  - City of Somerton
  - City of Yuma
  - Electrical District No. 3, Pinal County, Arizona
  - Golden Shores Water Conservation District
  - Mohave County Water Authority
  - Mohave Valley Irrigation and Drainage District
  - Mohave Water Conservation District
  - North Gila Valley Irrigation and Drainage District
  - Town of Fredonia
  - Town of Thatcher
  - Town of Wickenburg
  - Salt River Project Agricultural Improvement and Power District
  - Unit "B" Irrigation and Drainage District
  - Wellton-Mohawk Irrigation and Drainage District
  - Yuma County Water Users' Association
  - Yuma Irrigation District
  - Yuma Mesa Irrigation and Drainage District
- California participants:
  - California Department of Fish and Game
  - City of Needles
  - Coachella Valley Water District
  - Colorado River Board of California
  - Bard Water District
  - Imperial Irrigation District
  - Los Angeles Department of Water and Power
  - Palo Verde Irrigation District
  - San Diego County Water Authority
  - Southern California Edison Company
  - Southern California Public Power Authority
  - The Metropolitan Water District of Southern California
- Nevada participants:
  - Colorado River Commission of Nevada
  - Nevada Department of Wildlife
  - Southern Nevada Water Authority
  - Colorado River Commission Power Users
  - Basic Water Company
- Native American participants:
  - Hualapai Tribe
  - Colorado River Indian Tribes
  - The Cocopah Indian Tribe
- Conservation participants:
  - Ducks Unlimited
  - Lower Colorado River RC&D Area, Inc.
- Other interested parties:
  - QuadState County Government Coalition
  - Desert Wildlife Unlimited

[a] Upper Colorado River Endangered Fish Recovery Program [*http://www.fws.gov/coloradoriverrecovery/*] and San Juan River Basin Recovery Implementation Program [*http://www.fws.gov/southwest/sjrip/*] (2009).

[b] Bureau of Reclamation and U.S. Geological Survey (2008) and *http://www.gcmrc.gov/*.

[c] Lower Colorado River Multi-Species Conservation Program (2009, p. 7) and *http://www.lcrmscp.gov/*.

Hydrology and Sediment Supply Before Dams

Most of the Colorado River's streamflow enters the drainage network as snowmelt in three tributary watersheds in the middle and southern Rocky Mountains (fig. 1). The longest of these tributaries is the Green River, which has two co-equal forks, in terms of streamflow, that join at Echo Park in northwestern Colorado. The upper Green River drains the Wind River Range, Wyoming Range, and part of the Uinta Range of the middle Rocky Mountains, and the Yampa River drains part of the southern Rockies in northern Colorado. The tributary watershed with the largest unit runoff is the upper Colorado River, once called the Grand River. This watershed, including the Gunnison River that is its major tributary, drains most of the southern Rocky Mountains in central and southern Colorado. The San Juan River drains the southern part of the San Juan Mountains. These three headwater tributaries join to form the mainstem Colorado River in southeastern Utah. The only significant tributary further downstream, in terms of streamflow, is the Gila River.

Mean annual runoff in the Rocky Mountains is between 300 and 1,000 millimeters (mm) (Riggs and Wolman, 1990), and 54 percent of the total annual mainstem flow enters the network in the 15 percent of the basin comprising the exterior

A.

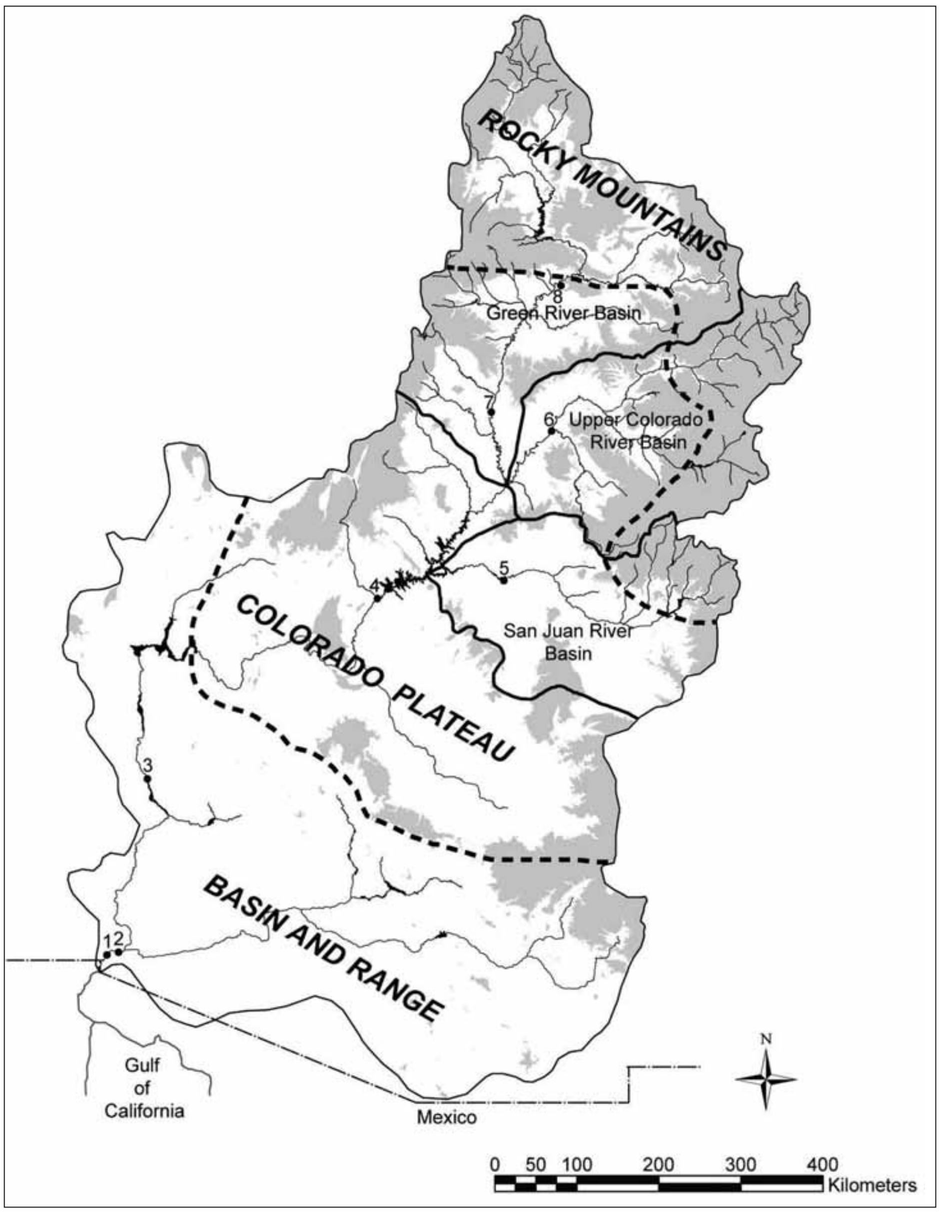


Figure 1. Colorado River Basin. (A) The three major headwater drainage basins and the three major physiographic provinces (Graf, 1987) that occur in the basin. Shaded areas are higher than 2,000 meters. Gaging stations referred to in the text are indicated by numbers: 1. Colorado River at the northern international border (NIB), 2. Colorado River at Yuma, 3. Colorado River at Topock, 4. Colorado River at Lees Ferry, 5. San Juan River near Bluff, 6. Colorado River at Cisco, 7. Green River at Green River, UT, 8. Green River at Jensen.

B.

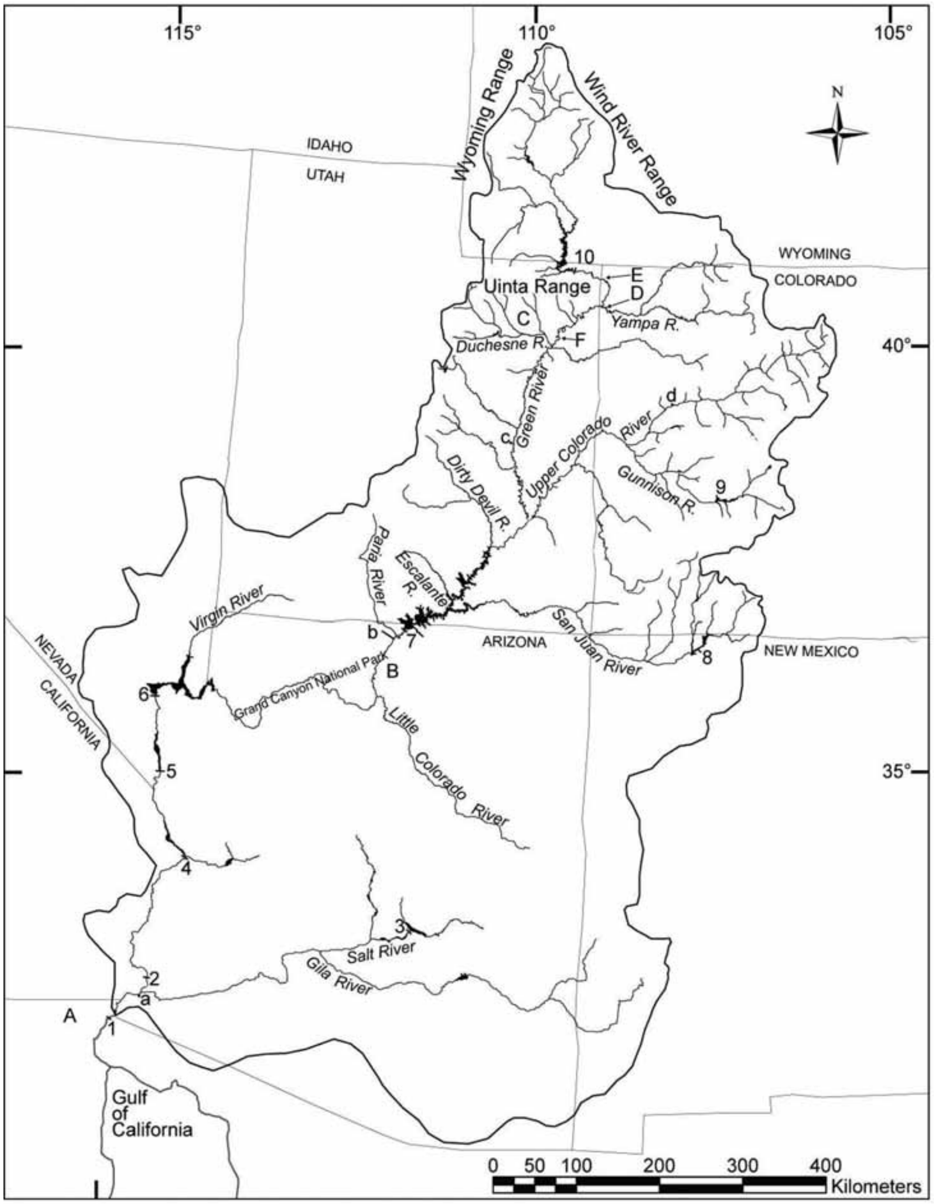


Figure 1. (continued) Colorado River Basin. (B) Locations mentioned in the text. 1. Morelos Dam, 2. Imperial Dam, 3. Theodore Roosevelt Dam, 4. Parker Dam, 5. Davis Dam, 6. Hoover Dam, 7. Glen Canyon Dam, 8. Navajo Dam, 9. Aspinall Unit, 10. Flaming Gorge Dam. A. Mexacali Valley, B. Marble Canyon, C. Uinta Basin, D. Echo Park, E. Browns Park National Wildlife Refuge, F. Ouray National Wildlife Refuge. a. Yuma, b. Lees Ferry, c. Green River, UT, d. Rulison.

watershed margin (fig. 2). The remaining 85 percent of the watershed upstream from Lees Ferry has unit runoff less than 50 mm. Before the construction of large dams, the peak flow at Lees Ferry typically occurred between late May and late June. The pre-dam, annual flood typically passed from Lees Ferry to the Gulf of California without significant change in magnitude or duration (Topping and others, 2003; Schmidt, 2007).

The Colorado River system has experienced periods of drought and times when runoff was high. The dendrohydrologic record of the Colorado River at Lees Ferry has been extended back to A.D. 762, and the mean annual runoff for the period between 1490 and 2005 is approximately $1.79 \pm 0.02 \times 10^{10}$ cubic meters (m^3) (Woodhouse and others, 2006). Pontius (1997) estimated that the long-term average annual flow entering the Colorado River's delta was 1.85×10^{10} m^3.

The Colorado River delivered about 1.0×10^8 megagrams per year (Mg/yr) of fine sediment to the Gulf of California in the beginning of the 18th century (Meade and others, 1990). Only the Mississippi River delivered more sediment from North America to the sea before extensive European settlement. The major sources of fine sediment to the

Colorado River are in the Colorado Plateau and Basin and Range Physiographic Provinces, downstream from the Rocky Mountains (fig. 3). Before construction of large dams, the average concentration of suspended fine sediment increased from the water-producing basin rim to the arid, sediment-producing central part of the watershed. Of the estimated pre-dam sediment load, approximately 27 percent came from the Green, 20 percent from the upper Colorado, and 20 percent from the San Juan River (Iorns and others, 1965). The rest came from the Dirty Devil, Escalante, Paria, Little Colorado, and Virgin Rivers, even though these streams deliver insignificant amounts of streamflow.

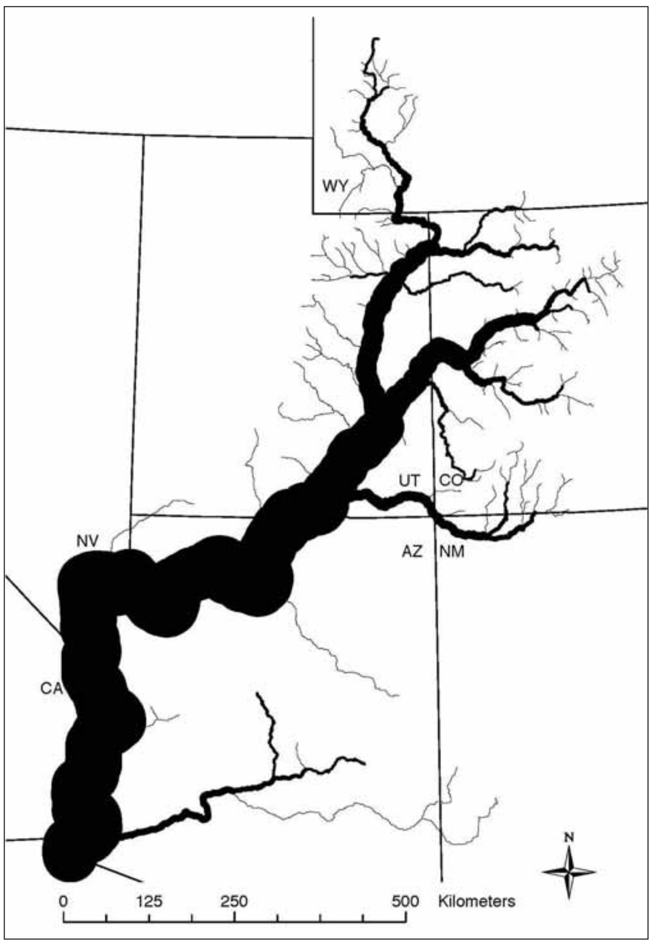


Figure 2. Relative amounts of streamflow in different segments of the pre-dam Colorado River system (reproduced from Schmidt, 2007). The majority of streamflow originated in the Rocky Mountains. The width of river segments is proportional to the widest line segment, which represents 510 cubic meters per second at the U.S.-Mexico border. Data are compiled from Iorns and others (1964) and pre-dam streamgaging records of the U.S. Geological Survey.

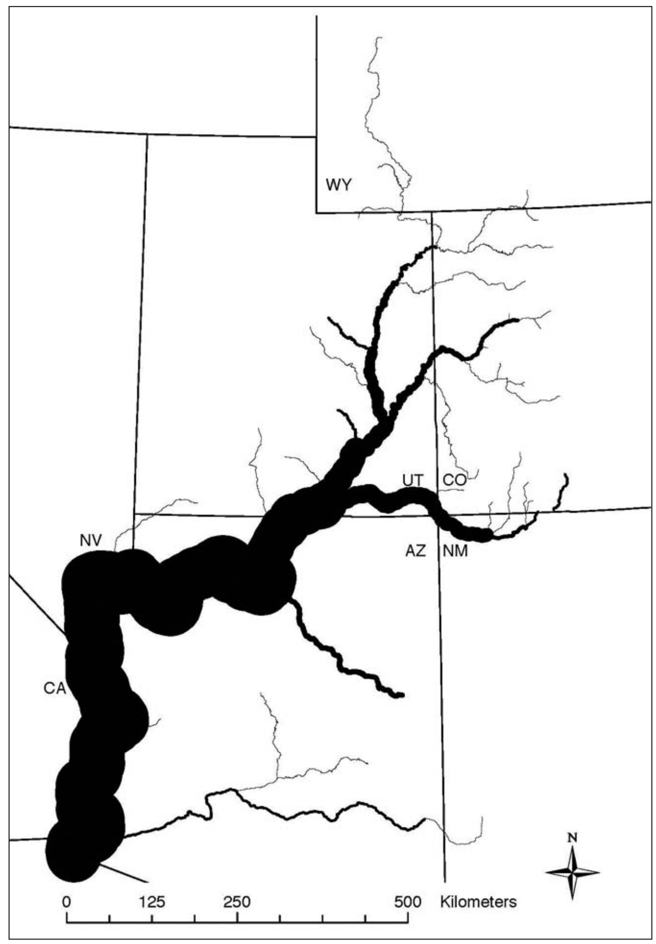


Figure 3. Relative amounts of suspended sediment in transport in the pre-dam river system (reproduced from Schmidt, 2007). These data are the estimates of Iorns and others (1965) for the period 1937–1955 and U.S. Geological Survey streamgaging stations downstream from Lees Ferry for the same period. The width of river segments is proportional to the widest line segment, which represents 1.53 x 10^7 megagrams per year at the U.S.-Mexico border.

Hydrology and Sediment Supply After Dam Construction

The first major water storage dam in the watershed was Theodore Roosevelt, completed in 1911, on the Salt River, a tributary of the Gila River. Hoover Dam, completed in 1936, was the first large dam on the mainstem, and its completion increased cumulative watershed reservoir storage to more than 4.0×10^{10} m^3 (fig. 4). Lake Mead is still the largest reservoir in the United States. In 1938, Parker and Imperial Dams were completed downstream from Hoover Dam (fig. 5), thereby facilitating large-scale diversions to southern California. Three additional diversion dams were completed on the lower Colorado River in 1944, 1950, and 1957, and Davis Dam was completed in 1953. Construction of large dams upstream from Lees Ferry was authorized by the Colorado River Storage Project Act of 1956, and these dams were completed in the mid-1960s. The total volume of reservoir storage is now 1.1×10^{11} m^3, which is nearly 7 times the long-term mean annual flow at Lees Ferry. Total basin consumptive uses are now about 1.5×10^{10} m^3, about 90 percent of the long-term average annual flow at Yuma. Consumptive uses upstream from Lees Ferry are about 5.0×10^9 m^3, or about 30 percent of the long-term annual flow at Lees Ferry. Total reservoir storage upstream from Lake Powell is 1.8 times the mean annual flow at Lees Ferry.

The transformation of the streamflow and sediment supply regimes caused by these reservoirs and by diversions has been profound (table 2). The transformation occurred earlier and the magnitude of the changes in streamflow was larger downstream from Hoover Dam, because dams there were built earlier and the total upstream reservoir storage is larger. Floods through Grand Canyon decreased greatly when Glen Canyon Dam was completed in 1963, and peak releases from the dam typically are less than 800 cubic meters per second (m^3/s) (fig. 6D). The largest dam releases have occurred when there was large snowmelt from the Rocky Mountains and when reservoir storage was full (1983, 1984, 1985, 1986), in order to create controlled floods, the purpose of which is rehabilitation of the downstream riverine ecosystem (1996, 2004, 2008), and for engineering tests or maintenance purposes (1965, 1980). Floodwaters are subsequently stored in Lake Mead. Floods on the lower Colorado River have been relatively low since 1936 when Hoover Dam was completed. Annual peak flow near Topock, AZ, downstream from Hoover and Davis Dams, typically does not exceed 900 m^3/s (fig. 6E). The largest flood released from Hoover Dam was 1,440 m^3/s in 1983, 53 percent of the pre-dam 2-year recurrence flood. Flood flows at Yuma, AZ, near where the Colorado River enters Mexico, are almost completely gone (fig. 6F).

The average hydrograph of the lower Colorado River and in Grand Canyon no longer shows a consistent, long-duration flood season, and base flows are much higher than they once

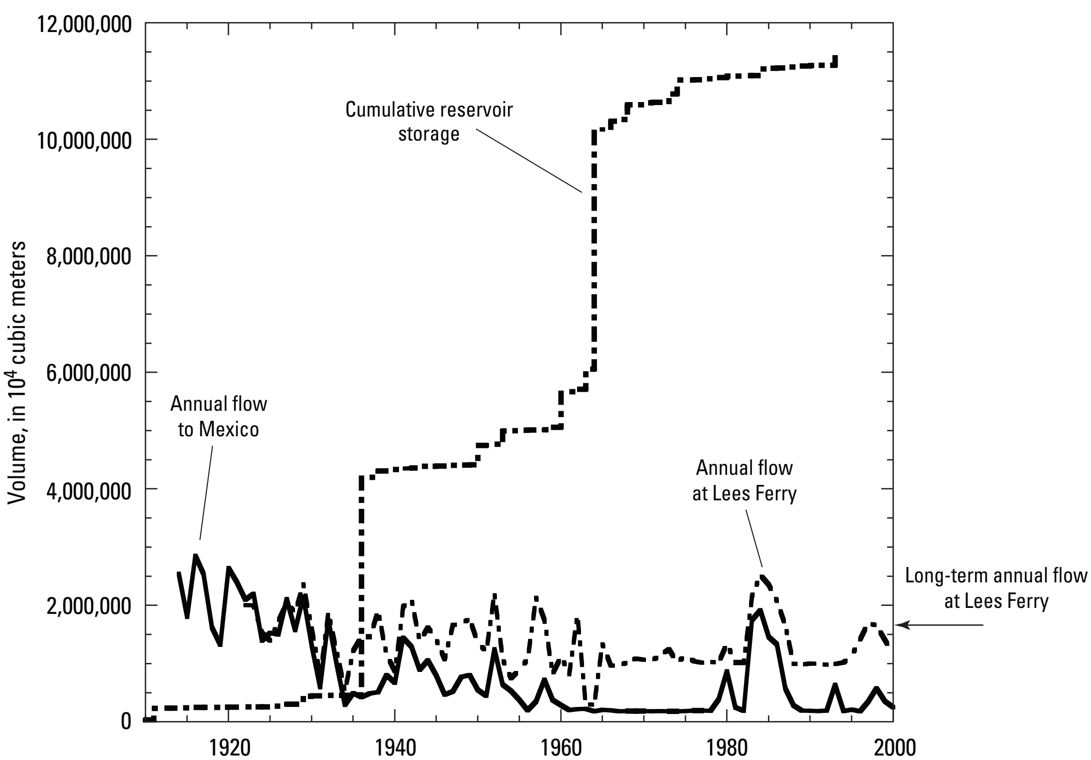


Figure 4. Cumulative reservoir storage in the entire watershed, annual flow at Lees Ferry, and annual flow crossing the international border to Mexico. The difference between annual flow at Lees Ferry and at the Mexican border is diverted to cities and towns, or is lost or stored in the regional groundwater system.

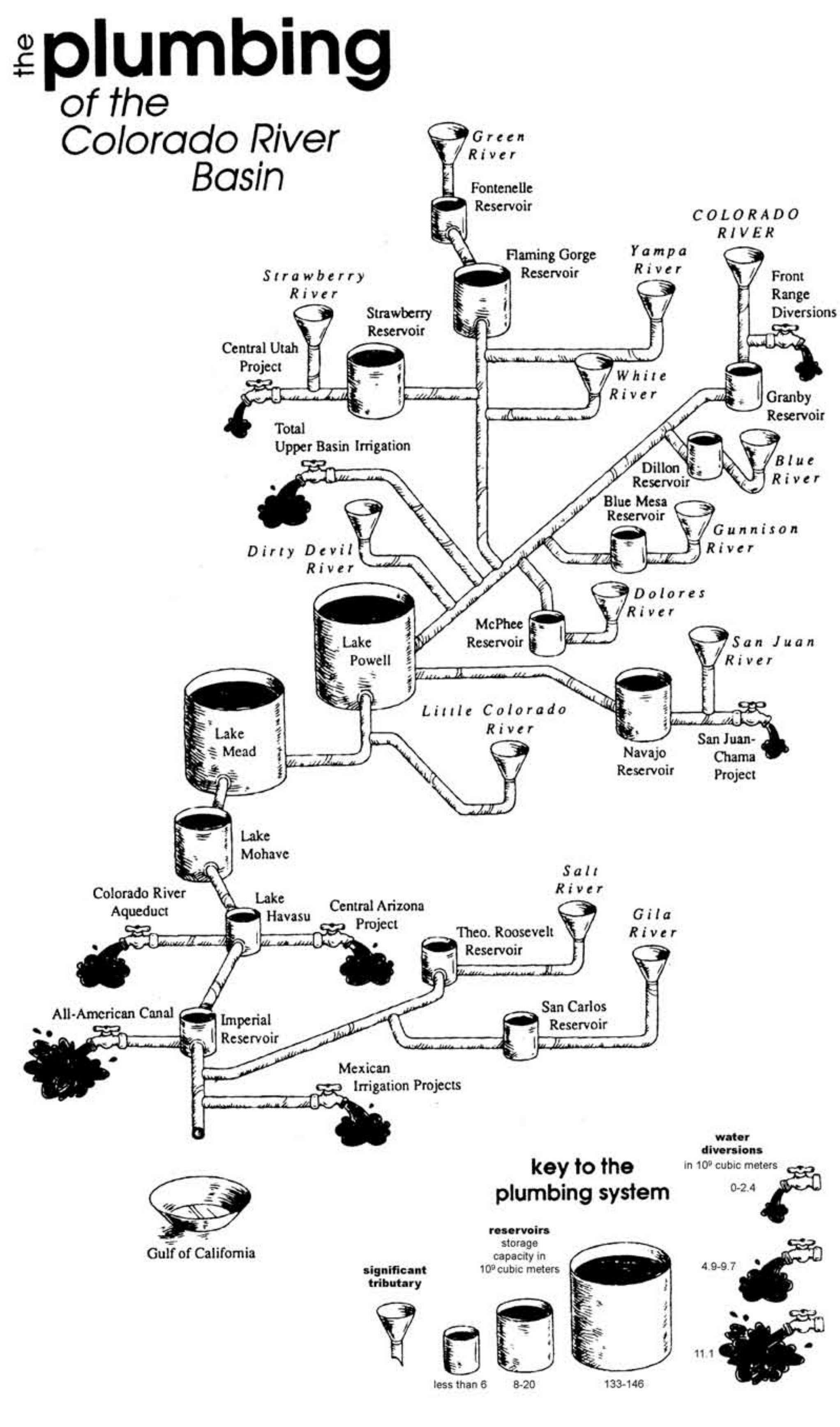


Figure 5. Distribution of reservoirs and diversions of the Colorado River system, depicted as a plumbing system. (Illustration by L. Dore and C. McKnight, reprinted with permission of High Country News.)

Table 2. Summary of changes in streamflow, sediment supply, and channel form in parts of the Colorado River system (data from Schmidt and Wilcock, 2008).

Segment number	River (Dam)	Location of upstream and downstream ends of reach, in kilometers downstream from dam	Attributes of streamflow and sediment supply				Attributes of the channel					Attributes of channel change	
			Pre-dam 2-year recurrence flood, in cubic meters per second	Post-dam 2-year recurrence flood, in cubic meters per second	Pre-dam mean annual sediment delivery, in million megagrams	Post-dam mean annual sediment delivery, in million megagrams	Pre-dam reach-average slope, in meters per meter	Post-dam reach average slope in meters per meter	Pre-dam bed material size, in meters	Post-dam bed material size, in meters	Mean flow depth at post-dam flood at nearest gage, in meters	Measured post-dam change in bed elevation, in meters	Ratio of post-dam width to pre-dam width
1	Colorado River (many dams)	0–37 [a]	725 (Cr1)	517 (Cr1)	1.6 (Cr2)	1.2 (Cr2)	NA	0.0020 (Cr3)	NA	0.058 (Cr3)	2.45	NA	0.92 (Cr2)
2	Colorado River (many dams)	67–90 [a]	725 (Cr1)	517 (Cr1)	1.6 (Cr2)	1.2 (Cr2)	NA	0.0018 (Cr3)	NA	0.058 (Cr3)	2.54	0.5 (Cr1)	0.79 (Cr4)
3	Gunnison River (Aspinall Unit)		490 (Cr1)	306 (Cr1)	1.5 (Cr4)	1.2 (Cr4)	NA	0.0012 (Cr4)	NA	0.046 (Cr4)	2.83	–0.6 (Cr1)	1.02 (Cr4)
4	Colorado River (many dams)	91–119 [a]	1,387	759	3.2	2.5 (Cr2)	NA	0.0013	NA	0.054 (Cr3)	3.01	NA	0.80 (Cr4)
5	Colorado River (many dams)	120–159 [a]	1,387	759	3.2	2.5 (Cr2)	NA	0.0010	NA	0.044 (Cr3)	3.64	0.4 (Cr1)	0.92 (Cr4)
6	Green River (Flaming Gorge)	0–18	339 (Cr5)	147 (Cr5)	0.46 (Cr6)	0.010 (Cr6)	NA	0.0021 (Cr6)	0.18 (Cr7)	0.18 (Cr7)	2.5	0 (Cr6)	0.95 (Cr6)
7	Green River (Flaming Gorge)	18–76	339 (Cr5)	147 (Cr5)	0.49 (Cr6)	0.030 (Cr6)	NA	0.00075 (Cr6)	NA	0.0010 (Cr7)	1.5	0 (Cr6)	0.81 (Cr6)
8	Green River (Flaming Gorge)	76–104	339 (Cr5)	147 (Cr5)	0.51 (Cr6)	0.052 (Cr6)	NA	0.0029 (Cr6)	0.18 (Cr7)	0.18 (Cr7)	2.0	0 (Cr5)	0.78 (Cr6)
9	Green River (Flaming Gorge)	105	626 (Cr5)	480 (Cr5)	1.6 (Cr6)	0.88 (Cr6)	NA	0.0010 (Cr5)	NA	0.025 (Cr7)	2.3	0 (Cr5)	0.90 (Cr5)
10	Green River (Flaming Gorge)	161–279	626 (Cr5)	480 (Cr5)	1.6 (Cr6)	0.88 (Cr6)	NA	0.00019 (Cr7)	NA	0.00035 (Cr8)	3.0	NA	0.96 (Cr8)
11	Duchesne River (many dams)	0–14 [b]	216 (Cr9)	102 (Cr9)	1.2 (Cr10)	0.37 (Cr10)	NA	0.0019 (Cr9)	NA	0.052 (Cr10)	1.2	0 (Cr9)	0.71 (Cr9)
12	Duchesne River (many dams)	18–27 [b]	216 (Cr9)	102 (Cr9)	1.2 (Cr10)	0.37 (Cr10)	NA	0.00023 (Cr9)	NA	0.0005 (Cr10)	1.5	0 (Cr9)	0.60 (Cr9)
13	Green River (Flaming Gorge)	465–490	740 (Cr11)	586 (Cr11)	15 (Cr12)	8.0	NA	0.00040	NA	0.032	4.0	0	0.85
14	Green River (Flaming Gorge)	475–509	740	586	15	8.0	NA	0.00020	NA	0.00050	4.0	NA	0.97
15	Colorado River (Glen Canyon)	0–25	2,400	860	57	0.24	0.00036	0.00027	0.00020	0.09	6.5	–4	NA
16	Colorado River (Glen Canyon)	25–120	2,400	860	61	3.5	0.0011	0.0011	0.26	0.26	5.4	0	NA
17	Colorado River (Glen Canyon)	170–180	2,288	820	83	13	0.0022	0.0022	0.26	0.26	6.5	0	NA
18	Colorado River (Hoover)	0–70	2,704	561	145	NA	0.00035	0.00036	0.00020	0.035	3.9	–3	NA
19	Colorado River (Hoover)	70–149	2,704	561	145	3.3	0.00035	0.00028	0.00020	0.00040	3.9	–3	NA
20	Colorado River (Hoover)	149–181	2,704	561	145	6.6	0.00035	0.00027	0.00020	0.00030	3.9	1	NA
21	Colorado River (Hoover)	181–193	2,704	561	145	4.1	0.00013	0.000063	0.00020	0.00030	3.9	0.7	NA
22	Colorado River (Parker)	0–45	2,248	421.9	145	NA	0.00026	0.000095	0.00020	0.0016	2.5	–4	NA
23	Colorado River (Parker)	45–95	2,248	421.9	145	1.3	0.00028	0.00026	0.00020	0.00070	2.5	–3	NA
24	Colorado River (Parker)	95–143	2,248	421.9	145	2.6	0.00029	0.00029	0.00020	0.0045	2.5	–1.5	NA
25	Colorado River (Parker)	143–204	2,248	421.9	145	3.3	0.00027	0.00020	0.00020	0.00045	2.5	0.8	NA

a distance downstream from Rulison, CO
b distance downstream from Randlett, UT
Cr1 VanSteeter and Pitlick (1998)
Cr2 Pitlick and Cress (2000)
Cr3 Pitlick and Cress (2002)
Cr4 Pitlick and others (1999)
Cr5 Grams and Schmidt (2002)
Cr6 Grams and Schmidt (2005)
Cr7 Schmidt (unpub. data)
Cr8 Lyons and others (1992)
Cr9 Gaeuman and others (2005)
Cr10 Gaeuman and others (2003)
Cr11 Allred and Schmidt (1999)
Cr12 Andrews (1986)
Cr13 Topping and others (2003)
Cr14 Grams and others (2007)
Cr15 Topping and others (2000)
Cr16 Schmidt and Graf (1990)
Cr17 Schmidt and others (2004)
Cr18 Borland and Miller (1960)
Cr19 Bureau of Reclamation (1950)
Cr20 Williams and Wolman (1984)

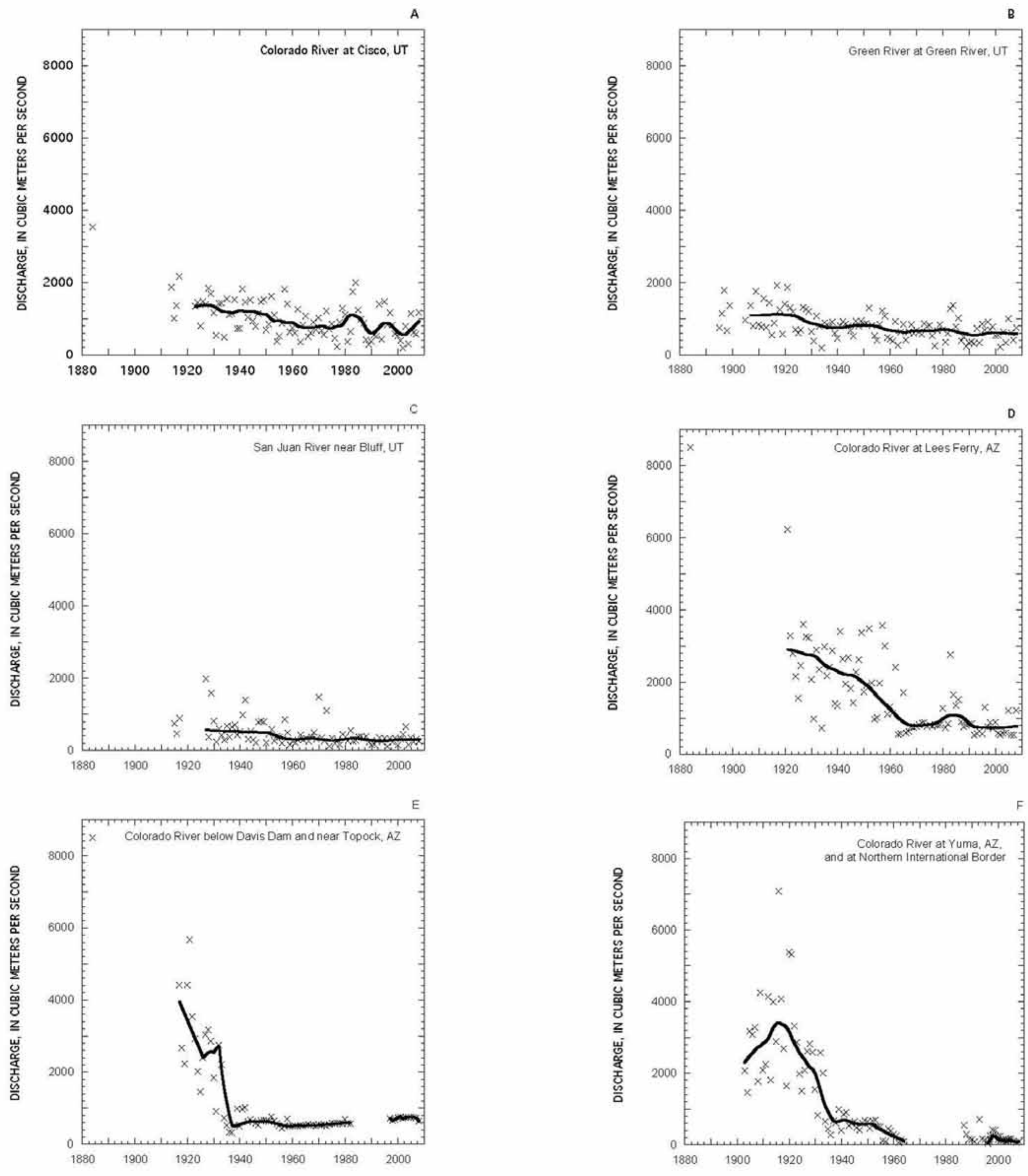


Figure 6. Annual maximum instantaneous discharge for selected streamgaging stations. The dark solid line is a running average calculated on the basis of the nearest 20 percent of the data series. Running average is calculated for the period of sequential years of record and does not include early period of occasional measurements. (A) Colorado River at Cisco, UT (gaging station 09180500). (B) Green River at Green River, UT (gaging station 09315000). (C) San Juan River near Bluff, UT (gaging station 09379500). (D) Colorado River at Lees Ferry, AZ (gaging station 09380000). (E) Combined record of Colorado River below Davis Dam (gaging station 09423000; 1997–2008) and near Topock, AZ (gaging station 09424000; 1884–1982). (F) Combined record of Colorado River at Yuma, AZ (gaging station 09521000; 1903–1964), Colorado River below Yuma Main Canal at Yuma (gaging station 09521100; 1996–2008), and Colorado River at the northern international border near Morelos Dam (gaging station 09522000; 1987–2005).

were. At Lees Ferry, AZ, streamflow is slightly higher in January and mid-summer when demand for hydroelectricity is greater than in other seasons (fig. 7D). Downstream from Hoover and Davis Dams, streamflow is greatest in spring and summer (fig. 7E). A base flow of about 30 m^3/s is released to Mexico, and no semblance of the pre-dam regime is evident (fig. 7F). This base flow is entirely diverted to the Mexicali Valley at Morelos Dam, a run-of-the-river diversion dam with no storage and a diversion capacity of 226 m^3/s. Streamflow in the 100 kilometers (km) downstream from Morelos Dam is intermittent, but some irrigation return flow and municipal effluent maintains perennial flow into the Gulf of California (Cohen and others, 2001). No flow passed Morelos Dam between the 1930s and the early 1970s. Today, in years of high basin runoff and full upstream reservoirs, releases from Hoover Dam sometimes exceed those needed for diversion and are in excess of Morelos' diversion capacity (fig. 4).

The cumulative effects of dams and transbasin diversions in the headwater tributary watersheds have decreased the magnitude of the annual snowmelt flood and increased the magnitude of base flows, but the duration of the annual snowmelt flood has not changed much. Because the large dams and major diversions in the headwater tributaries are located near the exterior rim of the basin, streamflow in the downstream parts of these same rivers reflects the cumulative effects of many reservoirs with different operating rules, different patterns of streamflow withdrawal, and inflow from unregulated tributaries. The cumulative effects of water storage and withdrawal are least on the Green River, as measured at Green River, UT, where typical floods have decreased from about 1,100 m^3/s before 1920 to about 600 m^3/s since 1990 (fig. 6B), and base flows are now typically about 100 m^3/s (fig. 7B). Similar changes in streamflow have been measured near Cisco, UT, on the upper Colorado River, where typical floods have decreased from about 1,400 m^3/s in the early 1920s to about 800 m^3/s today (fig. 6A). Base flows are now about 100 m^3/s (fig. 7A). Typical flood flows of the San Juan River near Bluff peaked at approximately 600 m^3/s in the early 1920s, but were highly variable from year to year (fig. 6C). Today's floods typically peak at about 300 m^3/s (fig. 6C), and base flows are about 30 m^3/s (fig. 7C).

The entire upstream sediment supply is now trapped in reservoirs, and none of the large dams release sediment. Essentially no sediment is delivered to the delta. Suspended sediment loads immediately downstream from each large dam are negligible. Annual sediment loads at Topock decreased from a pre-dam range between 50 x 10^6 and 400 x 10^6 Mg/yr to about 10 x 10^6 Mg/yr after completion of Hoover Dam (Williams and Wolman, 1984). Completion of Glen Canyon Dam caused a decrease of about 99.5 percent in the amount of fine sediment entering Grand Canyon (Topping and others, 2000). Downstream from Glen Canyon Dam, Schmidt (1999), Topping and others (2000), Hazel and others (2006), and Grams and others (2007) showed that there is mass balance deficit at least 170 km downstream from the dam and beyond the influence of the two largest sediment-contributing tributaries—the Paria and Little Colorado Rivers. Mass balance deficit is defined as the condition where less sediment is supplied to the reach than is the mass exported further downstream.

Unregulated tributaries that drain parts of the Colorado Plateau contribute significant amounts of sediment to the upper Colorado, Green, and San Juan Rivers. Sediment inflow from these desert watersheds significantly reduces the magnitude of post-dam sediment mass balance deficit and reduces the length of sediment deficit segments immediately downstream from the Aspinall Unit on the Gunnison River, Flaming Gorge Dam on the Green River, and Navajo Dam on the San Juan River (Schmidt and Wilcock, 2008). Annual sediment load of the upper Colorado River at Cisco, UT, decreased by about 20 percent. Grams and Schmidt (2005) computed a post-dam sediment budget for the 105 km nearest Flaming Gorge Dam and demonstrated that the uncertainties of sediment transport relations are too great to conclude that deficit conditions exist in most of this segment. Further downstream, the mean annual load at Jensen has only decreased by about 50 percent. Andrews (1986) and Allred and Schmidt (1999) showed that the Green River is accumulating sediment near Green River, UT, where the annual load has decreased by 35 to 50 percent.

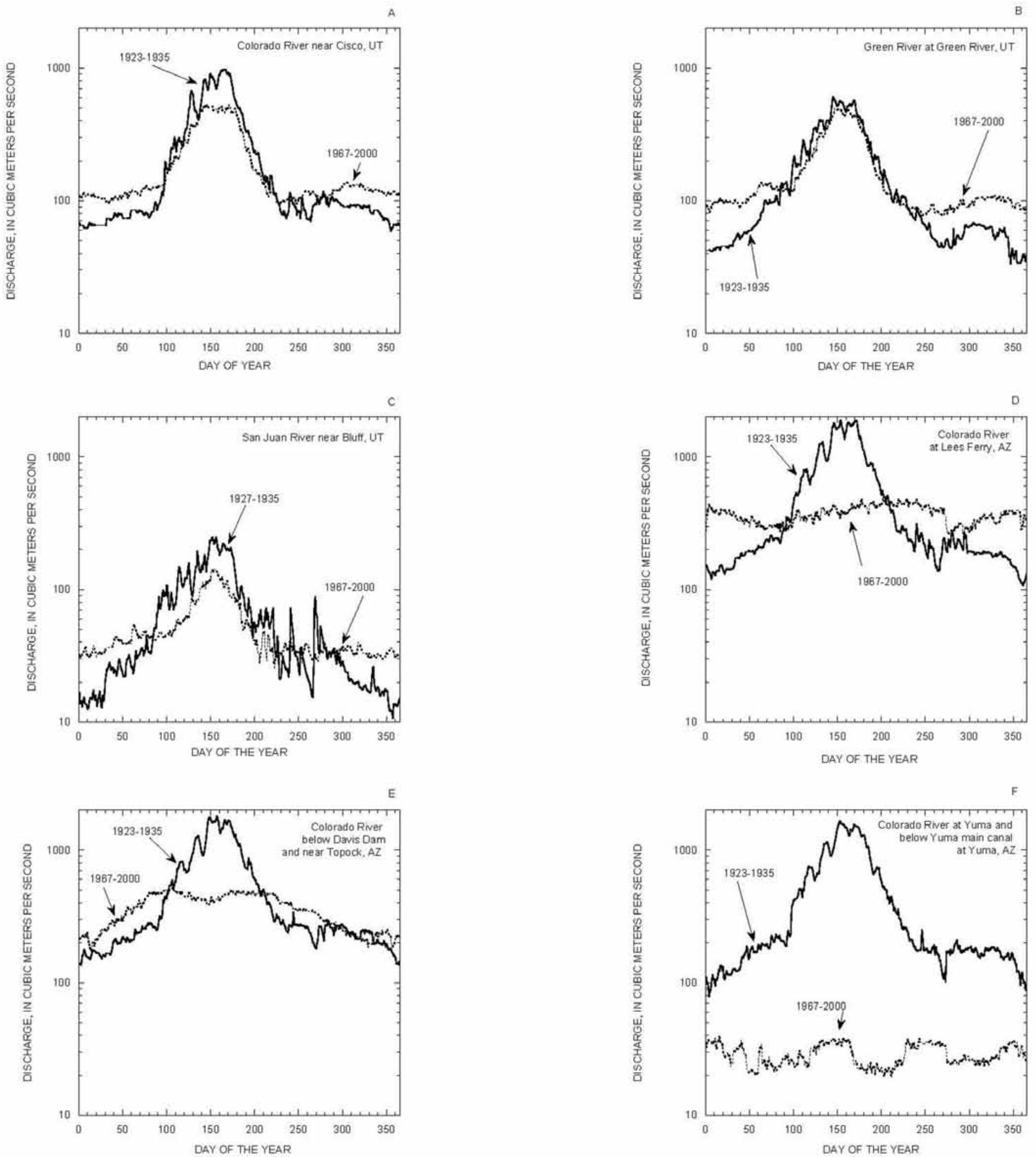


Figure 7. Median hydrographs of mean daily discharge at selected streamgaging stations in the Colorado River Basin for representative pre-dam and post-dam periods. (A) Colorado River at Cisco, UT (gaging station 09180500). (B) Green River at Green River, UT (gaging station 09315000). (C) San Juan River near Bluff, UT (gaging station 09379500). (D) Colorado River at Lees Ferry, AZ (gaging station 09380000). (E) Colorado River below Davis Dam (gaging station 09423000) and near Topock, AZ (gaging station 09424000). (F) Colorado River at Yuma, AZ, (gaging station 09521000) and Colorado River below Yuma Main Canal at Yuma (gaging station 09521100).

Channel Change

The Upper Colorado, Green, and San Juan Rivers

Most of the Green (Andrews, 1986; Lyons and others, 1992; Grams and Schmidt, 2002, 2005) and upper Colorado Rivers (VanSteeter and Pitlick, 1998) has narrowed and simplified (table 2). The Green River is between 10 and 25 percent narrower than it was at the beginning of the 20th century as measured in Browns Park (fig. 8; Grams and Schmidt, 2005), in the canyons of the eastern Uinta Mountains (Grams and Schmidt, 2002), in the Uinta Basin (Lyons and others, 1992), near Green River, UT (Allred and Schmidt, 1999), and further downstream (Graf, 1978). No evidence for bed incision is evident anywhere on the Green River, including immediately downstream from Flaming Gorge Dam (Grams and Schmidt, 2005). Narrowing has also occurred on the Duchesne River (Gaeuman and others, 2003, 2005; Schmidt and others, 2005) and on the upper Colorado River downstream from Rulison, CO (VanSteeter and Pitlick, 1998).

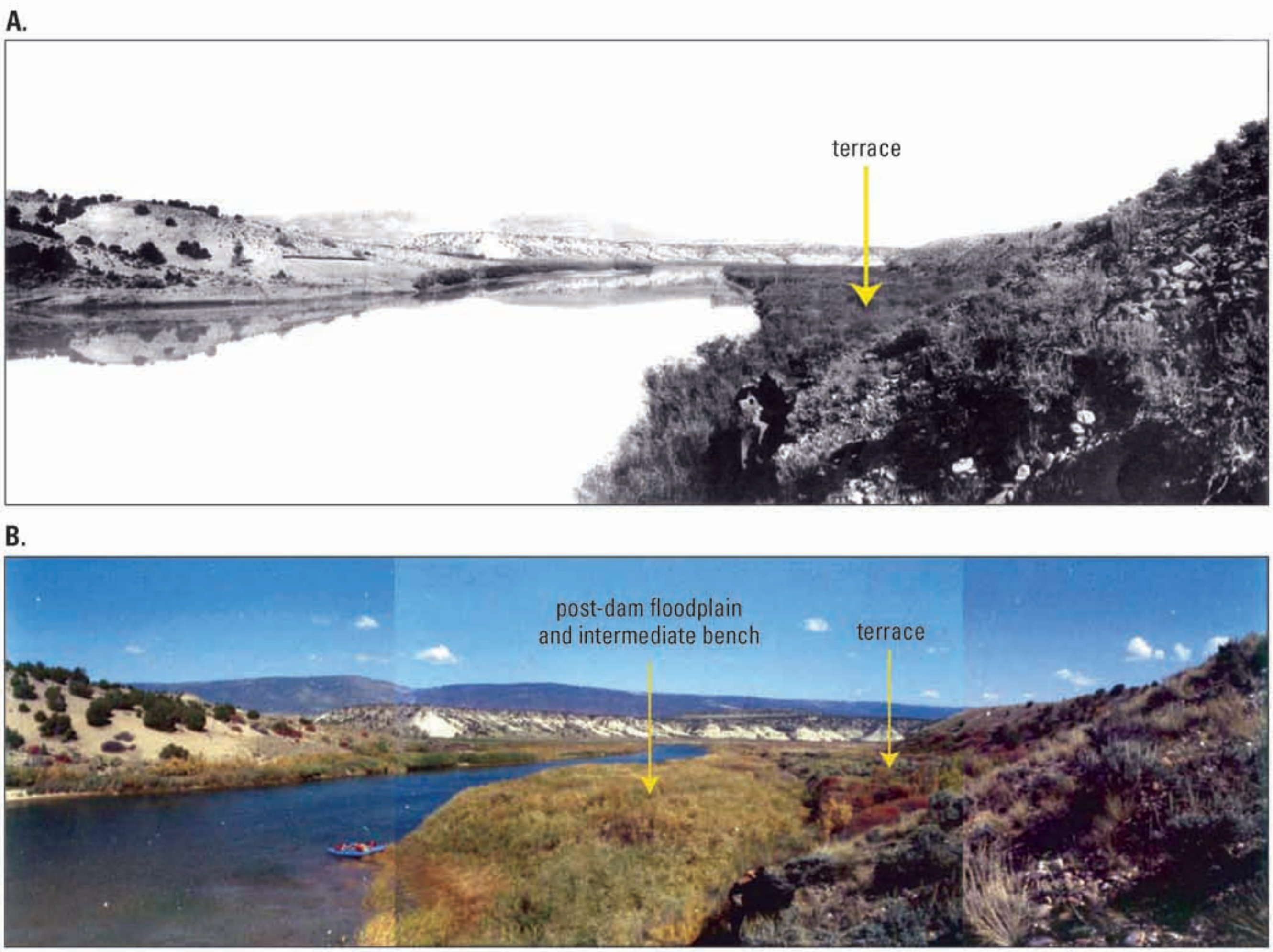


Figure 8. The Green River looking downstream from the location of the lower Bridgeport gage to the lower Bridgeport cableway. The original photograph (A) was taken October 13, 1911, by E.C. La Rue, and the repeat (B) was taken on October 7, 1999.

Glen and Grand Canyons

Large-scale bed incision downstream from Glen Canyon Dam has occurred in the 25 km between the dam and Lees Ferry that had a sand and gravel bed before dam construction. Here, pools were eroded about 6 meters (m) and riffles about 3 m (Grams and others, 2007). Water-surface elevation is now about 2.3 m lower near the dam, and the gradient has decreased about 25 percent (Grams and others, 2007). The present bed is established in what had been the underlying gravels. Today's channel is somewhat narrower and deeper than the pre-dam channel. Bed incision and reduction in flood magnitude caused abandonment of the former flood plain, and this surface is no longer inundated by typical post-dam floods (fig. 9).

Although most of the bed has been stripped of sand in Grand Canyon, there is no evidence of large-scale downward shifts in stage-to-discharge relations, because the bed profile in this debris fan-affected canyon is determined by the elevations of bouldery rapids that occur at the mouths of each steep, ephemeral tributary (Schmidt and others, 2004). Magirl and others (2005) showed that bed elevations of some bouldery rapids have increased since the 1920s. Nevertheless, fine sediment, transported in the mainstem as suspended load, has been removed from recirculation zones. The area of exposed sand in eddy bars was approximately 25 percent smaller in the 1990s than in the pre-dam era, and the thickness of sand irreversibly lost in some recirculation zones exceeds 2 m (Schmidt and others, 2004). Loss of sand from recirculation zones is because of wind deflation and fluvial erosion during post-dam base flows.

The Lower River

Completion of Hoover Dam initiated bed incision that ultimately extended approximately 150 km downstream (Stanley, 1951; Borland and Miller, 1960). Aggradation occurred in the 50 km farther downstream and extended into Lake Havasu reservoir (fig. 10). This longitudinal pattern of near-dam incision and aggradation farther downstream was repeated downstream from Parker Dam; completion of Davis Dam created a new phase of incision and shifted the aggradation reach farther downstream (Borland and Miller, 1960).

The Delta

The delta of the Colorado River once encompassed nearly 8,000 square kilometers (km^2) (Luecke and others, 1999) and was a place of tremendous biodiversity and abundance (Glenn and others, 2001). The distributary channels of the delta created a maze of shifting channels that changed course frequently. Today, the delta's extent is only about 600 km^2. The river is confined within levees for approximately 100 km downstream from Morelos Dam, and the area beyond the levees is mostly irrigated farm fields or cities. Vegetation is dominated by salt cedar (*Tamarix* spp.), but cohorts of native trees were established in the years of surplus runoff.

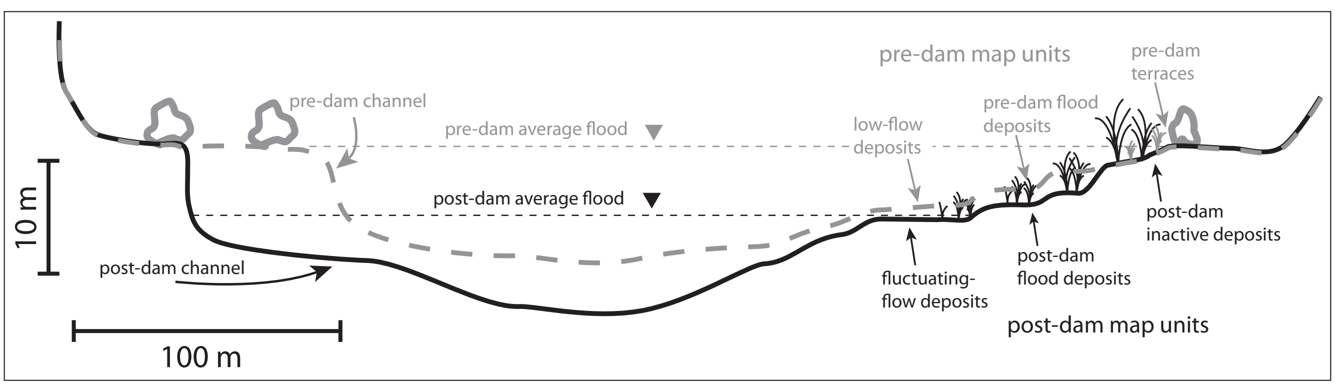
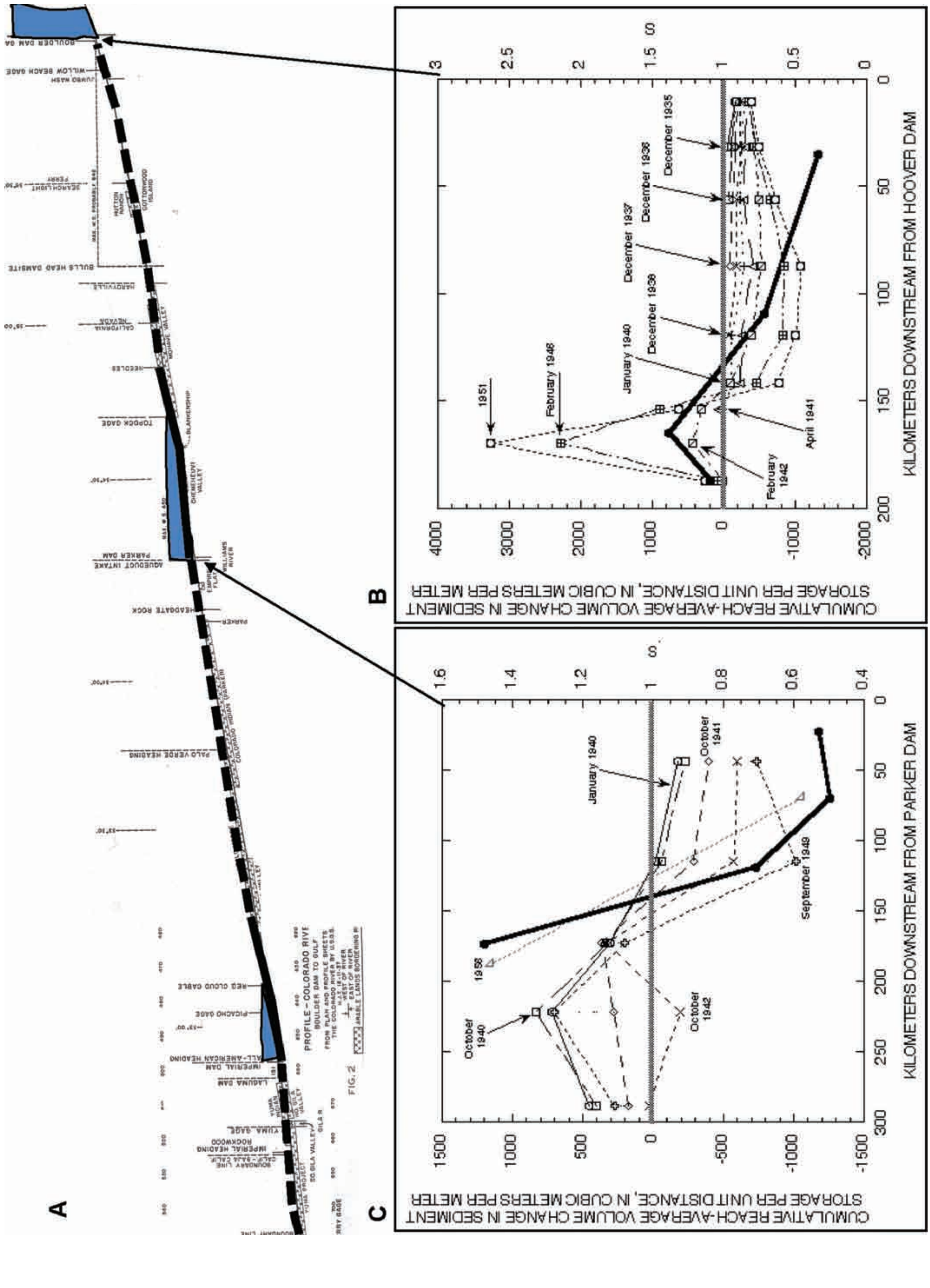


Figure 9. Cross-section changes in the Colorado River in Glen Canyon, downstream from Glen Canyon Dam. The approximate stages of the pre-dam average flood, 2,410 cubic meters per second (m^3/s), and the post-dam average flood, 890 m^3/s, are shown. The stage of the post-dam average flood now reaches elevations typical of the elevation of pre-dam, active channel bars. The combined effects of bed incision and lower typical floods have caused a transformation of the pre-dam riparian communities to upland vegetation communities. Figure adapted from Grams and others (2007).


A
B
C
KILOMETERS DOWNSTREAM FROM HOOVER DAM
KILOMETERS DOWNSTREAM FROM PARKER DAM
CUMULATIVE REACH-AVERAGE VOLUME CHANGE IN SEDIMENT STORAGE PER UNIT DISTANCE, IN CUBIC METERS PER METER
PROFILE - COLORADO RIVER
BOULDER DAM TO GULF
FIG. 2

Regional Comparison of the Magnitude of Perturbation of Sediment Mass Balance

Approach

Alterations of the flow regime and reduction in the sediment supply of many segments have caused imbalances between the sediment transport capacity and sediment supply, thereby causing some channels to evacuate sediment and others to accumulate sediment. The processes of sediment evacuation and accumulation in turn cause changes in aquatic and riparian ecosystems. Although sediment evacuation can manifest itself by a wide variety of changes in channel form, such as bed coarsening and pool scour, large-scale bed incision presents a particularly difficult challenge to river rehabilitation, because the frequency of flood-plain inundation is fundamentally changed (fig. 9). Another fundamental attribute of channel change that has significant impact on river rehabilitation strategies are changes in channel area and width. These changes typically scale with changes in the flood regime (Leopold and Maddock, 1953), and generally it is expected that channels get narrower if the magnitude of floods decreases. Andrews (1986) used this insight to anticipate channel narrowing on the Green River downstream from Flaming Gorge Dam. These three perturbations of geomorphic process or form—changes in sediment mass balance, large-scale changes in bed elevation, and changes in channel width—are the primary geomorphic causes of changes in aquatic and riparian habitat.

Changes in habitat are only one cause of species decline of the mainstem native fishery, and other factors include changes in streamflow temperature, fragmentation of the river network where dams block fish migration, and competition and predation from nonnative species. Nevertheless, it is instructive to evaluate the magnitude of changes in streamflow and sediment supply that drive habitat change, because changes in streamflow and sediment supply are available tools in large river rehabilitation.

Schmidt and Wilcock (2008) suggested an approach by which the relative magnitude of changes in streamflow and sediment supply in causing sediment deficit or surplus could be quantitatively compared. We summarize their work here as it pertains to the Colorado River system. Prediction of the post-dam mass balance is inspired by the widely cited proportionality of Lane (1955)

$$QsD \propto QS \tag{1}$$

where Qs is the rate of sediment supply and D is its grain size, Q is water discharge, and S is channel slope. Henderson (1966) developed a simple approximation of this proportionality by combining equations for momentum, continuity, flow resistance, and transport rate, leading to

$$S^* = \frac{(Qs^*)^{0.5}(D^*)^{0.75}}{Q^*} \tag{2}$$

where

$$S^* = \frac{S_{post}}{S_{pre}} \quad Qs^* = \frac{Qs_{post}}{Qs_{pre}} \quad Q^* = \frac{Q_{post}}{Q_{pre}} \quad D^* = \frac{D_{post}}{D_{pre}} \tag{3}$$

and where the subscripts $_{pre}$ and $_{post}$ indicate conditions before and after the dam. Schmidt and Wilcock (2008) used the 2-year recurrence flood as the index value of Q, and Qs was taken as the annual sediment load.

Values of S^* indicate the potential for sediment evacuation or accumulation in response to changes in flow and sediment supply. Values of $S^* > 1$ indicate that an increase in slope is needed to transport the post-dam sediment supply with the specified flow. Thus, post-dam sediment supply is too great for the post-dam streamflow regime and pre-dam slope; sediment accumulation, therefore, is predicted. Values of $S^* < 1$ indicate that the pre-dam slope is larger than needed to transport the post-dam sediment supply with the post-dam streamflow regime, and sediment evacuation is predicted to

Figure 10. (facing page) (A) Cumulative degradation or aggradation between completion of (B) Hoover Dam in 1934 and (C) Parker Dam in 1937 and indicated time, as well as S* for the same reaches. Field data are shown in thin lines for different time periods computed from Bureau of Reclamation (1950), Stanley (1951), and Borland and Miller (1960), and the thick line is S*.

occur until such point that the initial slope is reduced. Values of S^* do not predict the time domain over which adjustment to post-dam conditions occurs, but values of S^* do predict the nature of the initial perturbation to the downstream geomorphic system caused by each dam. Thus, where bed incision does not occur because the bed material is very large, a regulated river perpetually remains in sediment deficit. In cases where bed incision occurs, however, the post-dam slope might decrease sufficiently to reduce the magnitude of the post-dam sediment deficit.

This approach, however, only provides a reconnaissance level tool with which to compare the relative magnitude of sediment deficit or surplus in a watershed. The derivation of equation 2 depends on a simplified sediment transport relation applicable to sand and fine-gravel bed streams, the assumption of a simplified channel cross section, and assumptions about the relation between the size of the sediment supply and the bed material (Schmidt and Wilcock, 2008). Nevertheless, Schmidt and Wilcock (2008) showed that there is good agreement between the locations of degradation or aggradation measured in the field and the calculated values of S^*, such as on the lower Colorado River (fig. 10). There is also good agreement between S^* and predicted deficit conditions within 100 km downstream from Glen Canyon Dam (Topping and others, 2000; Schmidt and others, 2004; Hazel and others, 2006) and for the Green River downstream from Flaming Gorge Dam (Grams and Schmidt, 2005).

Schmidt and Wilcock (2008) suggested that the potential for large-scale bed incision can be described by a Shields number, τ^*

$$\tau^* \propto \frac{h_{post} S_{pre}}{D_B} \qquad (4)$$

where h_{post} is the mean depth of post-dam floods, S_{pre} is the slope of the channel at the time of dam completion, and D_B is a characteristic bed grain size at the time of dam completion. Schmidt and Wilcock (2008) found that significant incision occurs where $\tau^* > 0.1$ and where $S^* < 1$. Insignificant incision has occurred where $\tau^* < 0.1$. The magnitude of bed incision for large values of τ^* is highly variable, because of differences in substrate, time since dam completion, and magnitude of dam releases (Williams and Wolman, 1984). Schmidt and Wilcock (2008) found no consistent trend between channel narrowing and Q^*, although extreme narrowing to less than 60 percent of the pre-dam width has been observed where $Q^* < 0.4$.

Findings

Schmidt and Wilcock (2008) summarized changes in streamflow, sediment supply, and channel form of several large rivers of the Western United States, and they calculated Q^*, Qs^*, D^*, S^*, and τ^* for 25 segments of the Colorado River drainage network (table 3). In all cases, S^* increases in the downstream direction (fig. 11). In four cases (Green River downstream from Flaming Gorge Dam, and Colorado River downstream from Glen Canyon, Hoover, and Parker Dams) where S^* was calculated near and far from the dam, the degree of sediment deficit diminishes greatly and $S^* > 1$ in some cases. No data were available with which to calculate S^* immediately downstream from dams of the upper Colorado River, and the upper Colorado River downstream from Rulison, CO, is predicted to be in sediment surplus.

Comparison of S^*, τ^*, and Q^* demonstrates that the dams and diversions of the Colorado River Basin have caused very different types and magnitudes of perturbations in different parts of the watershed (fig. 12). There are many segments where dams perturbed the sediment mass balance into deficit, but large-scale bed incision has only occurred on a subset of these segments. For example, segments 15 (S15; Glen Canyon) and 16 (S16; Marble Canyon) are in sediment deficit but only S15 has incised its bed. Sediment deficit ($S^* < 1$) exists on some segments where Q^* is small and where τ^* is relatively large, indicating that channel narrowing has occurred under conditions of sediment deficit and bed incision. Elsewhere, Q^* is small where $S^* > 1$. Thus, channel narrowing has occurred under conditions of sediment deficit and sediment surplus.

Table 3. Changes in streamflow, sediment supply, post-dam sediment mass balance, and bed incision potential for selected reaches.

Segment number	River (Dam)	Location of upstream and downstream ends of reach, in kilometers downstream from dam	Ratio of post-dam to pre-dam flood (Q*)	Ratio of post-dam to pre-dam sediment delivery (Qs*)	Ratio of post-dam to pre-dam grain size of sediment supply (D*)	Post-dam sediment mass balance (S*)	Deficit (D), Surplus (S), Indeterminate (I)	Bed incision potential (τ*)
1	Colorado River (many dams)	0–37 [a]	0.71	0.75	1	1.21	I	0.05
2	Colorado River (many dams)	67–90 [a]	0.71	0.75	1	1.21	I	0.05
3	Gunnison River (Aspinal Unit)		0.62	0.81	1	1.44	S	0.04
4	Colorado River (many dams)	91–119 [a]	0.55	0.77	1	1.61	S	0.04
5	Colorado River (many dams)	120–159 [a]	0.55	0.77	1	1.61	S	0.05
6	Green River (Flaming Gorge)	0–18	0.43	0.02	1.1	0.36	D	0.02
7	Green River (Flaming Gorge)	18–76	0.43	0.06	0.85	0.51	D	0.68
8	Green River (Flaming Gorge)	76–104	0.43	0.10	0.82	0.63	I	0.02
9	Green River (Flaming Gorge)	105	0.77	0.56	1.0	0.97	I	0.06
10	Green River (Flaming Gorge)	161–279	0.77	0.56	1.0	0.97	I	0.99
11	Duchesne River (many dams)	0–14 [b]	0.47	0.32	1	1.19	S	0.03
12	Duchesne River (many dams)	18–27 [b]	0.47	0.32	1	1.19	S	0.42
13	Green River (Flaming Gorge)	465–490	0.79	0.52	1	0.91	I	0.03
14	Green River (Flaming Gorge)	475–509	0.79	0.52	1	0.91	I	0.97
15	Colorado River (Glen Canyon)	0–25	0.36	0.00	1	0.18	D	6.99
16	Colorado River (Glen Canyon)	25–120	0.36	0.06	1	0.67	D	0.01
17	Colorado River (Glen Canyon)	170–180	0.36	0.15	1	1.08	I	0.03
18	Colorado River (Hoover)	0–70	0.21	0.01	1	0.34	D	4.18
19	Colorado River (Hoover)	70–149	0.21	0.02	1	0.72	D	4.16
20	Colorado River (Hoover)	149–181	0.21	0.05	1.5	1.39	I	4.17
21	Colorado River (Hoover)	181–193	0.21	0.03	1.5	1.10	I	1.49
22	Colorado River (Parker)	0–45	0.19	0.01	1	0.53	D	1.98
23	Colorado River (Parker)	45–95	0.19	0.01	1	0.50	D	2.11
24	Colorado River (Parker)	95–143	0.19	0.02	1	0.71	D	2.23
25	Colorado River (Parker)	143–204	0.19	0.02	2.3	1.48	I	2.02

[a] Distance downstream from Rulison, CO.

[b] Distance downstream from Randlett, UT.

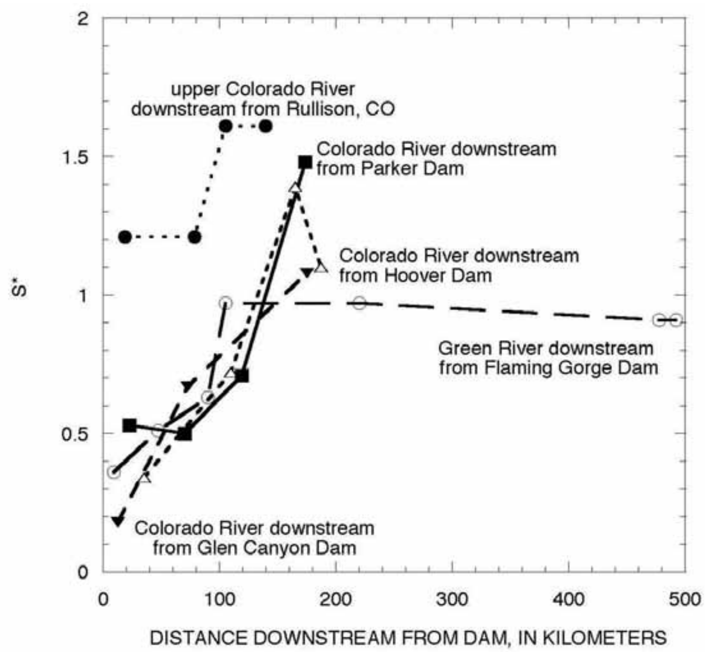


Figure 11. Downstream change in post-dam sediment mass balance (S*). Symbols and lines represent upper Colorado River downstream from Rulison, CO (solid circles and dashed line), Green River downstream from Flaming Gorge Dam (open circles and dashed line), Colorado River downstream from Glen Canyon Dam (solid, downward triangles and dashed line), Colorado River downstream from Hoover Dam (open, upward triangles and dashed line), and Parker Dam (solid square and solid line).

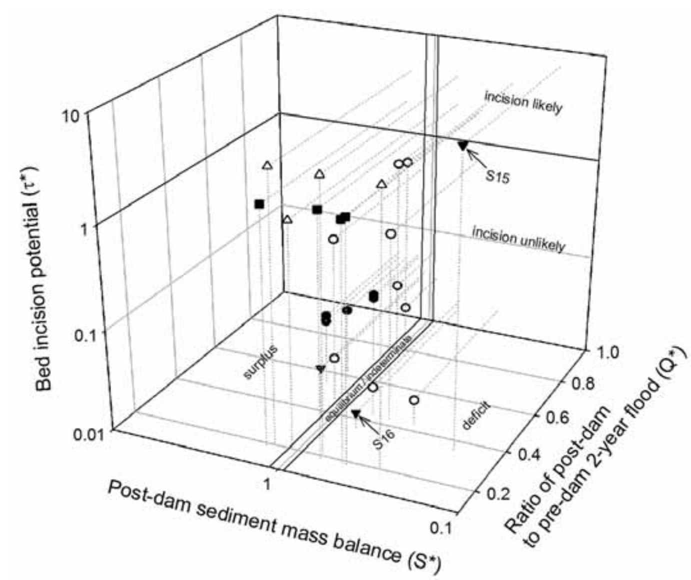


Figure 12. Post-dam sediment mass balance (S*), likelihood of bed incision (τ*), and change in flood regime (Q*) for different segments of the Colorado River system. Dams and diversions have caused a wide array of geomorphic perturbations in the drainage network and pose a range of challenges in river rehabilitation. In some cases, river segments are in severe sediment deficit, but may or may not be subject to large-scale bed incision illustrated by the difference in plotting position of S15 (Glen Canyon) and S16 (Marble Canyon).

Watershed-Scale Appraisal of the Rehabilitation Challenge

The differences in type and magnitude of perturbation illustrated in figure 12 primarily are because some dams are located in the Rocky Mountains and control streamflow but little sediment supply, but other dams are located within the Colorado Plateau and Basin and Range Provinces and control streamflow and sediment supply. Additionally, reservoirs are of different sizes and have different capacities to store flood flows; the sediment trapping efficiency of the large dams in the watershed is nearly 100 percent.

Because the network's channels have been perturbed differently, there is no one prescription concerning how to rehabilitate the entire Colorado River system, nor is it possible to generalize about how difficult rehabilitation is as a general task. Although most river segments have too little sediment for the available streamflow, other segments have too much, and the post-dam sediment mass balance defined in equation 2 provides a reconnaissance basis for assessing the effort of remediating sediment deficit or surplus. Some river segments have incised their beds, and reconnection of channels with flood plains is a significant rehabilitation challenge; elsewhere incision has not occurred. Some river segments have narrowed greatly, and elsewhere this has not occurred.

There is great diversity in river rehabilitation strategies that might be taken, even if only one of the perturbations described above is considered—post-dam sediment mass balance. Figure 13 shows that there is an infinite combination of possibilities by which sediment supply and flood regime could be changed to achieve post-dam sediment mass balance. The two end member approaches are to change only the flow regime or the sediment supply regime. These end member strategies differ in whether they also increase Q^* and thereby shift rivers back toward their early 20th century condition or further decrease Q^* and thereby shift channels toward a miniaturized condition. For example, sediment deficit conditions could be reversed by only increasing the supply of sediment, by only reducing the magnitude of floods released from the dam, or by some combination of both. The strategy of only increasing sediment supply to a river in deficit also shifts the river toward its pre-disturbance, or wild, condition. The strategy that only decreases flood flows to a river in deficit shifts the river toward miniaturized conditions and away from the natural disturbance regime of the river. Because native riverine ecosystems depend on a range of attributes of the natural flow regime (Poff and others, 1997), shifting a river into post-dam sediment balance while also shifting the flow regime toward its pre-disturbance flow and sediment supply regime is more desirable if rehabilitation of native ecosystems is the primary management goal. This is the case with all of the river rehabilitation programs listed in table 1.

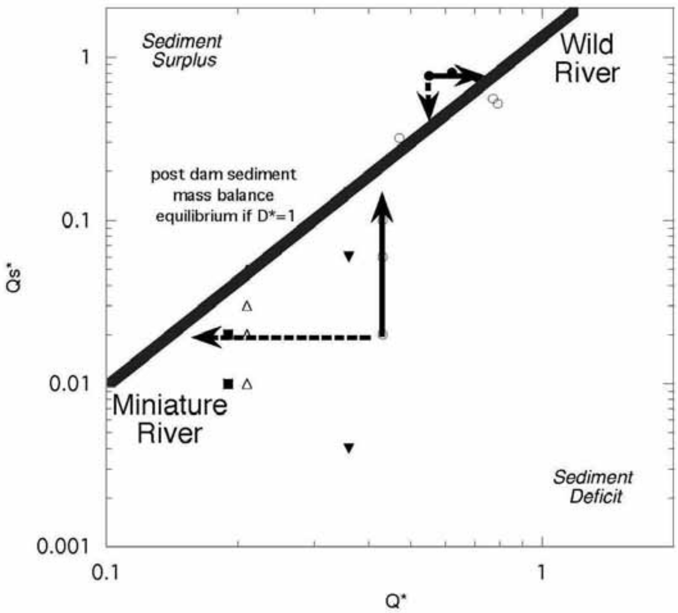


Figure 13. Stability field diagram of changes in flood flow and sediment supply that create sediment deficit or surplus. Each marker indicates a segment of the Colorado River system, and the symbols are the same as those in figures 11 and 12. Arrows indicate possible management actions for two representative segments, one in sediment deficit and one in sediment surplus. Solid arrows indicate change in sediment supply or flood regime that achieves post-dam sediment mass balance and the trajectory of which shifts the river toward pre-disturbance conditions. Dashed arrows indicate change in sediment supply or flood regime and the trajectory of which shifts the river toward further miniaturization.

Schmidt and Wilcock (2008) assessed the relative magnitude of potential rehabilitation actions that would achieve post-dam sediment mass balance and avoid further channel miniaturization by estimating the proportional increase in sediment supply or flood flows necessary to return some of the Colorado River segments to post-dam sediment equilibrium along paths indicated by solid lines as shown in figure 12 (table 4). Estimates were made by adjusting the value of Q^* or Qs^* in equation 2 such that $S^* = 1$.

These results are very imprecise but nevertheless demonstrate that there is a wide range of prescriptions for the Colorado River system if the objective were established to rehabilitate every segment into post-dam sediment mass balance. Some segments require additions of sediment while other segments require an increased flood regime. Many deficit segments require large proportional increases in sediment supply. In most cases, significant infrastructure changes would be necessary to implement these options. For example, application of equation 2 to the Colorado River in Grand Canyon indicates that augmenting the post-dam annual fine sediment supply with an additional 7.9×10^6 Mg/yr is necessary to eliminate deficit conditions, assuming that the post-dam flood regime is not changed. This amount of augmented sediment would only increase Qs^* to 0.13 and in no way can be considered restoration to pre-dam conditions. The required change in Qs^* is small, because the magnitude of post-dam floods has been reduced by approximately 60 percent. This amount of augmented sediment is nevertheless large in terms of engineering design. Randle and others (2007) estimated that augmentation of 4.3×10^6 Mg/yr to the Colorado River in Grand Canyon would cost between \$220 and \$430 x 10^6 in

Table 4. Proportional changes in sediment supply or magnitude of 2-year flood to achieve post-dam equilibrium sediment mass balance.

[km, kilometers]

	Ratio of post-dam to pre-dam sediment supply necessary for equilibrium mass balance conditions, assuming no change in flood regime	Proportional increase in post-dam sediment supply needed to achieve equilibrium mass balance	Ratio of post-dam to pre-dam flood conditions necessary for equilibrium mass balance conditions	Proportional change in post-dam flood flows needed to achieve equilibrium mass balance
Colorado River, 91–119 km downstream from Rulison, CO (reach 4)	0.3	–0.6	0.9	–0.4
Green River, 18–76 km downstream from Flaming Gorge Dam (reach 7)	0.24	2.9	0.2	–0.6
Colorado River, 25–120 km downstream from Glen Canyon Dam (reach 16)	0.13	1.2	0.24	–0.7
Colorado River, 70–149 km downstream from Hoover Dam (reach 19)	0.04	0.9	0.2	–0.6
Colorado River, 45–95 km downstream from Parker Dam (reach 23)	0.04	2.9	0.1	0.6

project capital costs and between \$6.6 and \$17 x 10^6 per year in annual operating costs. In comparison, the 7-day release of a controlled flood from Glen Canyon Dam in 1996 had an economic cost of \$2.5 x 10^6, which was 3.3 percent of the economic value of the hydroelectric power produced in that year (Harpman, 1999).

The Challenge of Rehabilitating the Entire River Network

Each part of the modern Colorado River system can be viewed as existing on a theoretical continuum between that segment's wild, pre-settlement channel form and fluvial processes and another condition where the channel, its streamflow, and its sediment supply are completely altered and transformed (fig. 14). Perhaps the only parts of the Colorado River watershed that are minimally impacted by humans are the small, headwater streams of federally designated wilderness and roadless areas of the Rocky Mountains and some ephemeral drainages of the Colorado Plateau. Parts of the lower Colorado River in the delta, especially downstream from Morelos Dam, may reflect the latter condition of complete alteration. The rest of the drainage network is somewhere between these end member conditions.

Although informed by river science, the decision of how far to attempt to shift present river conditions toward former wild conditions is a matter of public policy. In fact, it is a matter of public policy if such an effort should be attempted at all. A national political consensus does not exist to fully restore the Colorado River system, because such an effort would require decommissioning the large dams and diversions of the watershed and eliminating most hydropower production. Such an effort would change water and power supplies to urban centers in southern California, central and southern Arizona, southern Nevada, central New Mexico, the Colorado Front Range, Utah's Wasatch Front, and to agricultural centers such as the Imperial Valley.

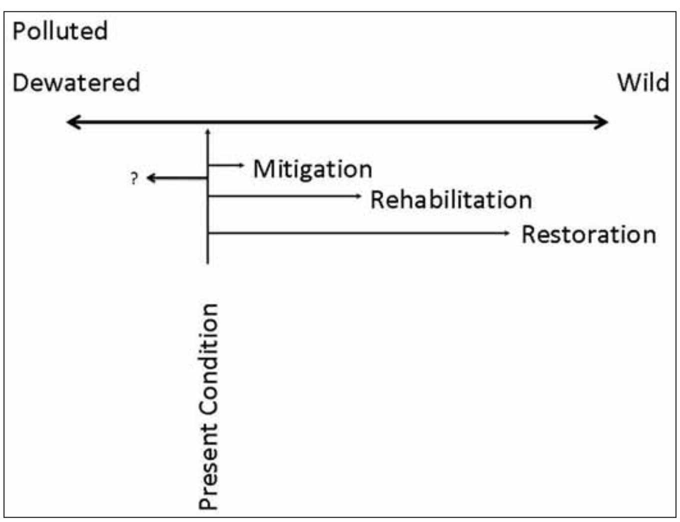


Figure 14. Each river segment in a basin exists on a continuum between its pre-disturbance wild condition and some fully transformed and degraded condition. The choice of how much to return a riverine ecosystem to its former wild condition is a matter of public policy.

Rehabilitation is a goal that improves some attributes of the native ecosystem but does not seek to fully return all aspects of channel form, flow regime, and sediment supply to pre-European conditions. This goal requires specification of which native ecosystem attributes are to be recovered and which attributes of the modern riverscape (e.g., dams, diversions) are not to be changed. A lesser goal for environmental management is mitigation, wherein specific attributes of the riverine ecosystem are targeted for improvement, but a transformed riverine ecosystem is accepted.

The adaptive management and endangered fish recovery programs upstream from Lake Mead (table 1) are rehabilitation programs, because program goals seek to recover fish populations while also assuring delivery of water supplies and hydroelectricity. Mitigation, such as is being pursued as part of the LCR MSCP, is achieved by adjusting streamflow and sediment supply as well as constructing new features of the riverine ecosystem, such as artificial wetlands. Such wetlands were constructed along the Green River in the Browns Park and Ouray National Wildlife Refuges as mitigation for lost wetlands inundated by Flaming Gorge Reservoir in the mid-1960s.

The analysis of perturbations to sediment mass balance and of the effort required to reestablish post-dam sediment mass balance demonstrates that achieving sediment mass balance equilibrium is a daunting, if not impossible, task at a watershed scale. Yet achieving sediment mass balance alone does not address issues related to reversing bed incision, reestablishing flood-plain connection, and channel narrowing. In light of the cumulative costs of rehabilitation and the impact of changing dam operations on water delivery and power supply, it is appropriate to ask, "What environmental management goals should be established for each part of the watershed?" and "Should decisions about goals be made at a segment scale by local stakeholders or at a watershed scale by national and regional interests?"

Many public policy answers to these questions are available. One answer could be to adopt the same management goal for every segment of the river network. Alternatively, each segment might have a different goal that is established by the local stakeholders. Another approach might have a different goal established for each segment based on the principle that each perturbed segment ought to be rehabilitated (1) to the same proportional extent or (2) such that the same proportional effort is expended in each segment in terms of dam reoperations or sediment augmentation. The level of effort in each rehabilitation program also might be established by the political process and reflect other priorities, such as landscape preservation, or solely focus on species populations.

Kondolf and others (2008) have asked similar questions concerning the watershed-scale approach to restoring streams of the Sacramento–San Joaquin River system in California.

Presently, there is no regional process by which the goals of each river rehabilitation program are compared, nor is there consideration of the tradeoffs between rehabilitation effort and magnitude of actual recovery. The analysis presented here indicates that the Grand Canyon segment and the lower river have been perturbed more than the tributaries of the upper basin. More money is now being spent to rehabilitate these river segments downstream from Lees Ferry (table 1), yet the task of rehabilitating the upper basin segments to sediment mass balance equilibrium is probably more tractable and less expensive.

Describing the relation between effort and recovery is one of the greatest challenges of river restoration science, and defining this relation is very difficult. However, defining even an approximation of this relation would further inform the decision of what environmental management goals to establish in each river segment. Two categories of relations can be conceived: one where there is a large degree of environmental improvement for relatively small degrees of initial investment and one where a relatively large degree of investment is required to achieve a significant degree of environmental response (fig. 15). The former category might be considered a politically "easy" path of public policy, because small financial and political compromises are needed to achieve significant environmental improvement. The latter category might be considered a politically "hard" path, because large costs are incurred for relatively small gains. It is probable that most river segments will never be fully restored; thus, those restoration programs that focus on politically "easy" problems might achieve a greater degree of ecosystem recovery.

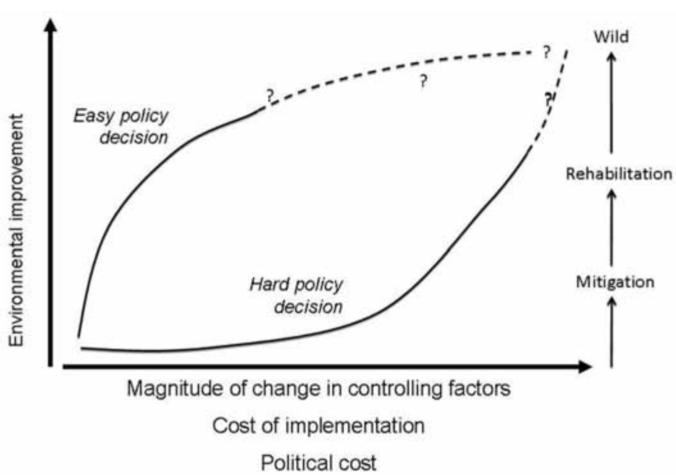


Figure 15. Conceptual graph of hypothetical relations between investment toward river rehabilitation and environmental improvement. Relations where small investments yield large returns are termed "easy" political decisions to adopt in the basin. Large investments with little return are considered "hard" political decisions.

The analysis described here has significant limitations, especially in focusing primarily on sediment mass balance as a metric to reflect a much larger range of ecosystem attributes that would have to be considered if native ecosystem restoration were to be achieved. On the other hand, native ecosystem attributes and processes track well with sediment mass balance in the Colorado River system, where the riverine ecosystems upstream from Lees Ferry generally are less perturbed than those further downstream. The need for a watershed-scale assessment identifying where the greatest return on investment can be gained has also been advocated for the Columbia River system, where Budy and Schaller (2007) showed that restoration of small headwater streams accomplishes much less for recovery of salmon populations than does removal of large dams on the lower Snake River.

Thus, the policy choices affecting the Colorado River watershed are fundamental:

Where should the most effort toward river rehabilitation be undertaken?

Are there parts of the river network where a miniaturized river should be accepted?

Are there parts of the river where even rehabilitation of parts of the native ecosystem ought to be abandoned?

The effort undertaken to date to reverse undesirable conditions of the Colorado River system has been significant, yet the return on investment has been limited in some places. Some parts of the river network are the focus of sophisticated and comprehensive scientific river science, monitoring, and adaptive management, but other parts of the river network receive far less scientific attention. The existence of an international treaty, two interstate compacts, an integrated reservoir management program, and an integrated electricity distribution system suggests that the various river rehabilitation programs also be considered within a watershed context. Some of this work might be accomplished by a basinwide riverine science organization whose focus is the hydrologic, sediment supply, geomorphic, and ecological processes and restoration potential of the entire watershed, rather than the politically defined boundaries of each stakeholder-defined adaptive management program.

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Water Management for River Conservation: Lessons From Outside of the Colorado River Basin for Moving From Sites to Systems

By Christopher P. Konrad[1]

Abstract

Water management at individual dam sites is an important part of river conservation, but its impacts are limited without systemwide coordination of water management and broad integration of resource management across a river basin. Four concepts for basin-scale conservation are illustrated: the benefits of monitoring over large spatial scales even if conservation actions are site specific, coordination of operations of dams in a river system, integration of different types of river management actions, and the potential for conserving biological diversity in parts of the river system. Coordination of operating policies at multiple dams requires flexibility in achieving conservation and other objectives (power generation, flood control, water supply, and recreation) across a river system rather than requiring standardization at all sites. Dam reoperation for conservation is only effective when it is integrated with management of sediment, flood-plain land use, water quality, and invasive species. Basin-scale approaches offer conservation benefits well beyond site-based management in many rivers, but these approaches are complex and require specific enabling conditions. The potential benefits from a basin-scale approach to water management must be assessed relative to constraints and available resources for more coordinated and integrated management activities across the Colorado River Basin.

Introduction

Efforts to conserve freshwater ecosystems and their native species face many challenges in the Colorado River Basin. Management of water resources is central among these challenges because of the essential role of streamflow and groundwater in freshwater ecosystems. A priority for freshwater conservation efforts in the Colorado River Basin, then, is how people can use water sustainably while maintaining sufficient streamflow and groundwater to support diverse communities of native species in rivers, lakes, and springs and on flood plains.

This challenge cannot be answered provincially—by efforts only focused on limiting human impacts and improving ecological conditions locally. There are too many sites degraded by human impacts to try to address each one individually (Richter and others, 1998). Ecological processes and populations of native species depend on connectivity in river networks that cannot be replicated by restoration efforts limited to local efforts to make sites appear to function as they did historically. Instead, long-term solutions are only possible by recognizing that a river is a system that functions in a huge landscape and over the time scales of geology and evolution. Expanding the focus of conservation actions to coordinated and integrated water management across a river basin can create opportunities to eliminate site-specific constraints and align the full complement of ecological conditions needed to achieve biodiversity goals.

This paper illustrates four concepts for basin-scale conservation: the benefits of monitoring over large spatial scales even if conservation actions are site specific, coordination of operations of dams in a river system, integration of different types of river management actions, and the potential for conserving biological diversity in parts of the river system. The examples are drawn from outside of the Colorado River Basin but demonstrate general principles of basin-scale efforts that could be applied to conservation in the Colorado River Basin.

Site-Based Water Management for River Conservation

Conservation of freshwater ecosystems is a national priority as indicated by river restoration efforts in every region of the country and an active role by many Federal and State agencies, non-governmental organizations, and commercial businesses (National Research Council, 1992). Changes in how water is released from dams has been a recent focus for

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many restoration efforts (U.S. Department of the Interior, 1996; U.S. Department of the Interior, 2000; Rood and others, 2005; Alexander and others, 2006; Richter and others, 2006; Warner, 2007; King and others, 2008; Moles and Layzer, 2008; Robinson and Uehlinger, 2008). Although reoperating a dam for ecological objectives cannot address the full range of the dam's impacts (e.g., loss of longitudinal continuity in sediment transport, water-quality changes, fish migration barriers) much less the impacts of other human activities on river systems, it is a tractable immediate-term strategy for addressing one of the most pervasive changes to rivers (Vörösmarty and others, 2004; Nilsson and others, 2005). The efficacy of reoperating a dam for freshwater conservation depends on the extent to which hydrologic alteration is the principal cause of degradation and limiting factor for recovery of a river ecosystem (e.g., Bednarek and Hart, 2005).

What's Missing in Site-Based Conservation?

Despite the need for efforts to protect and improve conditions by changing operations of individual dams, a site-based approach is not adequate to address many of the challenges in conserving river ecosystems. Foremost, the outcomes from changing operations at one dam extend only for a limited distance downstream (and, potentially, not at all upstream) that may not be significant for conserving biodiversity from an ecosystem and evolutionary perspective, except for situations where a reach presents a specific ecological bottleneck (e.g., a migration barrier) or a specific ecological benefit (e.g., a refugia during extreme high or low flows). Site-specific efforts may be unable to address "far field" controls on ecological processes, including routing of sediment from hillslopes through a river network, recruitment and processing of organic material, and meta-population dynamics, including migration, interbreeding, localized extripation, and recolonization. The inadequacy of approaching conservation site by site is even more pronounced where freshwater systems are impacted by pervasive activities, such as agricultural or urban land uses that occur over large regions. Conservation efforts that are focused on dam operations, or more generally the predominant management activity at a site, lack the ability to develop solutions from coordinated management of sites in a larger system and from integrated types of different management actions for river conservation. Thus, efforts focused on changing operations of a dam may not be able to address the variety of threats to rivers or eliminate constraints on potential solutions because of the incongruence in scales.

Moving to a Basin-Scale Perspective

Recent examples are available of the real or potential benefits from moving to a basin-scale perspective on freshwater conservation. Progress can begin simply when scientists coordinate monitoring and interpret the impacts of changing dam operations (or other conservation actions) in a regional context. Coordinated operations of a system of dams can improve ecological outcomes while maintaining or expanding the services provided by those dams by focusing conservation actions (reoperation, migration, removal) on dams with environmental impacts disproportionate to their benefits to human welfare and expanding the social functions (power, water supply, flood control, recreation) from other dams in the system. Finally, integrated river basin management can create conservation opportunities by combining different types of actions, such as increased power generation at a dam that funds downstream flood-plain protection and restoration. The common principles in these examples are coordination of actions across a river system and integration of different types of actions to improve overall management of rivers and conservation of their biodiversity.

Monitoring at a Regional Scale for Interpreting Ecological Effects of Changes in Dam Operations

The Skagit River in Washington is the largest river flowing into Puget Sound and is regionally significant for salmon recovery (Puget Sound Partnership, 2007). The Skagit River has three mainstem hydropower facilities operated by Seattle City Light. Water management at these facilities was revised in 1981 to minimize redd dewatering and fry stranding. Connor and Pflug (2004) documented increases in spawner abundance for Chinook (*Oncorhynchus tshawytscha*), pink (*Oncorhynchus gorbuscha*), and chum (*Oncorhynchus keta*) salmon following implementation of higher incubation flows and a reduction in the number of peaking events and daytime ramping rates. They found that Chinook salmon spawners stabilized but did not continue to increase over time. This result, however, can be interpreted as a success because Chinook spawner abundance generally declined in other unregulated rivers in the Puget Sound Basin. In this case, researchers would not have concluded that streamflow management was effective for Chinook conservation just by looking at the Skagit River below the dams. The broader understanding of the status of Chinook across the region was critical for recognizing that flow management was at least maintaining the status of these fish in the Skagit River while it was declining elsewhere.

Upper Mississippi Long Term Resource Monitoring Program (U.S. Geological Survey (USGS) Upper Midwest Environmental Sciences Center, 2006) provides a model for coordinated regional monitoring. Authorized as part of the U.S. Army Corps of Engineers Environmental Management Program under Water Resources Development Acts of 1986 (Public Law 99–662), 1990 (Public Law 101–640), and 1998 (H.R. 3866 [105th]), the upper Mississippi monitoring program has six field stations that use common methods and shared databases. This coordination provides an opportunity for developing and evaluating more robust and consistent methods, such as the trawl nets used for fish sampling on the upper Mississippi, efficiencies in information systems that should be able to be scaled up to multiple sites, comparative analyses between sites, and a regional perspective on the status of resources.

Coordinating Water Management Across a System and Over Longer Time Scale Creates Opportunities for River Conservation

Water management that is coordinated across a system of reservoirs or other facilities can create opportunities that may not be possible when each facility is operated independently. The Bureau of Reclamation's Yakima River Basin Project in Washington provides an example of water management integrated across a river basin with multiple reservoirs. The Yakima River is used to convey water from reservoirs in the upper basin to agricultural irrigators in the lower basin from spring through early autumn. Spring-run Chinook salmon migrate into the Columbia River in the late spring and move up into the tributaries like the Yakima River during the summer. The Chinook salmon remain in the river until late summer when they spawn. Elevated river stages in August and September from releases for irrigation attract salmon to build redds along the margins of the river. These areas dry out before the salmon fry have emerged from the redds when releases from the reservoirs are reduced at the end of the irrigation season. In response, runoff during the spring and summer is stored in Rimrock reservoir on the Tieton River. Just before spawning begins, releases to the upper Yakima River are dropped, and releases from Rimrock reservoir are increased (fig. 1). Water from the Tieton River maintains supplies for agricultural users, while water levels in the Yakima River can be maintained through the salmon's incubation stage.

The "flip-flop" operation does not make the hydrographs of either river more "natural," but it is an effective solution for supplying the water needed for irrigation in the basin and for salmon incubating in the river. This water-management policy would not be possible without the system of reservoirs available for storing water, the coordinated operation by the Bureau of Reclamation, and an ability to accrue environmental benefits from the joint operations against the environmental costs across the whole system. The policy would not be possible if there were specific and equitable conservation goals for the Yakima and Tieton Rivers. Indeed, the Tieton River ecosystem does not benefit from this operation, but the ecological costs are justified currently by the sustained reproduction of salmon in the upper Yakima River.

Extending this model beyond streamflow management, the Penobscot River Restoration Project in Maine will eliminate three dams on the mainstem of the river in order to facilitate fish passage (Federal Energy Regulatory Commission, 2009). The power generation capacity lost at these dams is offset by increasing hydropower production on tributaries that have less environmental impact and provide fish passage

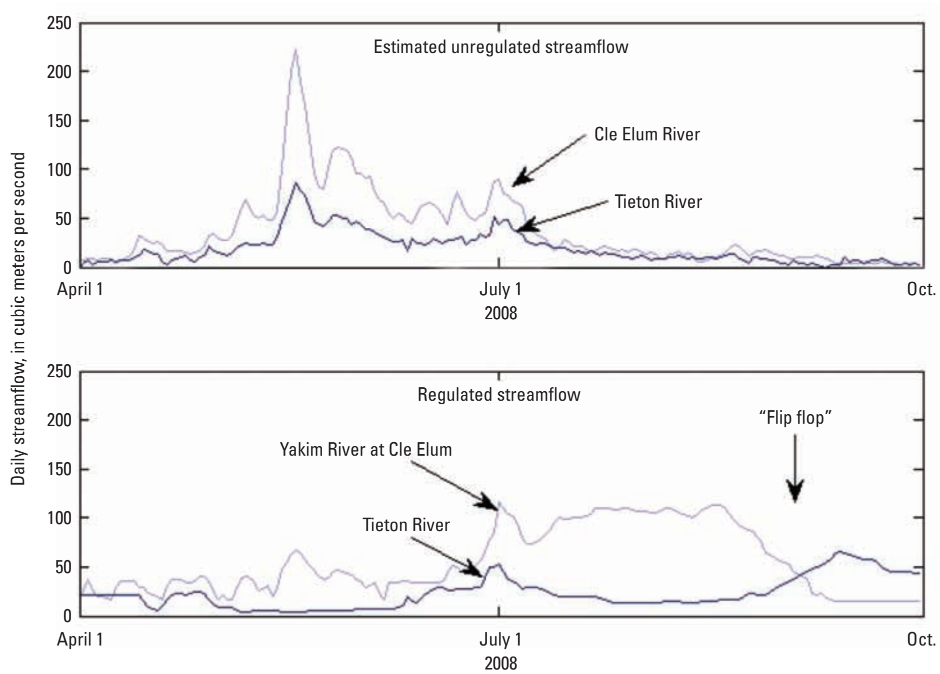

Figure 1. Hydrographs from the Yakima River Basin illustrating the "flip-flop" operation where water supplied by the upper Yakima River reservoirs is replaced by releases from Rimrock reservoir on the Tieton River so that flows in the upper Yakima can be lowered and maintained to keep salmon redds wet during incubation.

on the remaining mainstem dams. In this case as with the Yakima River, the ability to integrate management actions across multiple dams was essential for developing solutions.

The Yakima and Penobscot cases address the disconnect in scales between flow management at a single dam and broader conservation goals that extend to the status of migratory populations or ecological functions, such as routing sediment through a river network. Both involved tailoring different conservation goals for different parts of the river systems and targeting actions accordingly. A site-based approach to flow management can guarantee some minimum acceptable level of ecological condition at any point in a river system, but does not necessarily direct actions—streamflow regime needed for successful salmon reproduction in the case of the upper Yakima and eliminating key fish migration barriers in the case of the Penobscot—to the locations where they will have the greatest ecological benefits. The key to solution in both the Yakima and Penobscot Basins is the tremendous biological potential of parts of the river network that can be realized by focused management actions. For the Yakima, it is a stronghold for salmon spawning in the upper river above most of the agricultural land use in the basin that is not found in other tributaries such as the Tieton River. Similarly for the Penobscot, it is the presence of long, free-flowing river reaches without extensive human impacts above some of the dams in the system. Thus, a systemic view does not imply an inability to resolve differences in management goals for different parts of the system.

Basin-scale strategies for river conservation will only be successful if the ecological benefits accrue across the basin over time, for example, with more resilient core populations and better representation of natural ecological functions. The Truckee River, California, provides an example of coordinating management in time rather than in space for ecological objectives (Rood and others, 2005). Streamflow regulation and diversion led to a decline in flood-plain forests along the lower river in large part because Fremont cottonwood (*Populus fremontii*) seedlings could not become established after germinating. Flow prescriptions were developed, including high-flow pulses to promote Fremont cottonwood recruitment. In low water years when few trees are likely to survive, however, Rood and others (2005) recommend “water should not be directed toward population recruitment but should instead be allocated for the maintenance of riparian plants and other components of the riverine ecosystems” with a more realistic goal of getting good recruitment of riparian trees about once every 10 years. In this case and in the Savannah River, environmental flow prescriptions require the flexibility to change water management year to year, but also depend on coordination of water management over multiple years. Flow prescriptions for the Savannah River call for limits on high flows in years after successful germination of flood-plain trees to allow recruitment of the seedlings to saplings (Richter and others, 2006).

Integrating Dam Operations With Other Forms of River Management in a Basin Can Conserve River Ecosystem and Align Conservation With Human Welfare

Conservation focused on operation of a single dam cannot realize the benefits from integrating different types of actions that are necessary for protecting and restoring ecological functions in river systems. Even the constraints on reoperating a dam for conservation goals will not be surmounted without a broader focus on other actions in a basin that impact a river. Many dams serve flood control purposes and cannot be used to release large floods (by historical standards) because of downstream damage that would result, among other reasons. Conversely, low flows are elevated by dam releases for hydropower in many—though not all—rivers (Magilligan and Nislow, 2005), which downstream users depend on for assimilating wastewater discharges. Without coordination of river management for hydropower, water supply, water quality, and flood risk reduction, water managers may not be able to overcome constraints on implementing environmental flows.

Reoperating dams to create more natural flow patterns may not be effective alone without, for example, appropriate water quality of the releases, sediment for the river to carry, barrier-free fish passage, and connectivity between the river and its flood plain. Combining different types of conservation actions can have synergistic effects, as in the case of regulating the temperature of water released from a dam for environmental flows (Bednarek and Hart, 2005). Although actions aimed at reducing a specific type of human impact on river ecosystems are essential for freshwater conservation, the efficacy of these actions depends on a suite of other actions to address the full range of impacts (e.g., dam operations, diversions, wastewater and stormwater discharges, dredging, levees, flood-plain land uses, introduced species). These other impacts may be difficult to address in the context of site-based conservation that focuses on the impacts of the dominant management action at the site.

The Nature Conservancy has been working in the Yangtze River Basin to coordinate hydropower development with flood-plain management to conserve biological diversity (Harrison and others, 2007). The Jinsha Jiang (upper Yangtze River) flows from the eastern Tibetan plateau carrying runoff from the “rooftop” of the world down to the Sichuan Basin. The Jinsha Jiang has many freshwater ecosystems with significant biodiversity, including the mainstem of the river and a national native fish reserve (Heiner and others, in press). Planned hydropower development along the Jinsha Jiang (Yonghui and others, 2006) threatens these systems. The Nature Conservancy has proposed limiting dam operations for flood control, which requires seasonal drawdown of reservoirs for flood storage, and, instead, has maximized hydropower production by maintaining the “power pool” in the reservoirs at all times and increase the use of flood plains for flood control (Harrison and others, 2007). The dam releases would track inflows leading to more natural flow patterns, and the additional power revenue generated by maintaining the power pool would be used in part for flood-plain conservation.

Enabling Conditions for Basin-Scale River Conservation

Basin-scale river conservation efforts depend on four enabling conditions: multiple dams or other water-management facilities in a river system, flexibility to manage facilities for system benefits, shared conservation goals for river management, and potential to conserve or restore biodiversity. These conditions are closely related and, arguably, not separable. Nonetheless, each is worth considering to assess the viability for basin-scale river conservation in particular basins.

Coordinated management of dams or other facilities across a system is possible when these facilities are fungible to some extent: the services provided by one are interchangeable with those provided by another (e.g., because of interties in the water system or the electrical grid). Operating a group of dams for systemwide goals (e.g., generating hydropower, supplying water, or reducing flood flows) allows for management options that would not be possible when each dam must meet specific goals. Coordinated operations are facilitated when a single agency or utility operates the system for a common purpose as in the Jinsha Jiang, Penobscot River, and Yakima River. Coordinated water management can be difficult, however, even when there is only one principal water manager, such as on the Missouri River (National Research Council, 2002). Basins with multiple water managers face more daunting challenges that begin with the recognition of each other's management goals and extend to equity in achieving management goals. Water management coordinated in time depends on recognition that ecological benefits generated by an action such as a high-flow release have to be maintained in subsequent years or those benefits may be lost (Wright and others, 2008). Coordinated water-management systems can be encouraged by evaluating progress toward ecological goals cumulatively over time rather than incrementally each year and ecosystem function across a basin rather than the ecological conditions at each site.

Basin-scale conservation depends on flexibility to operate individual dams for ecosystem benefits. It may be more effective from a conservation perspective to have a high level of protection for ecological functions (e.g., runoff and streamflow, sediment transport, migration, biogeochemical cycling) from headwaters to mainstem in one part of a river network that supports resilient populations and diverse communities, rather than maintain minimal ecological functions throughout an entire basin with lower biodiversity and less resilience in the populations for critical species. Indeed, basin-scale conservation should not be assessed in terms of abundance (or presence) of species at each dam or other facility in a system, but instead requires integrated measures, such as population (or meta-population) size, total area of habitat in a basin, or ecosystem functions over the river network (sediment routing, nutrient cycling, reproduction and recruitment of juveniles to mature adults in migratory populations). Management flexibility at a site may be ill advised, however, in cases where it could negatively impact population and significantly increase the risk of extirpation or extinction.

Successful conservation at a basin scale requires integrating the range of management activities that affect rivers and flood-plain ecosystems. As with coordinated system operations, integrated river basin management can create solutions to freshwater conservation and water-management issues that would not be possible by only considering one type of management action at an individual site. The administrative challenges of integrating different types of dam operations and flood-plain management loom large in places like the upper Yangtze River, but ultimately surmounting these challenges is necessary to conserve river and flood-plain connectivity. Integrating management in a river basin depends on an alliance of stakeholders with shared ecological goals who are willing to work together rather than trying simply to comply with regulatory requirements applicable to their site.

The starting points for conservation at the scale of a river basin are potential for conserving biodiversity and options for doing so. Many of the examples presented here represent places with high biological diversity and ecosystem integrity. The Sustainable River Project started with the Green River, Kentucky, because of its significant aquatic biodiversity and endemism with 150 fish species and more than 50 mussel species (Silk and Ciruna, 2005; Moles and Layzer, 2008). The Yakima River retains three of its six native stocks of anadromous salmon and is a significant part of the mid-Columbia River evolutionarily significant units for spring-run Chinook salmon and steelhead (*Oncorhynchus mykiss*). The Yangtze River has a native fish reserve downstream from the proposed dams for the Jinsha Jiang and harbors tremendous aquatic biodiversity throughout its upper basin.

The advantages of basin-scale conservation compared to site-specific efforts depend on the availability of options for different spatial arrangements of conservation actions that could achieve conservation goals. These conservation options are analogous to management flexibility at sites: if the condition of every reach in a river network is subjected to the same environmental standards or objectives, there may be little opportunity to realize larger ecological benefits in terms of productivity or biodiversity across the basin rather than at each site. Alternatively, if there are options of achieving conservation objectives, there may be an opportunity to align conservation with other water-management objectives to promote basinwide improvement in the resiliency of species and ecosystem function.

Prospects for Freshwater Conservation at the Scale of the Colorado River Basin

Moving conservation actions to a basin scale will not be simple in the Colorado River Basin. The enabling conditions for basin-scale conservation are only pre-requisite for further action. Actions themselves will be difficult to plan, will be controversial, and may take a long time to implement. In the short term, scientists can use the results of monitoring and research in different parts of the Colorado River Basin to inform site-specific management. In this way, basin-scale conservation can begin with greater coordination of monitoring methods and sampling locations, collaboration on research questions, and shared information systems. Justification of basin-scale conservation efforts depends on the potential for improving biological strongholds that harbor native species or reestablishing streamflow and water-quality conditions that benefit biota throughout the system. It may be impractical to believe that conservation priorities emerging from a regional perspective on the river basin would be adopted locally, but it is not clear that conservation goals for the operation of single dams or other water-management facilities are a feasible and efficient route to protect ecological functions and viable populations of native species in the Colorado River system. At the very least, a broader perspective on freshwater ecosystems and river management options may be warranted at sites where neither freshwater conservation nor water management currently achieves their goals.

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In-Stream Flow Management: Past, Current, and Future Operation of Upper Colorado River Reservoirs

By Thomas Ryan[1]

Abstract

Operations of reservoirs in the upper Colorado River Basin have been modified largely because of environmental legislation. A major driving influence for reservoir reoperation has been endangered Colorado River fish. The Upper Colorado River Endangered Fish Recovery Program published flow recommendations for the Green River in 2000 and for the Colorado and Gunnison Rivers in 2003. The San Juan River Recovery Implementation Program published flow recommendations for the San Juan River in 1999. Flaming Gorge and Navajo Reservoirs are now being operated to meet authorized project purposes as part of the flow recommendations. An Environmental Impact Statement is currently underway to modify operation of the Aspinall Unit (Blue Mesa, Morrow Point, and Crystal Dams) to help achieve flow recommendations for the Gunnison River and portions of the Colorado River.

The operation of Glen Canyon Dam was modified to address environmental resource concerns with the passage of the 1992 Grand Canyon Protection Act and with the signing of the 1996 Record of Decision. The Glen Canyon Adaptive Management Program, which includes the Adaptive Management Work Group (a Federal Advisory Committee), has been in place since 1997 and makes recommendations to the Secretary of the Interior on the operation of Glen Canyon Dam for resource protection and impact mitigation below the dam. Future modifications to the operation of upper Colorado River reservoirs for environmental resources are foreseeable as new scientific information becomes available and as ecosystems and climate change.

Introduction

Reservoir operations in the upper Colorado River Basin have been modified largely because of environmental legislation. Four endangered fish species are native to the upper Colorado River Basin: (1) Colorado pikeminnow *(Ptychocheilus lucius)*, (2) humpback chub *(Gila cypha)*, (3) razorback sucker *(Xyrauchen texanus)*, and (4) bonytail *(Gila elegans*). The Endangered Species Act of 1973 (Public Law 93–205) has resulted in significant modifications to reservoir operations in the basin, and the 1992 Grand Canyon Protection Act (title XVIII of Public Law 102–575) has required modification of operations at Glen Canyon Dam for protection of downstream environmental and cultural resources. Flow recommendations to enhance recovery of endangered fish are described for segments of the Colorado River below major upper Colorado River Basin facilities—Flaming Gorge Dam on the Green River, the Aspinall Unit (Blue Mesa, Morrow Point, and Crystal Dams) on the Gunnison River, and Navajo Dam on the San Juan River.

Flow Recommendations

The Upper Colorado River Endangered Fish Recovery Program and the San Juan River Recovery Implementation Program have conducted extensive research to track population status and trends, threats, and habitats of endangered fish. The Upper Colorado River Endangered Fish Recovery Program published flow recommendations for the Green River in 2000 (Muth and others, 2000) and flow recommendations for the Colorado and Gunnison Rivers in 2003 (McAda, 2003). The San Juan Recovery Implementation Program published flow recommendations for the San Juan River in 1999 (Holden, 1999). These flow recommendations were developed by using a synthesis of research conducted over many years to determine habitat, flow, and temperature requirements likely necessary to achieve recovery of endangered fish. These flow recommendations are for river segments below major Federal dams: (1) Flaming Gorge Dam on the Green River, (2) the Aspinall Unit on the Gunnison River, and (3) Navajo Dam on the San Juan River.

A common element in all three sets of flow recommendations is that flows more closely mimic a natural hydrograph. River regulation by Flaming Gorge, Blue Mesa, and Navajo Dams reduces spring peak flows from pre-dam levels, while elevating base flows from those observed before the closure of the dams. Water temperatures for regulated rivers are much

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cooler than that of unregulated systems. While none of the flow recommendations advocate a complete return to a natural hydrograph, a shift in flows is proposed in all three sets of flow recommendations. Consequently, the flow recommendations reflect more water being released in the spring and less being released in the base-flow period when compared to reservoir operation practices in place at the time the flow recommendations were published.

Flaming Gorge and Navajo Dams are now being operated to meet authorized project purposes and the flow recommendations. For Flaming Gorge Dam, an Environmental Impact Statement (EIS) was completed in November 2005 (Bureau of Reclamation, 2005), a Biological Opinion was completed in August 2005 (U.S. Fish and Wildlife Service, 2005), and a Record of Decision (ROD) was signed in February 2006 (Bureau of Reclamation, 2006c). For Navajo Reservoir, an EIS was completed in April 2006 (Bureau of Reclamation, 2006a), and a ROD was signed in July 2006 (Bureau of Reclamation, 2006b). An EIS is currently underway to modify the operation of the Aspinall Unit to help achieve flow recommendations for the Gunnison River and portions of the Colorado River. A draft EIS on Aspinall Unit operations was published in February 2009 (Bureau of Reclamation, 2009), and a Programmatic Biological Assessment was submitted to the U.S. Fish and Wildlife Service in January 2009 (Bureau of Reclamation, 2008).

Operations to Achieve Spring Flow Recommendations in the Green River Downstream from Flaming Gorge Dam: A Case Study

"Flow and Temperature Recommendations for Endangered Fishes in the Green River Downstream of Flaming Gorge Dam" (Green River flow recommendations) was published in 2000 by the Upper Colorado River Endangered Fish Recovery Program (Muth and others, 2000). The Green River flow recommendations divide the Green River into three reaches, delimited by tributaries. Reach 1 extends from Flaming Gorge Dam to the confluence of the Yampa River. Reach 2 extends from the Yampa River confluence to the confluence of the White River. Reach 3 extends from the White River confluence to the confluence of the Colorado River.

Reach 1 has only minor tributary inflow with flow almost completely dominated by releases from Flaming Gorge Dam. Flows in Reach 2, however, are composed of a combination of releases from Flaming Gorge and the flow of the Yampa River. Reach 2 supports Colorado pikeminnow and a riverine population of razorback suckers. Reach 2 can be viewed as a two-headwater system; almost half of the natural flow in Reach 2 originates in the Yampa River Basin. The flow of the Yampa River is largely unregulated with high spring peak flows observed in all but the driest of years. Reach 3 is important for the reproduction and recruitment of humpback chub in Desolation Canyon and Colorado pikeminnow and razorback sucker below that point. The flow recommendations for Reach 2 and Reach 3 require releases from Flaming Gorge Dam to be coordinated with flows on the Yampa River.

In the spring, high releases from Flaming Gorge Dam are implemented with the occurrence of peak and post-peak flows on the Yampa River. The magnitude and duration of these flows are tied to the hydrologic conditions (percentiles of expected runoff) in the Green and Yampa Rivers. Generally, the wetter the hydrologic conditions, the higher the spring flow and the duration of the peak flow. Specific spring peak target flows for all three reaches are described in the Green River flow recommendations. The goals of the flow recommendations are to create and maintain in-channel habitats and inundate flood-plain habitats believed to be important for recruitment of endangered fish. While achieving spring flow targets in all three reaches is important, Reach 2 generally is regarded as the most important endangered fish habitat of the three.

Flaming Gorge Dam has been operated for the past 3 years in accordance with the ROD. In 2006, the Flaming Gorge Technical Working Group (FGTWG) was established to provide annual proposals to the Bureau of Reclamation on what flow regimes would best achieve ROD objectives on the basis of current year hydrologic conditions and the conditions of the endangered fish. The FGTWG is also charged with integrating, to the extent possible, any requests concerning flow recommendations from the Upper Colorado River Endangered Fish Recovery Program into the proposal so that recovery program research and adaptive management can be facilitated. The FGTWG is represented by technical staff from the U.S. Fish and Wildlife Service, Western Area Power Administration, and Bureau of Reclamation. This group also serves as the informal consultation body for Endangered Species Act compliance as has occurred historically and as directed by the ROD. Public outreach and information exchange occur through the Flaming Gorge Working Group, a public forum which typically meets twice annually.

Since the signing of the 2006 ROD, three different operations at Flaming Gorge have been implemented to achieve spring flow targets. In 2006, based on hydrologic conditions in the Green River Basin with consideration for research requests from the recovery program, an instantaneous peak target flow of 527 cubic meters per second (m^3/s) was targeted and achieved in Reach 2 as measured at the Green River at Jensen, UT, streamgaging station. To achieve this target, bypass releases of approximately 57 m^3/s were added to powerplant capacity releases of 127 m^3/s for a total peak release of 184 m^3/s from Flaming Gorge. This flow combined with the peak flow of the Yampa River achieved the target flow of 527 m^3/s. An instantaneous peak flow of 527 m^3/s in Reach 2 is required in 50 percent of the years under the Green River flow recommendations.

In 2007, drier conditions in the Green River Basin resulted in targeting a lower instantaneous flow at Jensen, UT. An instantaneous peak target flow of 235 m^3/s or greater was targeted and achieved in Reach 2. Powerplant capacity releases from Flaming Gorge Reservoir combined with Yampa River flows resulted in the peak flow in Reach 2 in 2007 being 363 m^3/s. Additionally, a flow duration of 235 m^3/s for 7 days was achieved in Reach 2. The flow recommendations require that this flow duration target be achieved in 90 percent of the years. This flow duration was also achieved in 2006.

In 2008, the hydrologic conditions in the Green River Basin were more favorable with "average" conditions above Flaming Gorge Reservoir and "moderately wet" conditions in the Yampa River Basin. A spring operation was implemented in 2008 to achieve a flow-duration target of 527 m^3/s for 14 days. The flow recommendations require that this flow duration be achieved in 40 percent of the years. Above average spring runoff in the Yampa River combined with powerplant capacity releases from Flaming Gorge Reservoir resulted in achieving the desired flow-duration target. Bypass releases were not required at Flaming Gorge in 2008, although river simulation modeling indicates that bypass releases will be required to achieve this particular target in some years.

During all years under ROD operations to date (2006–2008), temperature objectives as specified in Muth and others (2000) have been achieved through operations of a selective withdrawal system on Flaming Gorge Dam in concert with flow-specific ambient warming rates of the river itself.

Glen Canyon Dam Operations

The operation of Glen Canyon Dam has been influenced by the Endangered Species Act and the 1992 Grand Canyon Protection Act. The Grand Canyon Protection Act required the Secretary of the Interior to prepare an EIS on the long-term operation of Glen Canyon Dam for protection of downstream environmental and cultural resources. An EIS was completed in 1995 (Bureau of Reclamation, 1995), and a ROD was signed in 1996 (U.S. Department of the Interior, 1996; see Campbell and others, this volume, for details).

From the 1960s into the early 1990s, Glen Canyon Dam was operated as a peaking power facility, with releases often varying by over 700 m^3/s within a 24-hour period. The 1996 ROD implemented the modified low fluctuating flow operational alternative. The basis for the Secretary of the Interior's decision in the 1996 ROD was "not to maximize benefits for the most resources, but rather to find an alternative dam operating plan that would permit recovery and long-term sustainability of downstream resources while limiting hydropower capability and flexibility only to the extent necessary to achieve recovery and long-term sustainability." The 1996 ROD set flow parameters concerning minimum and maximum releases from Glen Canyon Dam and limited the rate at which flows could fluctuate.

The Glen Canyon Adaptive Management Program (AMP), which includes the Adaptive Management Work Group (a Federal Advisory Committee), was created by the 1996 ROD. The AMP has been in place since 1997 and makes recommendations to the Secretary of the Interior on the operation of Glen Canyon Dam for resource protection and impact mitigation below the dam. Numerous flow and nonflow activities have been coordinated through the program including high flow, fluctuating flow, and steady flow experiments to support restoration and scientific understanding of the ecosystem in Grand Canyon.

Drought

The Colorado River experienced extreme drought conditions during the 5-year period from 2000 to 2004. While flows were above average in 2005, flows in 2006 and 2007 were below average. The natural flow during the 8-year period from 2000 to 2007 was the lowest 8 consecutive year flow in the 100-year record of the Colorado River. The Colorado River Basin may be in a multidecadal drought. Drought conditions have lowered Lake Powell with current live storage (February 2009) at 54 percent of capacity. Releases from Lake Powell in water years[2] 2001 through 2007 met the minimum objective releases of 10,150 million cubic meters. In 2008, equalization releases were made according to the "Colorado River Interim Guidelines for Lower Basin Shortages and the Coordinated Operations for Lake Powell and Lake Mead" (Department of the Interior, 2007). These guidelines were adopted in December 2007. The total release from Lake Powell in water year 2009 was 11,070 million cubic meters.

Conclusions

Future modifications in the operation of upper Colorado River reservoirs for restoration are foreseeable as new scientific information becomes available, as ecosystems shift, and as the climate changes. Flow recommendations for river systems above Lake Powell were developed on the basis of the best available science. However, it remains to be seen if the desired ecological response (increased recruitment and reduced mortality of endangered fish) can be achieved. Research and monitoring may result in changes or refinements to flow recommendations to achieve the desired response.

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

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In an Era of Changing Climate—Description of Interim Guidelines for Lake Powell and Lake Mead

By Terry Fulp,[1] Carly Jerla,[1] and Russell Callejo[1]

Abstract

Combined, all of the reservoirs on the mainstream of the Colorado River have a total storage capacity of some 60 million acre-feet, approximately four times the river's average annual recorded inflow. During 2000 to 2005, the Colorado River experienced the worst drought in approximately 100 years of recorded history, and that drought continues. Although there have been shortages in Upper Basin tributaries, deliveries in the Lower Basin (downstream from Lees Ferry, Arizona) have been made with 100 percent reliability primarily as a result of the ability to capture water systemwide during high-flow years and to deliver that water during low-flow years.

With the onset and continuation of the current drought, the Bureau of Reclamation's (Reclamation) Upper and Lower Colorado Regions initiated a National Environmental Policy Act (NEPA) process in 2005 to develop Lower Basin shortage guidelines and coordinated management strategies for the operation of Lake Powell and Lake Mead. Following an intensive period of public input and analysis from late 2005 through 2007, the Secretary of the Interior implemented the "Colorado River Interim Guidelines for Lower Basin Shortages and the Coordinated Operations of Lake Powell and Lake Mead" (Interim Guidelines) in December of 2007. The guidelines provide a prescriptive methodology to determine the annual releases from Lake Powell and Lake Mead for an interim period (through 2026). The guidelines focus on encouraging conservation of water in the Lower Basin, considering reservoir operations at all water levels, and gaining valuable experience operating the reservoirs to improve the basis for making future operational decisions during the interim period and (or) thereafter.

In 2004, Reclamation's Lower Colorado Region initiated a research and development program, collaborating with other Federal agencies and universities, for the purpose of enabling the use of new methods for projecting possible future river flows that take into account increased hydrologic variability and potential decreases in the river's annual inflow owing to changing climate. As part of this effort and in conjunction with the development of the new Interim Guidelines, additional analyses were included in the 2007 Final Environmental Impact Statement that considered the impacts of greater hydrologic variability than have been seen in the 100-year record. Reclamation is committed to continuing this research and development program to further its ability to analyze the potential impacts of climate change and to use that information in water and power operations and planning studies to be able to adapt, as appropriate, the operation and management of the river to a changing future climate.

Introduction

The Colorado River is a critical resource in the Western United States; seven Western States and Mexico depend on the Colorado River for water supply, power production, recreation, and environmental resources. The Colorado River Basin (basin) is divided, both politically and physically, into the Upper and Lower Basins at Lees Ferry, Arizona—a result of the Colorado River Compact of 1922 (Compact). The Compact also divided the seven basin States into the Upper Division and the Lower Division States. The Upper Division States includes Colorado, New Mexico, Utah, and Wyoming. Arizona, California, and Nevada make up the Lower Division States (fig. 1).

Climate varies significantly throughout the basin. Most of the basin is arid and semiarid, and generally receives less than 10 inches of precipitation per year. In contrast, many of the mountainous areas that rim the northern portion of the basin receive, on average, over 40 inches of precipitation per year. The annual flow of the Colorado River varies considerably from year to year. As illustrated in figure 2, over the past approximately 100 years (1906 through 2008), the natural flow (estimate of streamflow that would exist without human development) at the Lees Ferry gaging station (located approximately 16 miles downstream from Glen Canyon Dam) has ranged from 5.5 million acre-feet (MAF) to 25.5 MAF, with an average of 15.0 MAF.

Recent tree-ring reconstructions provide a rich view of the magnitude and duration of the natural streamflow

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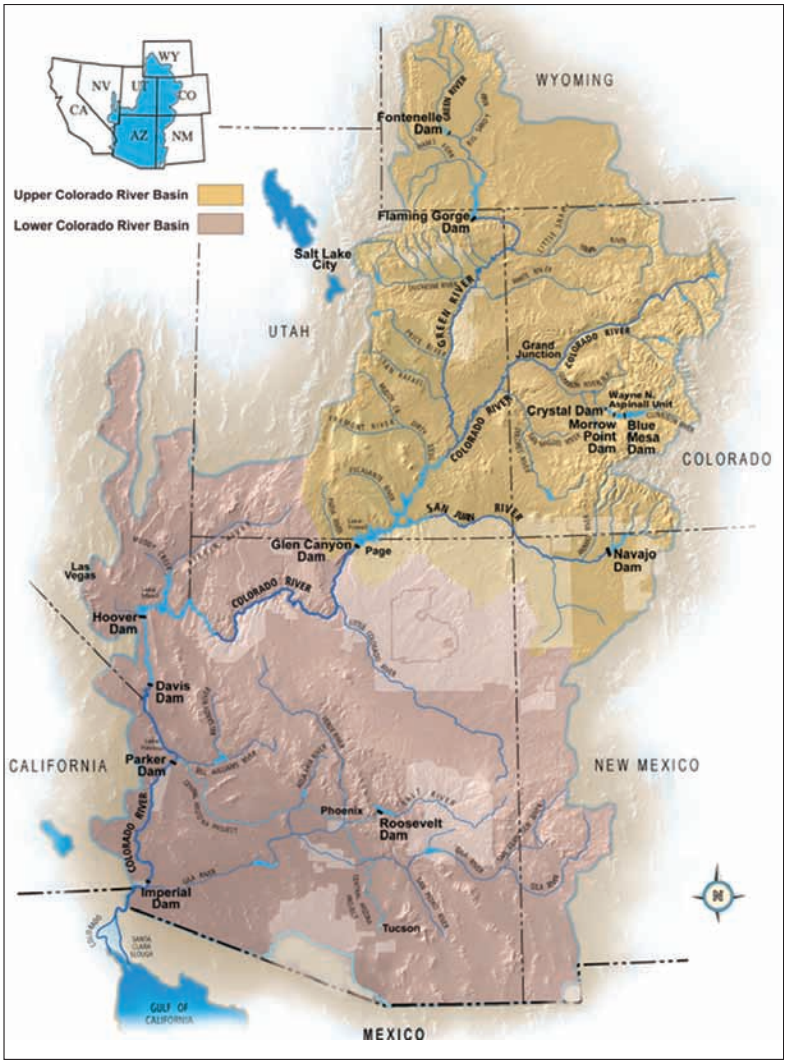


Figure 1. The Colorado River Basin.

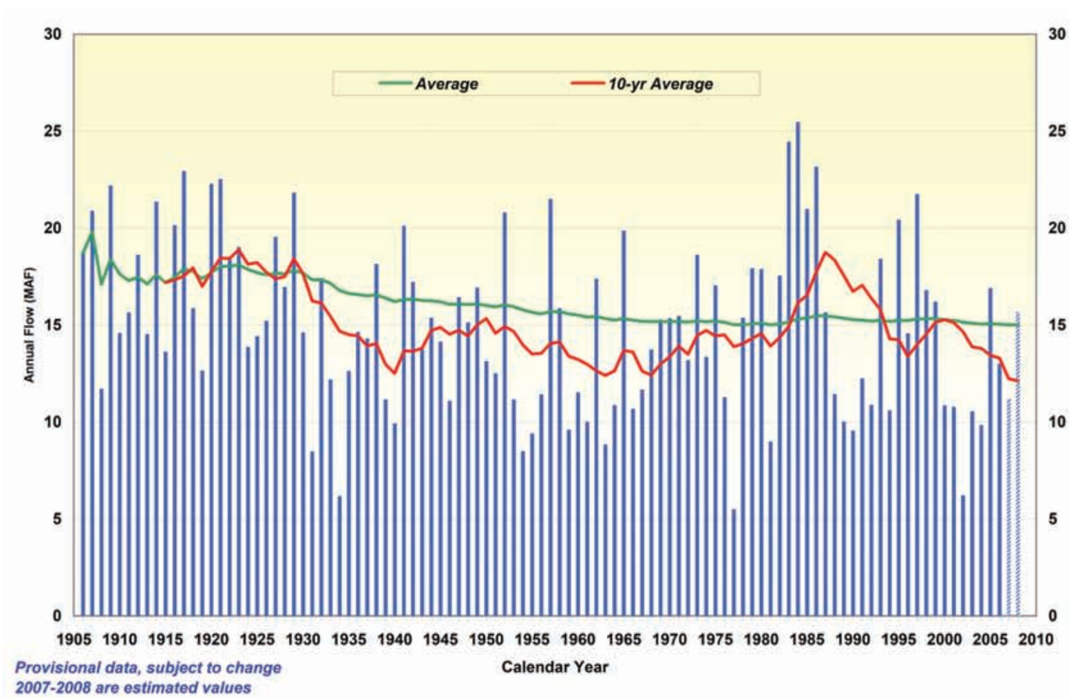


Figure 2. Natural flow of the Colorado River at Lees Ferry, AZ.

variability and indicate that the long-term average may be close to 14.7 MAF (Meko and others, 2007). As shown in figure 3, more severe droughts have occurred in the past 1,200 years, specifically during the 1100s. A severe drought, known as the Medieval Drought (1118–1179), occurred during this time. The Medieval Drought has the lowest 25-year mean of 12.6 MAF in the paleorecord and is characterized by a notable absence of high flows for a 60-year period (Meko and others, 2007).

The Secretary of the Interior (Secretary), acting through the Bureau of Reclamation (Reclamation), is vested with the responsibility to manage the mainstream waters of the Lower Basin of the Colorado River pursuant to applicable Federal law. This responsibility is carried out consistent with a body of documents referred to as the Law of the River, of which the Compact is the underpinning agreement. The Compact apportioned to the Upper Basin and Lower Basin, in perpetuity, the exclusive beneficial consumptive use of 7.5 MAF per year. The Compact also stipulated that the flow in the Colorado River at Lees Ferry not be depleted below 75 MAF for any period of 10 consecutive years. Furthermore, the Upper and Lower Basins agreed in the Compact to share in any deficiency in meeting future water commitments to Mexico, which was allocated 1.5 MAF annually in a 1944 treaty.

The Colorado River system is operated on a tight margin. Apportioned water in the basin totals 16.5 MAF, and the average natural flow of the observed record is 15.0 MAF. The Upper Basin has not fully developed and uses less than its 7.5 MAF apportionment. Consumptive use in the basin has averaged approximately 12.8 MAF over the last 10 years. The Colorado River system, which contains numerous reservoirs, provides an aggregate of approximately 60 MAF of storage, or roughly 4 years of average natural flow of the river. Lake Powell and the downstream Lake Mead provide approximately 85 percent of this storage. Although there have been shortages in Upper Basin tributaries since 2000, all of the requested deliveries were met in the Lower Basin despite having the worst 10-year drought in the last century.

Colorado River Drought: Impetus for the Interim Guidelines

During 2000 to 2005, the Colorado River experienced the worst drought in approximately 100 years of recorded history. This drought reduced Colorado River system storage, while demands for Colorado River water continued to increase. From October 1999 through the end of September 2005, combined storage in Lake Powell and Lake Mead decreased from 47.6 MAF (approximately 95 percent of capacity) to 27.2 MAF (approximately 54 percent of capacity) and was as low as 23.1 MAF (approximately 46 percent of capacity) in 2004. Although a drought of this magnitude is unprecedented in the modern history of the river, tree-ring records show that droughts of this severity have occurred in the past, and climate experts and scientists suggest that such droughts are likely to occur in the future.

In the spring of 2005, declining reservoir levels in the basin led to interstate and interbasin tensions. Specific guidelines to address the operations of Lake Powell and Lake

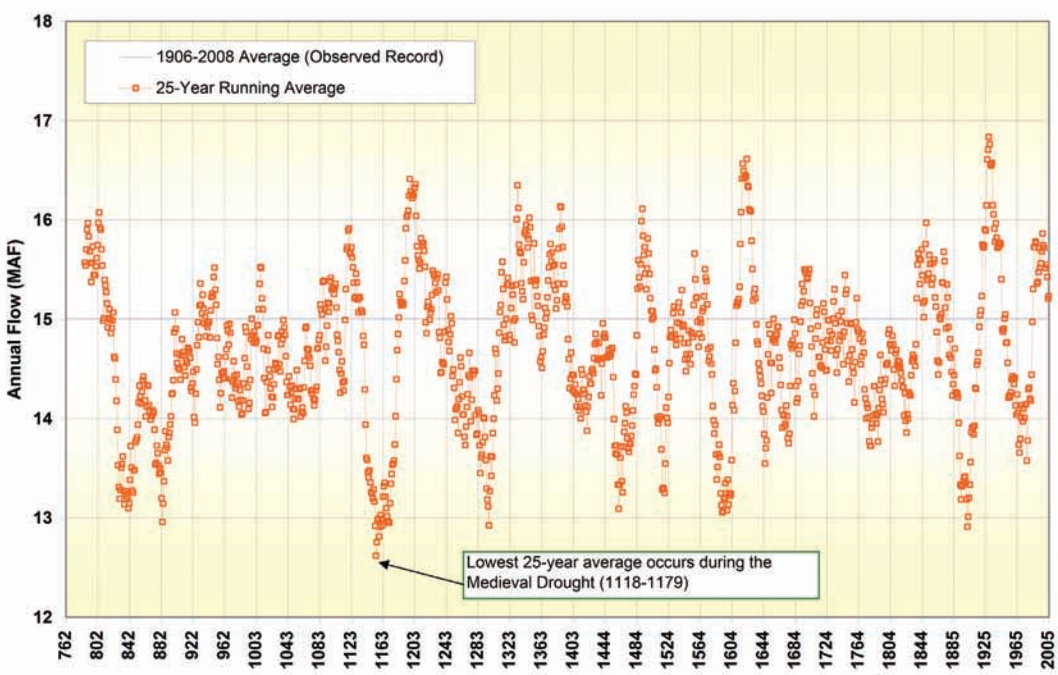


Figure 3. Paleo reconstructed flow of the Colorado River at Lees Ferry, AZ (from Meko and others, 2007).

Mead during drought and low reservoir conditions had not yet been developed, because these types of low-reservoir conditions had simply not been experienced with both reservoirs in place.[2] Storage of water and flows in the Colorado River had been sufficient so that it had not been necessary for the Secretary to reduce deliveries by determining a "shortage" on the lower Colorado River.[3] Without operational guidelines in place, water users in the Lower Basin who rely on Colorado River water were not able to identify particular reservoir conditions under which a shortage would be determined. Nor were these water users able to identify the frequency or magnitude of any potential future annual reductions in their water deliveries.

Operations between Lake Powell and Lake Mead were coordinated only at higher reservoir levels (at a Lake Powell capacity of 61 percent or higher) through an operation known as equalization. Below the equalization level, the Lake Powell release was governed by the minimum objective release of 8.23 MAF, without regard to the condition of the two reservoirs. To minimize shortages in the Lower Basin and avoid the risk of curtailments of Colorado River water use in the Upper Basin, a more coordinated approach to the operations between the reservoirs, for a full range of reservoir conditions, was needed.

These factors, along with the acknowledgment that lower reservoir conditions may occur more frequently because of changing hydrologic conditions and anticipated future demands on Colorado River water supplies, led the U.S. Department of the Interior to conclude that additional management guidelines were necessary and desirable for efficient management of the Colorado River.

The Development of the Interim Guidelines

In May 2005, the Secretary tasked the Upper and Lower Division States (basin States) to develop a consensus plan to mitigate drought in the basin. The Secretary was clear that the U.S. Department of the Interior was committed to developing guidelines with or without the States' consensus. Accordingly, the Secretary directed Reclamation to engage in a process to develop guidelines for Lower Basin shortages and the operation of Lake Powell and Lake Mead, particularly under drought and low reservoir conditions. Later that year, Reclamation announced its intent to initiate a National Environmental Policy Act (NEPA) process to develop such guidelines.

During the scoping phase of the NEPA process, three important considerations were identified: (1) the importance of encouraging conservation of water, particularly during times of drought; (2) the importance of considering reservoir operations at all operational levels, not just when reservoirs are low; and (3), the importance of establishing operational guidelines for an interim period to gain valuable operational experience to inform future management decisions. Out of these three considerations, four key operational elements emerged: (1) shortage strategy for Lake Mead and the Lower Division States, (2) coordinated operation of Lake Powell and Lake Mead, (3) mechanism for the storage and delivery of conserved system and nonsystem water in Lake Mead, and (4) modified and extended elements of existing Interim Surplus Guidelines (ISG). Each element was addressed in the broad range of reasonable alternatives analyzed in the 2007 Final Environmental Impact Statement (Final EIS; Bureau of Reclamation, 2007).

The alternatives were developed in coordination with a diverse body of stakeholders, including the basin States, a consortium of environmental nongovernmental organizations (NGOs), Native American Tribes, Federal agencies, and the general public. The basin States submitted a consensus alternative that signified a historic agreement on issues of this magnitude.

The preferred alternative, based on the basin States' alternative and the "conservation before shortage" alternative submitted by the environmental NGOs, was made up of four key elements, corresponding to those listed previously. First, the preferred alternative proposed discrete levels of shortage volumes associated with Lake Mead elevations to conserve reservoir storage and provide water users and managers in the Lower Basin with greater certainty to know when, and by how much, water deliveries will be reduced during low reservoir conditions. Second, it proposed a fully coordinated operation of Lake Powell and Lake Mead to minimize shortages in the Lower Basin and avoid risk of curtailments of use in the Upper Basin. Third, the preferred alternative proposed an Intentionally Created Surplus (ICS) mechanism to provide for the creation, accounting, and delivery of conserved system and nonsystem water, thereby promoting water conservation in the Lower Basin. Fourth, it extended the term of the ISG and modified those guidelines by eliminating the most liberal surplus conditions, thereby leaving more water in storage to reduce the severity of a future shortage should one occur.

A Record of Decision (ROD; U.S. Department of the Interior, 2007) was issued in December 2007, officially adopting the guidelines (Interim Guidelines). Prescribed operations at Lake Powell and Lake Mead under the Interim Guidelines are described in figure 4.

[2] Lake Mead first filled in 1935; Lake Powell first filled in 1980.

[3] The Secretary annually determines the water-supply condition for the Lower Division States; a "normal" condition is determined when 7.5 MAF of water is available, a "surplus" condition is determined when more than 7.5 MAF of water is available, and a "shortage" condition is determined when less than 7.5 MAF of water is available.

Lake Powell

Elevation (feet)	Operation According to the Interim Guidelines	Live Storage (maf)[1]
3,700	**Equalization Tier** Equalize, avoid spills or release 8.23 maf	24.3
3,636 - 3,666 (2008-2026)	**Upper Elevation Balancing Tier**[4] Release 8.23 maf; if Lake Mead < 1,075 feet, balance contents with a min/max release of 7.0 and 9.0 maf	15.5 - 19.3 (2008-2026)
3,575	**Mid-Elevation Release Tier** Release 7.48 maf; if Lake Mead < 1,025 feet, release 8.23 maf	9.5
3,525	**Lower Elevation Balancing Tier** Balance contents with a min/max release of 7.0 and 9.5 maf	5.9
3,490		4.0
3,370		0

Lake Mead

Elevation (feet)	Operation According to the Interim Guidelines	Live Storage (maf)
1,220	**Flood Control Surplus or Quantified Surplus Condition** Deliver > 7.5 maf (± ICS[2] if Quantified Surplus)	25.9
1,200 (approx.)[3]	**Domestic Surplus or ICS Surplus Condition** Deliver > 7.5 maf ± ICS	22.9 (approx.)
1,145	**Normal or ICS Surplus Condition** Deliver ≥ 7.5 maf ± ICS	15.9
1,105		11.9
1,075	**Shortage Condition** Deliver 7.167[5] maf + DSS[6]	9.4
1,050	**Shortage Condition** Deliver 7.083[7] maf + DSS	7.5
1,025	**Shortage Condition** Deliver 7.0[8] maf + DSS Further measures may be undertaken[9]	5.8
1,000		4.3
895		0

Diagram not to scale

[1] Acronym for million acre-feet.
[2] Acronym for Intentionally Created Surplus. See the 2007 Interim Guidelines.
[3] This elevation, and the corresponding storage value, is approximate. It is determined each year by considering several factors including Lake Powell and Lake Mead storage, projected Upper Basin and Lower Basin demands, and an assumed inflow.
[4] Subject to April adjustment which may result in a release according to the Equalization Tier.
[5] Of which 2.48 maf is apportioned to Arizona, 4.4 maf to California, and 0.287 maf to Nevada.
[6] Acronym for Developed Shortage Supply. See the 2007 Interim Guidelines.
[7] Of which 2.40 maf is apportioned to Arizona, 4.4 maf to California, and 0.283 maf to Nevada.
[8] Of which 2.32 maf is apportioned to Arizona, 4.4 maf to California, and 0.280 maf to Nevada.
[9] Whenever Lake Mead is below elevation 1,025 feet, the Secretary shall consider whether hydrologic conditions together with anticipated deliveries to the Lower Division States and Mexico are likely to cause the elevation at Lake Mead to fall below 1,000 feet. Such consideration, in consultation with the Basin States, may result in the undertaking of further measures, consistent with applicable Federal law.

Figure 4. Operational diagrams for Lake Powell and Lake Mead from the Interim Guidelines.

Efforts to Address Climate Change and Variability in the Development of the Interim Guidelines

In 2004, Reclamation's Lower Colorado Region initiated a research and development program—with a collaboration with other Federal agencies and universities—for the purpose of enabling the use of new methods for projecting possible future river flows that take into account increased hydrologic variability and potential decreases in the river's annual inflow owing to a changing climate. As part of this effort and in conjunction with the development of the Final EIS, a group of leading climate experts (Climate Technical Work Group) was empanelled to assess the state of knowledge regarding climate change in the basin and to prioritize future research and development needs. The findings and recommendations of the work group were published as Appendix U to the 2007 Final Interim Guidelines EIS. Owing to the time horizon of the decision (approximately 20 years) and the lack of precise knowledge of the potential impacts of climate change on the basin, the recommendation of the Climate Technical Work Group was to include additional analyses considering the impacts of greater hydrologic variability than has been seen in the 100-year record. Following this recommendation, a quantitative sensitivity analysis using paleoclimate evidence was included as Appendix N in the 2007 Final Interim Guidelines EIS, accompanied by a qualitative discussion of the potential impacts of climate change.

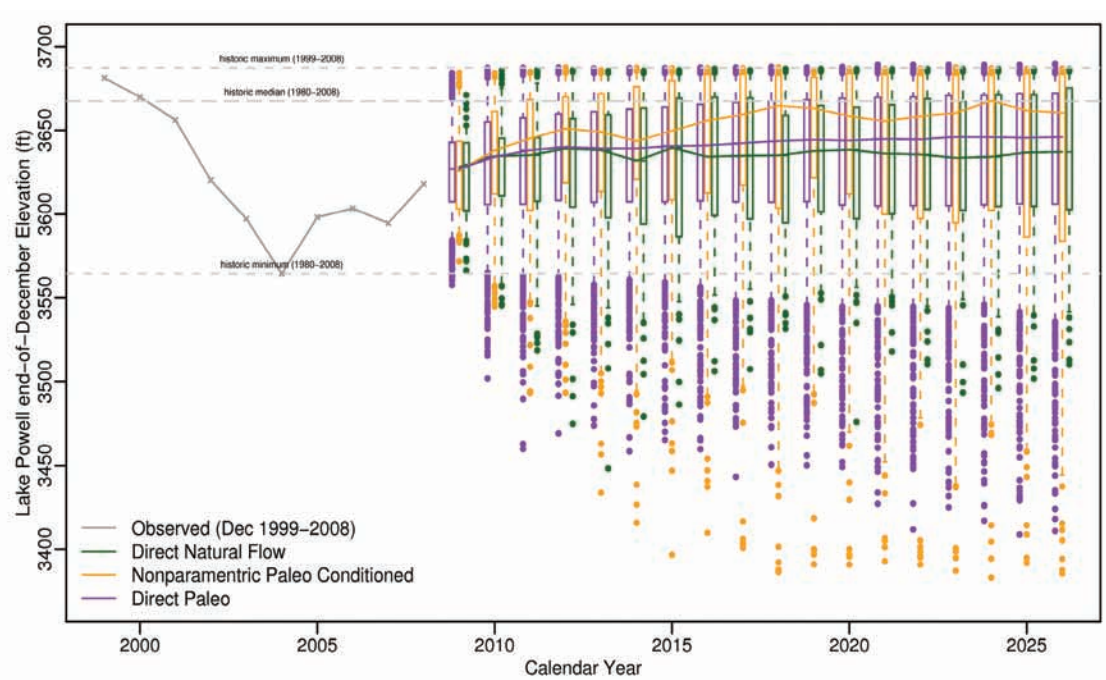
Appendix N analyzed the impacts of hydrologies outside the historical range of flows. In particular, the analysis focused on the sensitivity of hydrologic resources (e.g., reservoir storage, reservoir releases, and river flows) to alternative hydrologic scenario methodologies (e.g., derived from stochastic hydrology and tree-ring-based paleoreconstructions), particularly methodologies that generate sequences with greater hydrologic variability. Appendix N compared the “no action” alternative and the “preferred” alternative under three hydrologic scenario methodologies.

The first scenario, Direct Natural Flow, applies the Index Sequential Method (ISM) to the observed period of record (1906–2006), resulting in 101 hydrologic traces (Ouarda and others, 1997). The Direct Paleo scenario directly resamples the recent Lees Ferry reconstruction completed by Meko and others (2007) that extends back to the year 762 using the ISM, resulting in 1,244 hydrologic traces. The Nonparametric Paleo Conditioned scenario blends the hydrologic state (e.g., wet or dry) from the paleoreconstruction with the flow magnitudes from the observed record and results in 125 hydrologic traces (Prairie and others, 2007).

The results of the Interim Guidelines under these three alternative hydrologic scenarios in relation to Lake Powell elevations are shown in figure 5 for 2009 through 2026. The Nonparametric Paleo Conditioned scenario results in the highest median for all years; however, the historic median is still higher for every year during the interim period. It is evident that the alternative hydrologic scenarios increase the range of variability seen in Lake Powell elevations, particularly at lower elevations.

Adapting Colorado River Operations to a Changing Climate

The 2007 ROD implements a robust solution to the unique challenges facing Reclamation in managing the Colorado River. The Interim Guidelines, which extend through 2026, provide an opportunity to gain valuable operating experience and improve the basis for making additional future operational decisions during the interim period or thereafter. In addition, the Interim Guidelines were crafted to include operational elements that would respond if potential impacts of climate change and increased hydrologic variability are



Note: Solid lines through boxes connect the median. Bottom, middle and top of boxes represent the 25th, 50th and 75th percentiles, respectively. At whisker ends are 5th and 95th percentiles. Outliers are beyond the whiskers.

Figure 5. Projected Lake Powell elevations.

realized during the interim period. The coordinated operation element allows Lake Powell releases to be adjusted to respond to low reservoir storage conditions in either Lake Powell or Lake Mead. The shortage strategy element for Lake Mead includes a provision for additional shortages to be considered, after appropriate consultation. The Interim Guidelines also encourage efficient use and management of Colorado River water, and enhance conservation opportunities in the Lower Basin and the retention of water in Lake Mead through adoption of the ICS mechanism. Finally, the basin States have agreed to address future controversies concerning the Colorado River through consultation and negotiation before resorting to litigation. In sum, the Interim Guidelines preserve and provide Reclamation the flexibility to deal with and adapt to further challenges such as a future changing climate and persistent drought.

On December 13, 2007, Secretary of the Interior Dirk Kempthorne signed the ROD and called the Interim Guidelines the most important agreement among the seven basin States since the original 1922 Compact. The Interim Guidelines are in place through 2026 and include a provision that states, "Beginning no later than December 31, 2020, the Secretary shall initiate a formal review for purposes of evaluating the effectiveness of these Guidelines" (U.S. Department of the Interior, 2007, p. 56). Further knowledge of the impacts of a changing climate, both realized and projected, will be critical when such a review is initiated. Reclamation's Lower Colorado Region is committed to continuing this research and development program in order to do just that. For example, it is anticipated that the necessary tools will be in place in 2010 to analyze a suite of climate change scenarios within Reclamation's basinwide planning model (the Colorado River Simulation System, or CRSS). This and other efforts will further our ability to analyze the potential impacts of climate change and use that information in water and power operations and planning studies to be able to adapt, as appropriate, the operation and management of the river to a changing future climate.[4]

[4] See *http://www.usbr.gov/lc/region/programs/climateresearch.html* for a description of the research projects currently underway.

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Sustainability and River Restoration in the Colorado River Basin: A Climate Perspective

By Katharine L. Jacobs[1]

Abstract

Meeting the expanding demands of municipal water users while protecting hydropower, recreation, Tribal, agricultural, and environmental interests will become more challenging over time, particularly in the context of moving toward fuller utilization of upper Colorado River allocations. Additional stress will be placed on management systems by changes in the climate, particularly higher temperatures, which dramatically affect both water demand and water supply. Increasing demand, changing social values, and over-allocation of water supplies mean future "normal" droughts will lead to greater impacts and more water rights conflicts. Managing for sustainability involves being prepared for multiple climate-related challenges in addition to climate change—including difficulty in defining realistic management goals in light of long-term (decade-scale) "natural" variability in the context of a changing climate regime. Because water is a key "delivery mechanism" of climate change impacts, habitat managers need to be aware of expected changes in volume and seasonality of runoff and design adaptive strategies that will enhance the resilience of the habitats and species that they manage. More work is needed to better understand the impacts of climate change on groundwater supplies within specific watersheds and on the habitats that are directly or indirectly supported by groundwater. Finally, sustainability of managed ecosystems is not just about access to sufficient water, it is about access to money, information, and political support over time.

Introduction

Beyond the stresses caused by competing demands for water, multiple implications of climate variability and climate change need to be considered by habitat managers in the Colorado River Basin. Climate variability has always posed a significant challenge for habitat restoration and protection activities, but now variability occurs in the context of underlying climate change trends and the "Death of Stationarity" described by Milly and others, 2008. The "Death of Stationarity" message is that past climate conditions are no longer a good analogue for the conditions that will be experienced in the future. Although climate has never been "stationary" in the true sense of the word, anthropogenic change has added a new climate factor that is driving the system outside of its historical range. Greenhouse gases now entering the atmosphere will impact the climate system for centuries, even if humans start doing a better job of managing greenhouse gases in the short term (Solomon and others, 2009). As a result, in order to anticipate possible future conditions, managers will need to expand the range of historical, observed experience to consider a broader set of climate conditions to frame planning assumptions. For example, managers can build future scenarios based on instrumental records plus a blend of paleoclimate and (or) projected climate information, perhaps with the use of stochastic data to enrich the set of sequences that might be considered given the chosen climate context. These approaches may require new methods of integrating scientific information into decision processes in real time (Brekke and others, 2009).

In addition to needing to master the new uncertainties that come with climate change, ecosystem managers have not yet developed a full appreciation of variability beyond the seasonal-interannual (ENSO) timeframe. Sequencing of wet and dry years associated with decadal to multidecadal trends in sea surface temperature in the Atlantic and Pacific Oceans has been shown to influence both temperature and precipitation in various parts of the United States over the past centuries (Mantua and others, 1997; McCabe and others, 2007; McCabe and others, 2008; McCabe and Wolock, 2008). Some patterns in ocean conditions persist for multiple years and sometimes result in long-lasting climate trends that last a decade or longer. Strong correlations have been shown between these patterns in ocean temperature and climate conditions in some parts of the United States, particularly in the Southwest (fig. 1). At this time we have no way of predicting when the shifts in phase between wet and dry periods might occur because we do not yet have sufficient understanding of the mechanisms that cause them. The shifts can wreak havoc with water-supply planning and environmental restoration

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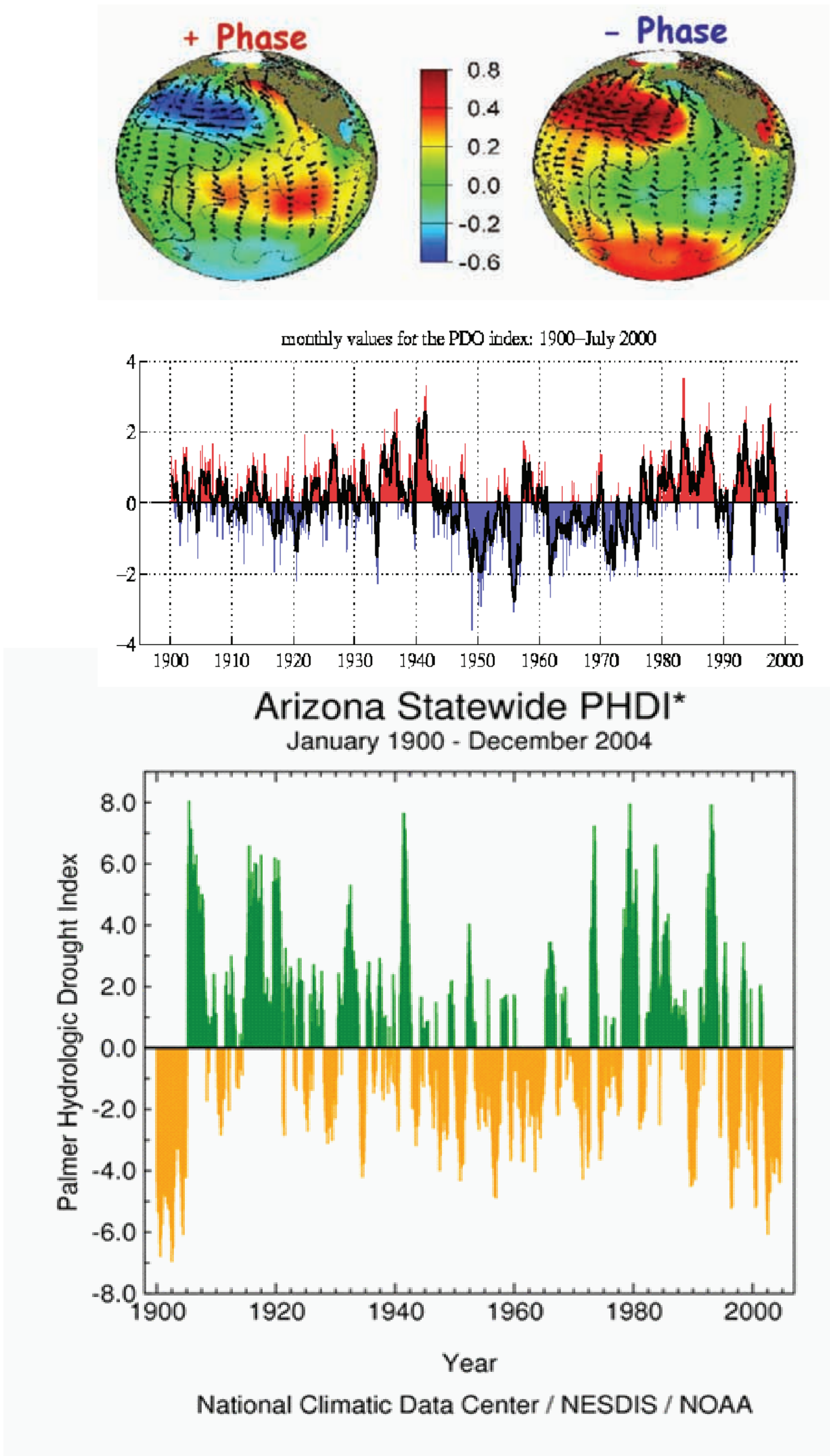


Figure 1. The Pacific Decadal Oscillation is a pattern of sea surface temperatures that is highly correlated with long-term (decadal) variability in precipitation and temperatures in parts of the United States.

efforts if they are not anticipated, and they need to be better understood in order to ensure that sufficient water supplies and (or) reservoir capacity are available even in the dryer portions of decadal cycles.

Further challenges come from the human values and regulatory requirements that control what condition managers are attempting to create through restoration efforts. If managers want to protect specific species in their current location (as is generally expected under Endangered Species Act requirements), there are different restoration challenges in the context of climate change than if it were possible/acceptable to facilitate a shift in managed areas to the north or to higher elevations. If, however, the restoration focus is to protect ecosystem functions rather than specific species, focusing on restoring the environmental conditions that protect those functions (e.g., seasonality of flows) may be the management objective, which leads to additional considerations related to the location of managed areas and access to water supplies.

Unfortunately, there is a general public perception that restoration efforts should recreate the “presettlement condition” (prior to human impacts), as if there were only one such condition. Since climate conditions have always changed, this expectation is not easy to meet. The fact that the extremes in the climate system are now moving outside of historical boundaries makes this even less reasonable. Further, the ability to create the quantity, quality, and seasonality of flows that are required for restoration supporting any specific ecosystem condition will be more difficult in light of the uncertainties associated with predicting the relevant variables into the future with enough specificity to make management decisions.

Anthropogenic climate change complicates the challenges posed by “natural” variability at various time scales. The consensus among climate experts is that across a variety of habitats, more extreme events, both floods and droughts, are likely to result (Karl and others, 2009). At this time we do not know whether climate change will modify the underlying drivers of “natural” variability (frequency of floods and droughts), but it is expected that the peaks will be exacerbated.

Incorporating Climate Information in Management Decisions

Water-management systems that are more responsive to changes in the climate system are needed. Most water rights systems allocate volumes of water based on an expectation of “normal” flows or at least flows within the historical range. New modes of management that reflect the increased understanding of the drivers of climate conditions are needed, with the potential to adjust management activities in real-time response to new types of science inputs, including probabilistic information about future conditions. The ability to respond to anticipated changes in seasonal and annual water availability, as well as changes in extremes (both floods and droughts), will be the hallmark of successful programs. It is possible that through enhanced monitoring and analysis efforts, trends can be identified much more quickly, allowing for adaptive management that incorporates a broader suite of information—from a variety of sources—including remote sensing, surface-water gages, the new National Phenology Network (which is designed to observe temporal and spatial changes in biological activity), groundwater-level monitoring, changes in species composition, etc. Combining all of these sources in real time presents significant cyberinfrastructure challenges, but an integrated understanding could also present opportunities for reducing the cost of habitat restoration and maintaining in-stream flows.

Where ecosystems are supported by groundwater, habitat managers need a better understanding of the changes in the groundwater system that may result from changes in precipitation and temperature associated with global warming. Although there is little empirical evidence, it seems likely climate change may result in a reduction in recharge in areas where temperatures are increasing, even if precipitation increases. However, these impacts are likely to be different from one groundwater basin to another because of differences in geology and recharge pathways. Changes in patterns of water demand and water supply for human uses as well as for ecosystems will emerge within tributary watersheds across the Colorado River Basin as temperatures increase and changes in precipitation patterns become more dramatic (Seager and others, 2007; Colorado Water Conservation Board, 2008). These changes in demand for groundwater also will affect the availability of groundwater to support environmental flows. Anticipating changes in the hydrologic cycle and impacts on water quality will be imperative for preservation and restoration of key environmental flow values.

Managing for Sustainability

Sustainability is a subjective concept and is particularly elusive as applied to natural ecosystems. Ecosystems have evolved in response to changes in climate and multiple other stresses for millennia, so managing specific ecosystems in specific locations as if there were a single “prehistoric” or “pre-intervention” condition is not consistent with the sustainability concept. Human interventions have already altered most hydrologic regimes. There are essentially no ecosystems that are untouched by human-induced changes, because the chemical composition of the atmosphere, atmospheric dynamics, and impacts to the climate system affect the entire globe even in places that are otherwise intact. Acknowledging that the desired management outcomes we select come from our own perceptions, experience, and values is an important step in defining sustainability for particular systems. Defining water sustainability goals requires decisions that result in a series of tradeoffs, with “winners” and “losers” associated with each intervention. For example, diversion of water from the mainstem of the Colorado River for habitat restoration in Arizona will limit the water available for ecosystems in the delta in Mexico. The Colorado River water that flows through the Central Arizona Project is viewed as a renewable and valuable water supply for Arizona, but it diverts water supplies that might otherwise have flowed into Mexico or California. Moving water from one location to another, or from one sector to another (such as agricultural to urban transfers), always results in impacts of some kind. The key to such adaptations is anticipating the impacts and mitigating them to the degree that is possible.

In this context, there is an increasing need to better understand how both climate variability and change (in combination) affect our ability to achieve habitat and species protection goals. As noted above, part of this challenge includes recognizing the impact of climate variability and climate change trends at multiple time scales on management outcomes. This approach requires continuing improvements in our understanding of the drivers of the climate system, the interactions between the climate system and ecosystems, and the development of monitoring and management systems that allow enough flexibility to experiment with using new information. Connecting science and decisionmaking in this context means building better relations that "bridge the gap" between habitat managers, researchers, climate scientists, and water managers. Such tools can include ways of visualizing trends in data, ways of explaining interrelations in complex systems, models that disclose statistical correlations between precipitation and temperature, and species viability, etc.

It is important for water and habitat managers to optimize the use of what we already know about climate change, rather than waiting for more detailed information that may or may not be more useful. There is a high probability of increases in temperature and changes in distribution and intensity of precipitation, so these changes need to be anticipated within the management system to achieve water and habitat sustainability goals. It has been established in the context of the Intergovernmental Panel on Climate Change (IPCC) that warming is "unequivocal" and that the likelihood that recent trends are significantly influenced by human activities is greater than 90 percent. The "new news" from the latest version of IPCC (Parry and others, 2007) is a strong conclusion based on 20 of 22 models that northern Mexico and the southern portions of the Southwest are expected to have less winter precipitation in addition to warmer temperatures. This widely accepted conclusion (Milly and others, 2005; Seager and others, 2007; Dettinger and Culberson, 2008) is critical to managing habitat in this region. Further, evidence exists that droughts are increasing in length and severity and that the intensity of precipitation is increasing because of the higher moisture content in the atmosphere that accompanies higher temperatures. This tendency toward more extremes—at both the high and the low end of the spectrum—will further challenge water and habitat managers.

Climate Change Impacts

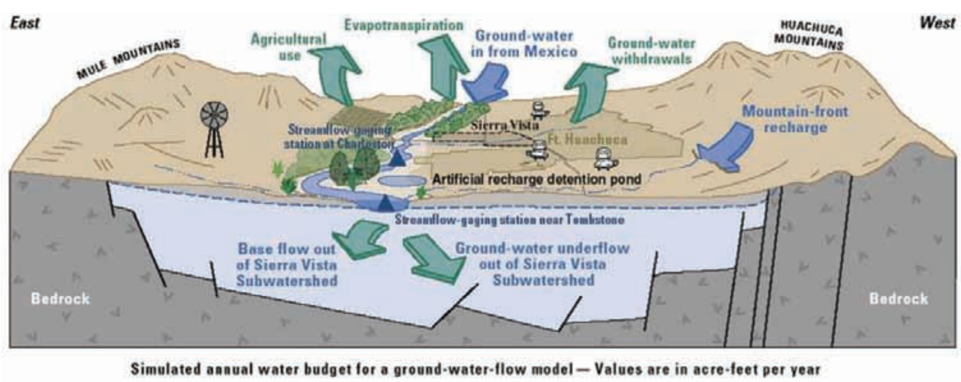
Water is a key delivery mechanism of climate change impacts—it is through the hydrologic cycle that the majority of climate change impacts can be felt. The observed changes in hydrology that are connected to climate change include changes in snowpack, seasonal patterns of runoff, increases in extreme precipitation, longer or more intense droughts, changes in water temperature and water quality, etc. (Stewart and others, 2005; Knowles and others, 2006; Karl and others, 2009). The impacts on human populations and the resources they value may be dramatically different depending on location and livelihoods. For example, ranchers who depend on rain-fed irrigation for grazing their cattle may have significantly more difficulty finding reliable forage; forest managers will face increasing risk from fire and bark beetles because of drought and more frost-free days; managers of habitat with endangered species need to be concerned that seasonal water availability could change dramatically, etc. For habitat managers, an important impact is that changes in timing of precipitation and runoff will affect environmental flow components that are critical for ecosystem health (low flows, high-flow pulses, floods).

Considerable focus has been placed on the likely reductions in flow of the Colorado River associated with climate change—the changes in temperature alone have significant impacts on both the supply side (increased evaporation from reservoirs, lower soil moisture, etc., leading to lower water availability) and the demand side (increased drought stress in plants, more water needed for irrigation, energy demand, etc.). Recent studies conducted within the National Oceanic and Atmospheric Administration (NOAA) Regional Integrated Science Assessments in the West (including researchers at NOAA, Bureau of Reclamation, Scripps, the University of Colorado, the University of Washington, and the University of Arizona) have reached a preliminary conclusion that a good estimate for reductions in supply is in the range of 15 to 25 percent by the year 2050, though this work is ongoing and no final conclusion has been reached. It is a useful exercise in any case to try to analyze the reasons why different models, methods, and datasets yield substantially different conclusions. Precipitation-runoff estimates at high elevations is an issue that is still being addressed. This is important since such a large proportion of the flow in the Colorado River is generated from snowpack at high elevations.

Although there has been a lot of focus on the Colorado River itself, little research exists on the implications for smaller tributaries, wetlands, or groundwater supplies within the watershed. Loss of snowpack—and resulting changes in seasonality of streamflow—will clearly impact these water supplies, but very few researchers have addressed the issue of groundwater implications of climate change or the implications for habitats dependent on the groundwater–surface-water interface (where surface water recharges the aquifer or groundwater aquifer outflow supports surface-water flows). It seems likely that a reduction in total streamflow will occur, and that this will result in less recharge, although in some cases major flood events have had significant impacts on aquifer storage. More research is needed in this area, because the implications of reductions in snowpack and changes in seasonality and intensity of precipitation differ for each watershed. The associated implications of climate change for water quality are understood at a conceptual level (e.g., higher temperatures reduce the oxygen level in streams), more fires will result in higher stream sediment loads, and higher runoff events can flush a load of pollutants into water bodies, but little is known at a scale that is useful for management decisions.

One way to think about the impacts of climate change within watersheds is changes in "partitioning" of precipitation—how much water is evaporated from bare soils, how much is evapotranspired by plants, how much runs off as surface flow, and how much enters the ground and recharges the groundwater supplies. This concept focuses on alternative pathways in the hydrologic cycle that can change in response to climate "drivers" like temperature. The following illustration of a cross section of the San Pedro watershed shows the fluxes in the hydrologic cycle as arrows (fig. 2). Clearly, if there is a reduction in winter snowpack, the amount of water that enters the aquifer as mountain front recharge will be reduced, which ultimately is likely to reduce the groundwater outflows that support the San Pedro River. Changes in seasonality of runoff are also critical for those who are working to protect habitat quality, because perennial flows are required for some species, and changes in the flow regime can affect multiple life-cycle components in ecosystems. There is much work to be done to enable us to understand the implications of reductions in snowpack, changes in seasonality of flows, and changes in intensity of precipitation for even one watershed, so generalizing lessons learned across the basins of the West is very challenging.

Managing for sustainability also requires a long-term perspective on how climate has varied in the past. Recently, Meko and others (2007) completed reconstructions of streamflows based on tree ring records that extend back more than 1,200 years. This reconstruction provides an opportunity to see how variability has changed over time, and also puts the climate of the past 100 years into perspective. As it turns out, the last 100 years generally was wetter than previous centuries, and the drought of the 1950s, which has always been considered to be "design drought" for the Southwest, was neither as deep or as long as droughts that occurred many centuries ago. The message that this sends to 21st century managers is that even without human-induced climate change, there have been devastating droughts that lasted for decades. The potential that the droughts of the future will be worse in a warmer world is very real, and most resource managers do not feel prepared for such droughts. Drought severity in a warmer world likely will be worse than recent historical drought experience because higher temperatures cause higher moisture stress, even if drought spells and reoccurrence patterns do



Simulated annual water budget for a ground-water-flow model — Values are in acre-feet per year

GROUND-WATER INFLOW

	Estimated range	2002 Estimates	2011 Projections
—Natural recharge	11,200–16,000	15,000	15,000
—Underflow from Mexico	3,000–3,400	3,000	3,000
—Total		18,000	18,000

GROUND-WATER OUTFLOW

	Estimated range	2002 Estimates	2011 Projections
—San Pedro base flow	3,250–6,250	3,250	3,250
—Net ground-water withdrawals		16,500	18,600
—Riparian and wetland evapotranspiration	6,230–7,700	7,700	7,700
—Ground-water underflow at Tombstone streamflow-gaging station	300–440	440	440
—Total		27,900	30,000

ANNUAL STORAGE CHANGE (no management measures)

—2002 Estimated	–9,900
—2011 Projected	–12,000

Figure 2. A cross section of the San Pedro River watershed in Arizona in the annual water budget.

not change. Evidence of this potential has been found by Breshears and others (2005) in analyzing tree mortality in the recent drought as compared to the drought of the 1950s.

A long-term perspective on climate variability is also helpful when managing for specific outcomes. Understanding trends in water-supply data when only 100 years of observed data are available can be very limiting—and in many watersheds fewer years of record are available. Depending on what years the trend line starts and ends, it is possible to come to entirely different conclusions about what is really happening to the water supply. For example, the long-term trend over the last century in flows in the Colorado River was clearly downward; however, if shorter time periods are selected for analysis, such as the period from 1955 to 1985, a very different conclusion would be reached about future water-supply availability in the region (fig. 3).

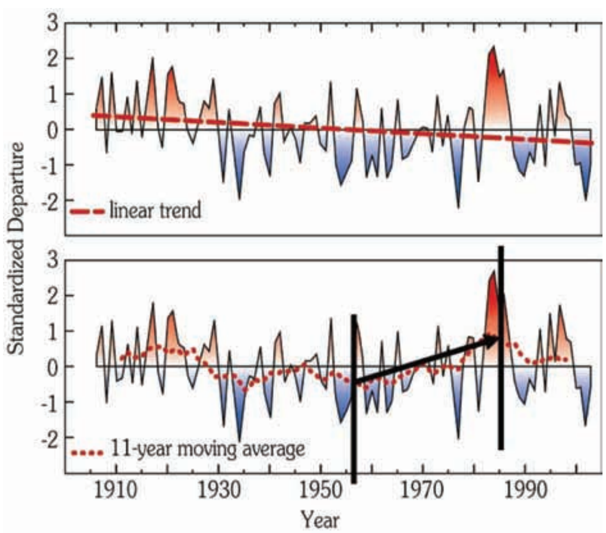


Figure 3. Long-term perspective of streamflow in the Colorado River (modified from McCabe and others, 2007).

Adaptive Management

Adaptive management is focused on monitoring the impacts of decisions that are made over time, in light of the fact that management decisions must proceed even if information is incomplete or inadequate. This management approach is essentially an ongoing experiment in optimization and a process for probing to learn more about the resource or system being managed. Thus, learning is an inherent objective of adaptive management. This is particularly appropriate in light of changing climate conditions. As we learn more, we can adapt our policies to improve management success and be more responsive to future conditions (Johnson, 1999).

Although adaptive management as a management framework is not always embraced by decisionmakers because it has a mixed record in the academic literature (Jacobs and others, 2003), it is better than managing changing systems as if there were no uncertainty. Clearly resource managers have to experiment with management options, because there are no perfect solutions available.

One approach to dealing with uncertainty is developing scenarios of a range of plausible future conditions and assessing how management objectives are affected by these alternative conditions. The careful use of scenarios can be helpful, because they can be used to assist in brainstorming potential options, evaluating the interaction between different kinds of variables, etc., before actually making decisions. The process of building scenarios is itself a learning process, because the work required to build credible baselines and trends builds understanding of the relations within complex systems. Further, the process of building scenarios can result in new knowledge networks among agency and academic scientists and researchers that can be useful resources for managers.

It is clear that we can make progress by improved monitoring of changing conditions and making better use of the data that we do collect. There is also a need to be more strategic about what is being monitored at what scale and time interval in order to identify and respond to regional and local trends and, thus, allow for better early warning systems. For example, because snowpack is a critical impact area for water resources, measuring snowpack dynamics in critical parts of the Colorado River watershed can improve our ability to project runoff conditions. There is also a need to continue to fund long-term observation stations to ensure the collection of longitudinal data, and for climate experts to engage more fully with resource managers in designing such systems.

Suggestions for enhanced monitoring while minimizing cost could include:

- Focus on critical or vulnerable systems;
- Build in operational, real-time delivery of observations;
- Provide better data access, storage, retrieval, and analysis systems;
- Provide for real-time trend analysis and visualization of data and develop "smart" monitoring systems;
- Provide feedback and evaluation of management impacts as part of each monitoring system.

Opportunities for environmental protection in the context of a changing climate include:

- Prepare for vulnerability in ecosystems by managing invasive species, protecting critical features of the natural hydrographs including low-flow standards, and providing for pulse flows that have important ecological benefits;
- Prepare for extreme events by protecting key habitat components, as preservation is always cheaper than restoration;

- Restore and maintain watersheds as an integrated strategy for managing water quality and quantity;
- Analyze effects on groundwater of drought and climate and protect groundwater recharge areas in critical habitats.

These suggestions are useful in any context—not just in the context of climate change. There are, however, multiple institutional and resource-related reasons why they are difficult to achieve.

Conclusions

Managing for water sustainability in the context of a changing climate brings multiple challenges. The demand for water supplies in many parts of the West is increasing over time because of shifts in use patterns at the same time that it appears supplies will be decreasing. This may be a zero sum game—and many decisions will have economic, political, or social consequences that overwhelm the ecological considerations. Key messages are that at a fundamental level, the past is no longer a good analogue for the future, as described in the "Death of Stationarity" article. Implications exist for water management and ecosystem management at multiple scales of time and space. Building planning scenarios of likely future outcomes to assess the impacts of a range of possible changes is one way to deal with uncertainty. A second important response is building flexibility into water management and ecosystem management systems and actively monitoring and assessing the effectiveness of management efforts. Although there are tradeoffs in flexible management systems because there is a reduction in certainty and a requirement for more professional judgment, still, decisions should be made that consider the ability of systems to remain resilient in the context of a range of future conditions. Finally, engagement between resource managers and climate experts could help frame the questions that need to be answered to incorporate both long-term climate trends and shorter scale variability into more sustainable resource management outcomes.

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Aquatic Production and Carbon Flow in the Colorado River

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Abstract

Dams alter physical and biological processes in rivers in predictable ways, yet we have little understanding of how dams alter carbon fluxes into rivers and secondary production (elaboration of biomass through time) of animals. Production is essential to understand how the size of fish populations might be limited by the amount of available energy. We hypothesize that dams reduce inputs of transported organic matter to downstream river reaches with a subsequent increase in photosynthesis providing the energy base for the food web. We have begun measuring primary and secondary production in the Colorado River below Glen Canyon Dam. Primary production, i.e., the rate of photosynthesis, increases with declining suspended sediment concentrations and can equal rates from small, well-lit streams suggesting primary production is an important carbon source for the river food web. Aquatic invertebrates derive a large portion of their diet from algae when rates of primary productivity are high. Secondary production, i.e., the rate of invertebrate biomass accumulation, ranged from high below Glen Canyon Dam to low downstream near Diamond Creek; this variance likely is driven in part by the availability of carbon from photosynthesis. Knowledge of carbon flow within a managed tailwater like the Colorado River will assist in predicting outcomes of management decisions that alter energetics of food webs.

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Introduction

The Colorado River drains a large fraction of the arid Intermountain West and is a primary water supply for users in seven States. The river holds a unique assemblage of fish species; of the 36 fish species that are native to the Colorado River system, 64 percent are found nowhere else (Carlson and Muth, 1989). The Colorado River has been extensively altered by dams to facilitate water storage and power generation. These dams alter the physical habitat and temperature regime in predictable ways (Ward and Stanford, 1983) and decrease biotic integrity, causing fish and invertebrate species to become locally extirpated. For example, the Green River in Utah below Flaming Gorge Dam lost more than 90 percent of its mayfly species following dam construction (Vinson, 2001) and now supports a productive, but nonnative, trout fishery. Four species of native fish are no longer found in the Grand Canyon reach of the Colorado River (Gloss and Coggins, 2005); one of the remaining species—humpback chub (*Gila cypha*)—is listed as endangered under the Endangered Species Act.

An important part of maintaining biological integrity at higher trophic levels is ensuring that there is a sufficient food supply to support the population. This need has been translated into policy as part of the strategic plan of the Glen Canyon Dam Adaptive Management Program, whose first goal is “Protect or improve the aquatic food base so that it will support viable populations of desired species at higher trophic levels.” But prior to managing the river for maintenance of an adequate food base it is necessary to measure carbon inputs to the ecosystem and determine how these are transferred up the food web to fish populations.

Declines in native fish populations and other undesirable changes in ecosystem function are, in part, a problem of energetics. Food limitation can be one of several aspects (e.g., predation, spawning habitat, migration) that can limit fish recruitment and production. For example, in the Colorado River tailwater of Glen Canyon Dam, artificially low water temperatures during most of the year limit rates of fish and invertebrate growth; high light penetration because of clear-water leads to increased rates of primary production; nonnative New Zealand mud snails (*Potamopyrgus antipodarum*) may represent a dead end for carbon flow in the food web because their thick shells protect them from fish predation; and nonnative trout, an important sportfish in the tailwater reach, may compete with and prey upon native fish in downstream reaches. Measuring organic matter flow into a river reach and through the food web in a common currency (g organic matter·m^{-2}·y^{-1}) provides a powerful framework for evaluating the effect of management actions on animal populations in the river. In addition, lower trophic levels will respond more quickly to changing dam operations than will slower-growing fish. The goals of this paper are to (1) describe why measurements of rates and sources of organic matter input into the river and associated production of animals can help us understand ecological function in heavily altered sections of the Colorado River, and (2) demonstrate the utility of these approaches from data we are collecting in the Grand Canyon reach of the Colorado River.

Carbon Inputs to the Base of River Food Webs

Animal production in any ecosystem, including rivers, is ultimately limited by the amount and quality of food resources entering the bottom of the food web. Physical conditions (e.g., habitat quality, temperature) certainly regulate the total animal production of an ecosystem, but the ultimate limits are set by the availability of carbon resources. Rivers with high rates of primary production or terrestrial inputs of carbon (i.e., leaf litter from streamside trees) can have higher rates of secondary productivity, assuming the physical conditions are also conducive to high production. For example, removing leaf litter inputs dramatically reduced secondary production of invertebrates in a mountain stream (Wallace and others, 1997). Secondary production of New Zealand mud snails in warm springs of the Yellowstone region are some of the highest ever measured for animal populations, but this is only possible because primary production of these springs is also extremely high (Hall and others, 2003). In turbid desert rivers, fish abundance can be higher in streams with higher rates of primary production (Fellows and others, 2009), suggesting that primary production is an ultimate control. In addition to the quantity of food resources, the quality of that food resource can also determine production. For example, adding nutrients to a heavily forested stream increased the nutritional quality, but not the quantity, of leaf litter that forms the base of the food web, thereby increasing invertebrate production (Cross and others, 2006).

We can categorize two main sources of carbon to rivers. Allochthonous carbon sources originate from outside the channel, such as leaves from streamside trees or organic matter that has been transported from a small headwater stream downstream to a large river. In contrast, autochthonous carbon is fixed by photosynthesis within the river channel by organisms such as algae or aquatic plants. Allochthonous inputs can dominate the carbon budget of many streams (Fisher and Likens, 1973) and rivers (Meyer and Edwards, 1990) and can be a dominant carbon source to consumers in food webs (Hall and others, 2000). Most streams and rivers are net heterotrophic, meaning that consumption of organic matter exceeds production of new organic matter, because allochthonous inputs allow ecosystem respiration to exceed primary production (Howarth and others, 1996; Webster and Meyer, 1997). Autochthonous production can exceed ecosystem respiration when the ecosystem is highly productive (e.g., small desert streams with warm water that receive abundant sunshine) and (or) when allochthonous inputs are minimal (e.g., spring streams that are for the most part isolated from the surrounding landscape) (Minshall, 1978). More often than not, the relative amounts of allochthonous versus autochthonous inputs vary through time; e.g., autochthonous algal production may dominate at certain times of the year when conditions promote high rates of photosynthesis (Roberts and others, 2007). For example, Roberts and others (2007) found that in a small Tennessee stream, autochthonous production dominated for roughly a 1-month period in the spring before leaf-out. Later in the spring and summer, shading by overstory trees limited algae growth, and in fall and winter leaf litter inputs supported elevated rates of ecosystem respiration, and autochthonous production was low.

Measuring the relative inputs of allochthonous versus autochthonous organic matter is an important step in a food web study because these resources represent the base of the food web, but relative differences in the quantity of these resources may not control which resource is actually providing the carbon source for animal consumers in a river. Algae, such as diatoms, are often a high-quality food source relative to more refractory allochthonous organic matter, so even a relatively small amount of primary production in a highly heterotrophic ecosystem may provide the primary energy source for food webs. In small streams, invertebrates derive their carbon from autochthonous sources at higher rates than predicted by relative differences in autochthonous and allochthonous inputs (McCutchan and Lewis, 2002). Evidence from large rivers suggests that algal production supports much of the animal secondary production, even in turbid rivers that carry large quantities of terrestrial organic matter where algal production is minimal, (Thorp and Delong, 2002). The Riverine Productivity Model (Thorp and Delong, 2002) posits that, despite large quantities of terrestrial inputs either from flood plains or from inefficient processing by upstream reaches, locally produced algal carbon should provide the base for riverine food webs. Evidence supports this model. Carbon isotope data from turbid, desert rivers show that primary production within the river channel supplies nearly all of the carbon to animals, despite high terrestrial inputs (Bunn and others, 2003). Primary production was locally high in these rivers even though they were turbid, and the combination of locally high production with high nutritional quality of algae relative to terrestrial inputs likely contributed to the importance of algae to the food web (Bunn and others, 2003). Hamilton and others (1992), also reported that in grass-dominated flood-plain lakes, animals received nearly all of their carbon from attached microalgae and not from the grass itself.

Production in the Colorado River Below Glen Canyon Dam

Primary Production and Consumption by Invertebrates

"Open-channel" methods are being used to measure primary production on the Colorado River (Odum, 1956; Hall and others, 2007). This procedure measures the change in oxygen (O_2) concentrations in the river as a surrogate for carbon because photosynthesis releases O_2 at approximately the same molar ratio as carbon fixation. Seasonally, we measure O_2 concentration throughout 2 nights and 1 day at five locations in Grand Canyon ranging from Marble Canyon to Diamond Creek. To calculate gross primary production (GPP; i.e., the rate of photosynthesis not including algal respiration), we use a model fitting procedure following Van de Bogert and others (2007), where we fit the following model to the O_2 data:

$$C_t = C_{t-1} + \frac{GPP}{z_t} \times \frac{PAR_t}{\sum PAR} + \frac{CR\Delta t}{z_t} + K(C_s - C_{t-1})\Delta t$$

C_t and C_{t-1} are O_2 concentrations across a 5-minute time step (Δt); C_s is the calculated saturation concentration of oxygen at a given temperature and barometric pressure. K is the rate of oxygen exchange at the air-water interface (1/d) and is calculated on the basis of measured oxygen exchange in the first 20 kilometers (km) of river (R.O. Hall and others, unpub. data, 2009); z_t is water depth (meters, m) at time t; PAR_t is the instantaneous amount of light hitting the river ($\mu E\ m^{-2}\ s^{-1}$) over a reach length equal to 80 percent of the O_2 travel distance; and ΣPAR is the total light summed for the day. Modeling oxygen concentrations and solving for GPP and ER (ecosystem respiration) is superior to standard calculations (Hall and others, 2007) because it allows calculating uncertainty in any one metabolism estimate. We calculated light as a function of river topography by following Yard and others (2005). The two variables that were solved for were GPP (g $O_2\ m^{-2}\ d^{-1}$) and community respiration (g $O_2\ m^{-2}\ d^{-1}$, CR). Because the river was consistently supersaturated with O_2, it was not possible to accurately estimate respiration using this technique, so we solved for CR, but the values were not reported. CR is not robust because it is not known what the O_2 concentration would be in the absence of biological activity. The common assumption is that streams would be at air-saturation if there were no CR and that CR lowers O_2 from this air saturation. Because the river was supersaturated, we have no reference point for which to measure respiration. GPP estimates, on the other hand, are robust because we are modeling the amplitude of the diel excursion and not the absolute concentration. We were able to measure rates of GPP despite extremely high rates of reaeration driven by rapids. Diel changes in oxygen concentrations were about 0.1 to 0.4 milligrams of oxygen per liter (mg O_2/L), which is small but easily modeled (fig. 1). We solved the model by minimizing the negative log-likelihood function between the model and the data. Because we measure invertebrate production by using g ash-free dry mass (AFDM, equivalent to organic matter), we converted these oxygen fluxes to organic matter assuming molar ratios between organic matter and O_2 = 1.

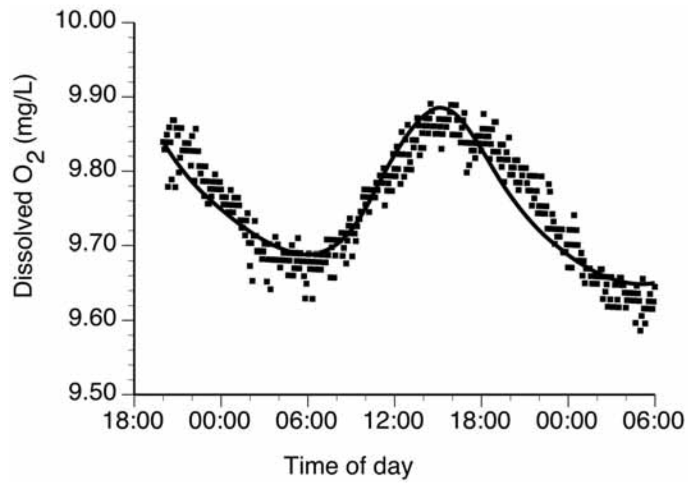

Figure 1. Example of oxygen data (points) versus model (line) for one metabolism calculation near National Canyon, AZ, from July 2008. Gross primary production was 2.6 g $O_2\ m^{-2}\ d^{-1}$.

Gross primary production was strongly a function of suspended sediment concentrations (fig. 2); high sediment concentrations block light, thus reducing primary production. Because sediment concentration increases downstream, production tends to decline downstream when considering all seasons. The rates of primary production were similar to those in small streams across the United States (Mulholland and others, 2001), including those from high-light areas, which ranged from 0.2 to 24 g organic matter $m^{-2}\ d^{-1}$ (Hall and Tank, 2003). Rates of primary production were greater than 10-fold higher than for water-column-based rates in tropical rivers (Lewis, 1988). The role of benthic algae in contributing to production in these tropical rivers was unknown but considered small (Lewis, 1988). Most production in the Colorado River is likely from river-bottom algae, though planktonic algae likely contribute to primary production because of a moderate amount of chlorophyll in the water column (0–2 micrograms chlorophyll *a* per liter (µg Chl *a* /L)). Because rates of GPP in the Colorado River can be as high as rates from small streams, the flux of carbon to this river from autochthonous primary production may be high enough to be important for

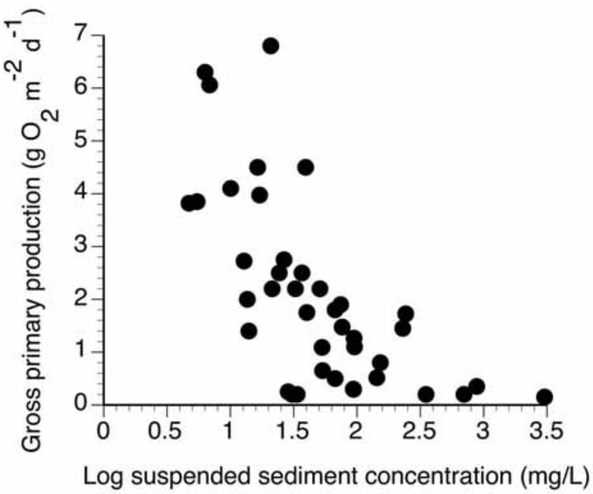

Figure 2. Daily rates of gross primary production decline as a function of $\log_{10}$ suspended sediment concentrations.

consumers, even though rates essentially are zero during times of year of low water clarity.

We also are attempting to make open-channel measurements of GPP in the Glen Canyon tailwater. However, because dam operations contribute to daily changes in O_2, in addition to the primary production of interest, open-channel measurements of GPP will require a different approach that is still being developed. Rates of GPP estimated from chambers that contained individual algae-covered rocks were very high: 15 g organic matter m^{-2} d^{-1} (Brock and others, 1999). This rate is up to 10 times higher than average rates for Grand Canyon. However, comparisons between chamber estimates and open-channel estimates must be made with caution, because high spatial heterogeneity in the distribution of river-bottom algae makes scaling rates measured on individual rocks up to the entire reach difficult. Nonetheless, these limited chamber data suggest that rates of production in the Glen Canyon reach are likely to be very high.

The rates of GPP in Grand Canyon are high enough for algae to represent a significant food resource for animal consumers. We have been measuring the diets of animals from all locations and across all seasons to calculate flows of carbon from basal resources into animal populations. These data show that algae (in this case, mostly microscopic algae known as diatoms) can constitute a large fraction of invertebrate gut contents (fig. 3); diets for the two taxa shown in figure 3 (*Simulium arcticum*, a filter-feeding blackfly, and *Gammarus lacustris*, a small crustacean) can contain up to 60 percent diatoms. Further, the proportion of diatoms consumed is positively related to the rate of primary production at the time and place the invertebrates were collected. These preliminary data suggest that below Glen Canyon Dam, primary productivity supports the growth and production of animal consumers. This finding is consistent with what is known about other desert rivers (Bunn and others, 2003) and theories of carbon flow in big-river food webs (Thorp and Delong, 2002).

Secondary Production of Invertebrates

The effect of large dams on diversity and assemblage structure of invertebrates in downstream ecosystems is well known. Many species of invertebrates have lifecycles that are cued in some way to temperature (Elliott, 1978). Because of relatively cold and constant temperatures downstream from high-head dams, many invertebrates are unable to complete their lifecycle and therefore become locally extirpated (Sweeney and Vannote, 1978). Consequently, the number of

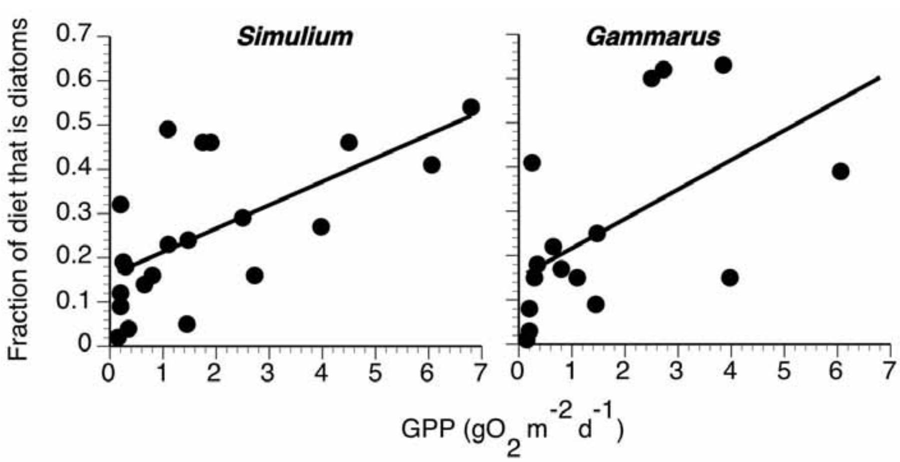

Figure 3. The fraction of invertebrate diet derived from diatoms increases with increasing rates of gross primary production (GPP). Lines are statistically significant least squares regressions, r^2 for *Simulium* = 0.41 and for *Gammarus* = 0.31.

invertebrate species often is lower below large dams than in free-flowing rivers. Before construction of Flaming Gorge Dam, the Green River contained more than 30 species of mayflies. After the closure of Flaming Gorge, the number of mayfly species declined to one common and two rare species (Vinson, 2001). Fewer data are available for the Colorado River below Glen Canyon Dam. Upstream from Lake Powell in Cataract Canyon, Haden and others (2003) found 49 invertebrate taxa of which 9 were mayflies. The Colorado River downstream from Glen Canyon Dam contains about 10 common taxa, none of which are mayflies (Stevens and others, 1997; W.F. Cross and others, unpub. data, 2009). The most common species in this reach are nonnative (i.e., *Oligochaetes*, *Gammarus lacustris*, and *Potamopyrgus antipodarum,* the New Zealand mud snail), suggesting that the Colorado River downstream from Glen Canyon Dam is best suited for stenothermic, cosmopolitan taxa.

To examine the degree to which the amount of invertebrates available for consumption by fish potentially limits the abundance of fish populations, it is necessary to estimate invertebrate production. Invertebrate production represents the amount of invertebrate biomass produced per area (square meters) per time (month, year). In other words, invertebrate production measures the flow of carbon per time through invertebrate assemblages. Although the exact procedures for determining invertebrate production are complicated, production is essentially the product of invertebrate biomass and invertebrate growth rates (Benke, 1984). Invertebrate biomass in tailwater sections immediately below dams is often high (Vinson, 2001), but it is not possible to estimate secondary production based solely on biomass because growth rates are strongly and positively related to temperature and taxonomic identity (Benke, 1984; Huryn and Wallace, 2000).

In contrast to what is known about benthic invertebrate assemblage structure, little is known about how dams alter invertebrate production. We have begun measuring assemblage-level secondary production from six sites in the Colorado River. The upstream site is in the tailwater and runs from Glen Canyon Dam to Lees Ferry. The downstream site is 240 river miles from the dam, at Diamond Creek, and four sites, more or less evenly spaced, are in between. To measure secondary production, we measure taxon-specific (a taxon is grouping of organisms, for example, mayflies) abundance and biomass monthly (Glen Canyon and Diamond Creek) or seasonally (four sites in Grand Canyon). We collect 18 to 20 samples per site each sampling period from a variety of habitats, sort, identify, and measure the length of invertebrates to the nearest 0.1 mm to estimate biomass using length-mass regressions for each taxon. We multiply these estimated biomasses by empirically measured, size-specific growth rates to calculate production as a flux (g organic matter m^{-2} y^{-1}). Secondary production is habitat weighted to reflect the fraction of different habitat types (e.g., cobble bars, cliff faces, sand, etc.) that are present within that particular reach of river. Currently we have data analyzed for 1 year at Glen Canyon and Diamond Creek.

Invertebrate secondary production was about 50 times higher at Glen Canyon than Diamond Creek (fig. 4). At Glen Canyon, production was dominated by New Zealand mud snails, scuds, and freshwater worms (subclass Oligochaeta). Annual invertebrate production in this reach is high relative to many streams and rivers and is in the upper 25 percent of values sampled from the literature (R.O. Hall, unpub. data, 2009). In contrast, annual secondary production in the Colorado River near Diamond Creek is in the bottom 10 percent of values from other streams and rivers and is in the range of “low production” values from Huryn and Wallace (2000). This difference in productivity between the two reaches is likely caused by higher primary production and more abundant hard surfaces in Glen Canyon; the sandy and unstable surfaces that are common along downstream reaches support lower invertebrate biomass and secondary production. It should be noted that the invertebrates that formally were present in this river may have had higher biomass and production in sandy sediments than those currently found.

Which of these two rates of secondary production is likely closest to that for pre-dam conditions? We do not know at this time because there are no secondary production estimates for river reaches in the Colorado River Basin, but we can examine invertebrate biomass at other sites as a first approximation. For example, average biomass on cobble habitats in the relatively unimpacted Cataract Canyon reach was 0.4 $g \cdot m^{-2}$ (Haden and others, 2003), which is comparable to our preliminary estimate of 0.55 $g \cdot m^{-2}$ for cobble habitats at Diamond Creek. For comparison, invertebrate biomass in the Glen Canyon tailwater reach is 7 $g \cdot m^{-2}$, or 17-fold higher than Haden and others’ (2003) value for Cataract Canyon. Despite

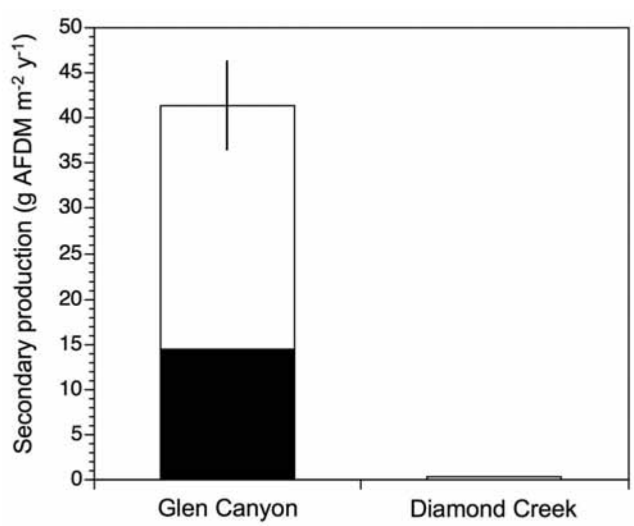

Figure 4. Secondary production in Glen Canyon reach is much higher than that for Diamond Creek reach of the Colorado River, AZ. The dark section of the bar for Glen Canyon is secondary production of New Zealand mud snails (*Potamopyrgus antipodarum*). Error bar is 95 percent bootstrapped confidence interval.

that biomass is similar between these two sites, we cannot speculate that production is the same because the thermal regime and assemblage structure are so different between the two sites that it is likely that assemblage-level biomass turnover and, therefore, secondary production will strongly differ also. Thus it may be that the high secondary production found immediately below a dam may be anomalously high relative to unregulated reaches or reaches where sediment inputs constrain primary and secondary production.

Prospectus

Despite a large body of research examining primary production (Mulholland and others, 2001; Roberts and others, 2007) and secondary production (Huryn and Wallace, 2000) in small streams, knowledge of primary and secondary production in nontidal rivers lags far behind. Measurements of phytoplankton and benthic production for many rivers using chamber approaches (e.g., Lewis, 1988; Cotner and others, 2006; Fellows and others, 2009) show that primary productivity can range from very low to high. In the Colorado River, rates of primary production essentially are unknown outside of rates for reservoirs (e.g., Gloss and others, 1980), and we are only beginning to measure rates of secondary production for animals. A limitation of our research in Grand Canyon is that we have no such data from before the construction of the dam, so we do not have a firm understanding of ecosystem function in the absence of a large dam. Currently, the only way to approximate pre-dam conditions is to perform similar measurements is parts of the Colorado River less altered by dams and other human activities, e.g., Cataract Canyon, Westwater Canyon, and the Yampa River. The huge reductions in downstream carbon transport and insect biodiversity (Vinson, 2001) and changes to habitat suggest that sections of the Colorado River less altered by dams will function much differently.

We argue that knowing rates of organic matter flow in the food web is critical for evaluating how management actions affect animal populations and ecosystem processes; evaluating the effect of management actions on resources is a critical step in the adaptive management process. For example, temperature strongly controls growth rates of invertebrates (Cross and others, in press; Huryn and Wallace, 2000). If a selective withdrawal structure is installed on Glen Canyon Dam to raise the temperature of releases, as was done for Flaming Gorge Dam, how will temperature-mediated increases to invertebrate growth rates alter secondary production and thus food availability for fish? If sediment inputs to the Colorado River increase because of sediment augmentation, how will reductions in water clarity alter riverine primary production and also secondary production of animals? Answers to these questions require detailed knowledge of food web energetics.

Acknowledgments

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An Overview of the Spread, Distribution, and Ecological Impacts of the Quagga Mussel, *Dreissena rostriformis bugensis*, with Possible Implications to the Colorado River System

By Thomas F. Nalepa[1]

Abstract

The quagga mussel (*Dreissena rostriformis bugensis*) was first found in the Great Lakes in 1989 and has since spread to all five lakes. Although its spread through the system was slower than that of the zebra mussel (*Dreissena polymorpha*), once established, it replaced zebra mussels in nearshore regions and is colonizing deep regions where zebra mussels were never found. Outside the Great Lakes Basin, quagga mussels do not appear to be increasing to any extent in the Ohio and Mississippi Rivers, even after being present in these rivers for over a decade. In contrast, numbers in the Colorado River system have continued to increase since the quagga mussel was first reported. It will likely become very abundant in all the reservoirs within the Colorado River system, but attain limited numbers in the mainstem. Ecological impacts associated with the expansion of quagga mussels in the Great Lakes have been profound. Filtering activities of mussel populations have promoted the growth of nuisance benthic algae and blooms of toxic cyanobacteria. In addition, the increase in quagga mussels has led to a major disruption of energy flow though the food web. An understanding of food webs in the Colorado River system, particularly the role of keystone species, will help define future ecological impacts of quagga mussels in this system.

Introduction

Two species of dreissenid mussels*, Dreissena rostriformis bugensis and Dreissena polymorpha* (quagga mussel and zebra mussel), are part of a group of biofouling, filter-feeding bivalves that are spreading around the world (Karatayev and others, 2007). When established in a new water body, these dreissenid species can increase rapidly and attain densities that generate far-reaching changes in physical, chemical, and biological components of the ecosystem. Many studies have documented ecological impacts of these two invading species, and broad patterns have emerged that are consistent across water bodies. Thus, to a certain extent, some ecological impacts can be predicted and prepared for. Yet other impacts have been unexpected and unique to a given taxa or habitat associated with the invaded system.

For several reasons, less is known of the specific life history, environmental tolerances, and impacts of quagga mussels compared to zebra mussels. The zebra mussel colonized North America first and quickly attained high densities, resulting in ecological changes that were widely evident and well documented (Nalepa and Schloesser, 1993). In comparison, the quagga mussel spread less rapidly, and impacts could not, at least at first, be readily discerned from the zebra mussel. Recent evidence, however, suggests that although ecological changes in the Great Lakes resulting from the proliferation of quagga mussels are functionally similar to those of the zebra mussel, the changes are more severe and pervasive in scope. As studies show, the quagga mussel spreads just as rapidly as the zebra mussels once established, is more flexible in colonizing different habitats, and attains higher densities in certain lake areas.

This paper summarizes current knowledge of the spread, life habit characteristics, and broad ecological impacts of the quagga mussel. Given the discovery of quagga mussels in the Colorado River system, such a summary may be useful when assessing ecological risks to this system. Quagga mussel characteristics and ecological impacts are presented in relation to the zebra mussel since both species have been introduced into Western States, and both frequently co-inhabit an invaded system during the early stages of the colonization process. Both dreissenid species attach to hard substrates and create clogging problems for power companies, water plants, and other raw-water users; however, it is beyond the scope of this summary to include a discussion of control options.

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Expansion Patterns and Taxonomic Definition

The quagga mussel was first reported in North America in 1989 in the eastern basin of Lake Erie. Like the zebra mussel, which was discovered several years earlier, the quagga mussel was likely introduced into North America via the discharge of ballast water from transoceanic ships. Based on genetic studies, these first North American individuals appear to have originated from the lower Dnieper River, Ukraine (Spidle and others, 1994; Therriault and others, 2005). Although given a common name, the taxonomic status of the quagga mussel was at first unclear, but was later determined to be *Dreissena bugensis* on the basis of allozyme data and morphological characters (Spidle and others, 1994). In subsequent analysis, this species was also found to be genetically similar to *D. rostriformis*, which is a brackish water species found in the Caspian Sea (Therriault and others, 2004). Given this clear separation in environmental tolerances (freshwater versus brackish water) and following rules of nomenclature, *D. bugensis* is currently considered a freshwater race of *D. rostriformis* and referred to as *Dreissena rostriformis bugensis.*

After first discovery in Lake Erie, the quagga mussel proceeded to spread into all the other Great Lakes, first into Lake Ontario, then into Lakes Michigan and Huron in 1997, and finally into Lake Superior in 2005 (Nalepa and others, 2001; Grigorovich, Kelley, and others, 2008). It was found in the Ohio and Mississippi Rivers in the mid-1990s, and in Lake Mead within the Colorado River system in 2007 (U.S. Geological Survey Web site: *http://nas.er.usgs.gov/taxgroup/mollusks/zebramussel/*). The spread of the quagga mussel within the Colorado River system has been rapid; by the end of 2008, it was reported in over 30 lakes and reservoirs in Arizona, California, Colorado, Utah, and Nevada. The likely vector by which mussels spread from the east to the far west was via the overland transport of recreational boats.

A unique aspect of quagga mussel populations in North America is the presence of two phenotypes. Although genetically similar (Claxton and others, 1998), these two phenotypes prefer vastly different habitats. In the Great Lakes, one phenotype (*D. r. bugensis* "sensu stricto-eplimnetic"; Claxton and others, 1998) is found exclusively in shallow-warm bays and basins, and the other phenotype (*D. r. bugensis* "profunda"; Dermott and Munawar, 1993) is found mostly in deep, cold offshore regions but also in some nearshore areas above the thermocline. The profunda phenotype has not been specifically reported from European waters, but some individuals from the Ukraine resemble North American specimens (A. Protosov, Institute of Hydrobiology, Ukraine, written commun., January 2009). Interestingly, North American specimens of profunda are more genetically similar to North American specimens of the epilimnetic phenotype than to specimens from the lower Dnieper River, Ukraine (Spidle and others, 1994). The dominant phenotype found in various water bodies in the Western United States, including the Colorado River system, is not clear at this time.

Although the quagga mussel has been found in large river systems in eastern Europe, it tends to reach greatest abundances in lakes and reservoirs (Mills and others, 1996; Orlova and others, 2005). This species was confined to its native range in the lower Dnieper–Bug River systems (northern Black Sea) until the late 1940s/early 1950s when a series of reservoirs were constructed on the Dnieper River system (Orlova and others, 2005). It is believed that these impoundments led to environmental changes (i.e., reduced water velocity, more stable temperatures), which better suited this species. Over the next several decades, the quagga mussel gradually expanded its range into the Volga River and Don–Manych River systems and more recently (2004–2007) into the Rhine, Danube, and Main Rivers in central Europe (Popa and Popa, 2006; Molloy and others, 2007; van der Velde and Platvoet, 2007). Overall, population growth in European rivers has been less rapid than in lakes/reservoirs, and abrupt, unexpected declines have been reported in some river systems (Zhulidov and others, 2006). Similar expansion patterns (i.e., a preference of lakes/reservoirs over rivers) are apparent in North America. For instance, while quagga mussels increased rapidly once established in Lakes Ontario and Michigan (Mills and others, 1999, Nalepa and others, 2009), a recent study in the upper Mississippi and Ohio Rivers found that quagga mussel distributions had not greatly expanded since being reported 10 years earlier, and densities remained uniformly low (Grigorovich, Angradi, and others, 2008). For the Colorado River system, these expansion patterns would indicate that the quagga mussel will increase more rapidly and attain greater abundances in the reservoirs of this system than in the river itself.

Physiological/Environmental Tolerances and Morphological Characteristics

The quagga mussel has several physiological and morphological features that allow it to proliferate in lake habitats where environmental conditions limit zebra mussels. In laboratory studies of both species, quagga mussels had a lower respiration rate under different seasonal temperatures and a higher assimilation efficiency, particularly at low food concentrations (Baldwin and others, 2002; Stoeckmann, 2003). Lower respiration and higher assimilation efficiency allow quagga mussels to better survive and grow under a wider variety of food regimes. In the Great Lakes, quagga mussels are expanding in offshore regions where food resources are naturally low and are also attaining high densities in shallow regions where food can be limiting during certain seasonal periods. These physiological traits are the likely reason why quagga mussels are displacing zebra mussels in many lake areas (Wilson and others, 2006; Nalepa and others, 2009). In addition, quagga mussels can reproduce at lower water temperatures compared to zebra mussels. Both quagga mussel phenotypes displayed gonadal development and spawned at water temperatures of 4–9 degrees Celsius (°C), whereas zebra mussels showed no reproductive activity at these low temperatures (Claxton and Mackie, 1998). Thus, quagga mussels can not only reproduce and thrive in deep, hypolimnetic regions, but can also spawn earlier in the spring than zebra mussels in shallow, epilimnetic regions.

As noted, population growth of quagga mussels in large river systems such as the Ohio and Mississippi has been slow (Grigorovich, Angradi, and others, 2008). Large rivers usually have elevated levels of suspended inorganic sediments (silt and clay), which negatively affect dreissenids in various ways. Inorganic particles foul gills and interfere with respiratory function. Also, these particles, although filtered, have no nutritional value. Mussels expend energy in expelling these particles that is better spent for growth and reproduction. Since quagga mussels are less widely distributed than zebra mussels in the Ohio and Mississippi Rivers (Grigorovich, Angradi, and others, 2008), it seems logical to assume that quagga mussels are less suited physiologically to handle suspended particulates. Yet laboratory experiments have shown that the two species respond similarly to elevated levels of suspended material; to a degree, both species were able to adapt to increased levels of turbidity (Summers and others, 1996). Regardless of some ability to adapt, both species are negatively affected by high concentrations of suspended sediments. The potential for zebra mussel growth was zero/negative at suspended sediment concentrations greater than 100 milligrams per liter (mg/L) (Madon and others, 1998). Concentrations typically found in the Ohio and Mississippi Rivers are below this level, whereas levels in the Missouri River are far above it (Summers and others, 1996). This may explain why few zebra mussels and no quagga mussels were found in the Missouri River system in a recent study (Grigorovich, Angradi, and others, 2008).

While quagga mussels thrive in deep, continuously cold environments, of relevance to their expansion in the Southwest United States is their tolerance to high summer temperatures. In several laboratory studies, quagga mussels were found to be less tolerant of elevated water temperatures compared to zebra mussels. The upper thermal tolerance limit for quagga mussels was about 30 °C, but could be as low as 25 °C because mussels could not be maintained in the laboratory at the latter temperature (Domm and others, 1993; Spidle and others, 1995). Given this, it is unlikely temperature will limit populations in the mainstem of the Colorado River system where temperatures range between 5 and 20 °C (Kennedy, 2007). In Lake Mead, mean monthly temperatures in the summer are 26–29 °C in shallow regions (<4.5 meters (m) water depth), and thus temperature-induced stress may eventually limit populations in this region. However, mean temperatures do not exceed 22 °C in deeper regions (>18 m water depth) (*http://www.missionscuba.com/lake-mead/lake-mead_average-water-temp.htm*). When considering temperature limits, other environmental factors must also be considered such that laboratory studies of upper lethal temperatures are often not good predictors of success in the natural environment. Quagga mussels exposed to unfiltered Ohio River water survived high, sublethal temperatures (>30 °C) better than zebra mussels (Thorp and others, 2002). This was attributed to the differential ability of quagga mussels to obtain and assimilate food at higher temperatures. Relevant to this issue, it is noted that quagga mussels are presently very abundant even in the shallow, warmer regions of Lake Mead (B. Moore, University of Las Vegas, oral commun., November 2008).

Besides temperature, another important environmental variable that affects quagga and zebra mussel distributions, and eventual population densities, is calcium concentration. Mussels require calcium for basic metabolic function and for shell growth. Based on field distributions, quagga mussels apparently have a slightly higher calcium requirement than zebra mussels. In the St. Lawrence River, quagga mussels were not found in waters with calcium concentrations lower than 12 mg/L, while zebra mussels were present (but not abundant) at concentrations as low as 8 mg/L (Jones and Ricciardi, 2005). Calcium concentrations in the Colorado River system are far greater than the values above, so calcium limitation is not an issue (Whittier and others, 2008). Indeed, high concentrations in this system (>80 mg/L) may favor quagga mussels over zebra mussels (Zhulidov and others, 2004). A summary of mussel tolerance limits for other environmental variables, such as dissolved oxygen, pH, and salinity, is provided in Cohen (2007).

The shell morphology of the quagga mussel differs from the zebra mussel in that it has a rounded ventral margin compared to one that is sharply defined. The lack of a flattened ventral surface does not allow the quagga mussel to attach as tightly to hard surfaces as the zebra mussel, and may prohibit it from easily colonizing habitats with strong water velocities such as found in some rivers. Unlike zebra mussels, however, quagga mussels do not necessarily need to attach to hard substrates. They can lie unattached on their longer, wider lateral side, which is an advantage in soft substrates because it prevents sinking. The profunda phenotype has an incurrent siphon that is far longer than the incurrent siphon of both the epilimnetic phenotype and the zebra mussel (fig. 1). This elongated siphon, which can be three-fourths the length of the shell, is a characteristic of bivalves adapted to inhabiting soft sediments. It allows filtration above the layer of fine inorganic particles generally found suspended at the sediment-water interface.

General Ecological Impacts

Ecological impacts of dreissenids, both quagga mussel and zebra mussel, are a function of achieved densities and characteristics of the invaded system. Where conditions are favorable and dreissenids become abundant, fundamental changes in energy and nutrient cycling occur, and all

Figure 1. Comparison of the incurrent siphon of the zebra mussel (top), quagga mussel-epilimnetic phenotype (middle), and quagga mussel-profunda phenotype (bottom). Note the longer siphon of the profunda phenotype.

components of the food web are affected. Several articles have provided excellent, detailed summaries of ecological impacts of dreissenids (Strayer and others, 1999; Vanderploeg and others, 2002), so only far-reaching changes that have strong implications to resource managers will be presented here. Dreissenids are filter feeders and hence remove phytoplankton and other particulates from the water. These filtered particles are ingested and assimilated or deposited on the bottom as feces or pseudofeces. Feces is material that is ingested but not assimilated, and pseudofeces is material that is filtered but not ingested (rejected). As a result of these filter-feeding activities, dreissenids divert food resources from other food web components, such as invertebrates inhabiting both the water (zooplankton) and bottom sediments (benthos). On average, dreissenid colonization in a given lake or river has been accompanied by a greater than 30 percent increase in water clarity, a greater than 35 percent decline in phytoplankton biomass, and a greater than 40 percent decline in zooplankton (Higgins and Vander Zanden, 2010). Impacts on benthic invertebrate communities have varied depending on feeding mode and habitat of the particular species (Ward and Ricciardi, 2007). Species able to feed on dreissenid biodeposits (i.e., feces and pseudofeces) or positively influenced by greater habitat complexity offered by mussel beds (predation refuge) have increased in abundance. On the other hand, species that filter feed or depend on fresh sedimentary inputs of phytoplankton have declined. Changes in the abundance and composition of pelagic and benthic invertebrate communities ultimately affect the fish community because fish rely on these invertebrate groups as a source of food. Fish impacts depend on habitat, diets, and population state of the particular species (Vanderploeg and others, 2002; Strayer and others, 2004; Mohr and Nalepa, 2005), but dreissenid impacts on the fish community are now becoming more apparent as quagga mussels increase and expand into new habitats.

The re-direction of energy and nutrient flow by dreissenids has been broadly termed the “nearshore shunt” (Hecky and others, 2004). In brief, dreissenids have shifted nutrient resources from pelagic to benthic zones and have focused them in nearshore relative to offshore regions. As an example, phosphorus concentrations in nearshore waters of the Great Lakes are increasing despite stable or decreased external loads (Higgins, Malkin, and others, 2008). The likely reason is that phosphorus associated with particles is being sequestered, mineralized, and excreted in soluble form by dreissenids found at high densities in nearshore areas. In addition, phosphorus associated with dreissenid feces and pseudofeces is being deposited on the bottom, enriching bottom sediments and near-bottom waters. The combination of greater light penetration resulting from increased water clarity and the greater availability of phosphorus has led to increased growth of benthic algae and macrophytes (Lowe and Pillsbury, 1995; Skubinna and others, 1995). In particular, there has been resurgence in the nuisance benthic algae *Cladophora* in the Great Lakes since dreissenids became established (Higgins, Malkin, and others, 2008). Overall, nearshore regions have become more nutrient enriched and benthic productivity has increased, whereas offshore regions have become more nutrient starved and pelagic productivity has declined.

Dreissenids have also been implicated in the resurgence of cyanobacteria blooms in some bays and basins of the Great Lakes and in some inland lakes (Vanderploeg and others, 2001; Knoll and others, 2008). Blooms were common in the Great Lakes before the mid-1970s as a result of excessive nutrient input (phosphorus), primarily from point-source loads. After nutrient abatement programs were initiated in the mid-1970s, cyanobacteria blooms were rarely observed. Blooms began to reappear just after dreissenid colonization in the early 1990s, and now extensive blooms occur almost every summer (Vanderploeg and others, 2002). Cyanobacteria are a group of phytoplankton associated with taste and odor problems in drinking water, and some species/strains produce toxins that are detrimental to human, animal, and ecosystem health. The most frequent bloomer in the Great Lakes is *Microcystis*, a taxa that produces microcystin, a hepatotoxin. During bloom events, microcystin concentrations often exceed the World Health Organization limit for drinking water of 1·µg L^{-1} (Dyble and others, 2008). Dreissenids promote cyanobacteria through the process of selective rejection (Vanderploeg and others, 2001). As dreissenids indiscriminately filter phytoplankton from the water, they reject toxic strains of cyanobacteria as pseudofeces because of unpalatable taste or size. The rejected cells are still viable, and when the pseudofeces is resuspended in the water during turbulent mixing events, these cells grow rapidly because of diminished nutrient competition from phytoplankton that are filtered and assimilated by the mussels. An increase in cyanobacteria has not occurred in all water bodies invaded by dreissenids (i.e., Hudson River, some Dutch lakes). Some factors influencing whether or not a bloom occurs include the fraction of water column filtered by dreissenids, relative taste/size of the particular strain of cyanobacteria, and nutrient and light regimes (Vanderploeg and others, 2001).

A Case History: Quagga Mussels in Lake Michigan

The quagga mussel was first found in northern Lake Michigan in 1997 and within 5 years had spread throughout the lake (fig. 2) (Nalepa and others, 2001, 2009). Regular monitoring of populations at 40 sites of various water depths in the south indicated that abundances at sites shallower than 50 m increased rapidly after 2002 and began to peak by 2007 (Nalepa and others, 2009). Abundances at sites deeper than 50 m did not begin to increase until 2005 and were still increasing as of 2007. The quagga population in the main basin of the lake consists entirely of the profunda phenotype. While zebra mussels were present in the lake since 1989 and ecological impacts were long evident, several important aspects of the quagga mussel expansion were relevant in effecting additional ecological changes. First, in shallow regions (<50 m) the quagga mussel population attained mean densities that were seven times greater than mean densities ever achieved by zebra mussels. Second, the quagga population is presently increasing in the deeper, offshore regions where zebra mussels were never found. The net result is that overall dreissenid biomass (wet weight; tissue and shell) in the lake has increased dramatically since the expansion of quagga mussels. Based on lakewide sampling, dreissenid biomass increased from 2.6·g m^{-2} in 1994/1995 when only zebra mussels were present to 188·g m^{-2} in 2005 when quagga mussels became dominant (Nalepa and others, 2009). Estimated lakewide biomass increased to 529·g m^{-2} in 2007, which is a 203-fold increase in just 12 years.

The proliferation of quagga mussels in Lake Michigan has led to many ecological changes that were not evident when only zebra mussels were present in the lake. Spring chlorophyll concentrations have declined fourfold, recently falling to below 1 microgram per liter (µg/L) (G. Fahnenstiel, National Oceanic and Atmospheric Administration, unpub. data, 2009). Chlorophyll is an indicator of phytoplankton biomass, and levels usually peak in the spring because of an increase in diatoms. Diatoms are a phytoplankton group rich in essential nutrients and thus are an important food source for many pelagic and benthic invertebrates. The decline in the spring diatom bloom can be linked to the filtering activities of quagga mussels. During unstratified conditions in the spring, the water column is well mixed, and bottom-dwelling mussels in deep areas (below the thermocline) have access to phytoplankton found throughout the water column. Further, because mussels occur at the sediment surface, they can filter out diatoms before this rich food settles to the bottom and is available to sediment-dwelling organisms.

Since dreissenids became established in Lake Michigan, water clarity in nearshore areas has increased twofold (Bootsma and others, 2007), and similar increases have been noted in offshore areas since quagga mussels became abundant (G. Fahnenstiel, National Oceanic and Atmospheric Administration, oral commun., 2009). Dissolved phosphorus in nearshore waters has also increased (Bootsma and others, 2007). The combination of increased light and available phosphorus has led to a proliferation of *Cladophora.* Biomass of this nuisance algae has increased nearly threefold along the rocky western shoreline between the pre-mussel period and 2006, with most of the increase occurring since quagga mussels became abundant (Bootsma and others, 2007). In late

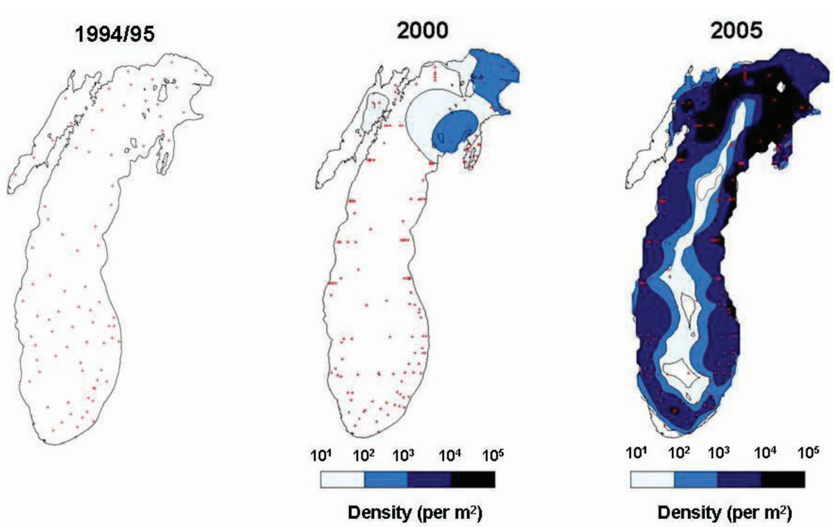

Figure 2. Mean density (number per square meter) of quagga mussels in Lake Michigan in 1994/1995, 2000, and 2005. The small red crosses are the locations of sampling sites.

summer when temperatures increase, the *Cladophora* dies, floats to the surface, and gets washed up on shoreline beaches. The decaying algae harbors bacteria, creates a foul smell, and severely limits beach use (*http://dnr.wi.gov/org/water/greatlakes/cladophora/*).

Whereas zooplankton biomass in the lake has declined since quagga mussels became abundant (S. Pothoven and H. Vanderploeg, National Oceanic and Atmospheric Administration, unpub. data, 2009), perhaps the most dramatic change in the invertebrate component of the lower food web has been the loss of the native amphipod *Diporeia* spp. (Nalepa and others, 2009). This organism once dominated benthic biomass in Lake Michigan (>70 percent) and was a keystone species in the cycling of energy between lower and upper trophic levels. *Diporeia* lives in the top few centimeters of sediment and feeds on organic material settled from the water column, being particularly dependent upon the spring settling of diatoms as a food source. Declines in *Diporeia* were first observed in the early 1990s, just a few years after zebra mussels became established in the lake in 1989. As zebra mussels spread, declines in *Diporeia* became more extensive and by 2000 *Diporeia* had disappeared from large areas of the lake shallower than 50 m in water depth. The decline of *Diporeia* extended to lake areas greater than 50 m once quagga mussels expanded to these depths. This amphipod is high in lipids and a rich food source for fish, and studies have shown that its decline is having a negative impact on the fish community. For one, growth and condition of lake whitefish (*Coregonus clupeaformis*), an important commercial species that feeds heavily on *Diporeia*, have decreased 27 percent since the mid-1990s (Pothoven and others, 2001). Also, the abundance and energy density of many prey fish have declined where *Diporeia* was no longer present (Hondorp and others, 2005). Prey fish (alewife, *Alosa pseudoharengus*; sculpin, *Cottus* spp.; bloater, *Coregonus hoyi*; etc.) serve as prey for the larger piscivores (salmon, trout; *Oncorhynchus* spp.) within the lake. Lakewide biomass (wet weight) of prey fish declined from 91 kilotonnes in 2005 to 31 kilotonnes in 2007, which is down from 450 kilotonnes in 1989 (C. Madenjian, U.S. Geological Survey, oral commun., 2009). Most of the recent decline can be attributed to the collapse of the alewife population, which is a pelagic planktivore, but at times feeds on *Diporeia*. In contrast, lakewide biomass (wet weight) of quagga mussels was 36 kilotonnes in 2005 and estimated at 113 kilotonnes in 2007 (Nalepa and others, 2009). Thus, total biomass of the quagga mussel population in the lake is now estimated to be about 3.8 times greater than total prey fish biomass. Mussel mass represents a major energy sink and a disruption of energy flow through the food web. Some fish species are feeding on quagga mussels, but the problem with fish switching from food sources like *Diporeia* to mussels lies in the ingestion of the mussel shell, which comprises more than 80 percent of the total dry mass in quaggas. The shell offers little nutrition to the fish and represents an energetic cost to the fish in terms of handling, ingestion, retention time, and egestion. Further, there is an energetic cost to the mussel to produce the shell. Thus, energy is lost to the food web when the shell is ingested and also lost when the shell is produced.

Implications to the Colorado River System

It is difficult to predict all the relevant ecological changes that will result from the quagga mussel invasion of the Colorado River system. In the Great Lakes, some changes, such as increased water clarity, decreased phytoplankton biomass, and an increase in benthic productivity, could have been predicted from the European experience. Yet other important impacts, such as the return of cyanobacteria blooms and the loss of the native amphipod *Diporeia,* were unexpected. Ecological impacts are a function of mussel densities, and since mussels are proliferating in large reservoirs of the Colorado River system (i.e., Lake Mead; Moore and others, 2009), some changes in these reservoirs might be expected. In contrast, high levels of suspended sediment and high inorganic:organic particle ratios will limit, if not prevent, mussel expansion in the mainstem portions of the river (Kennedy, 2007). Yet even if mussels do not proliferate in the mainstem, some ecosystem changes may occur. The mainstem river is coupled to upstream reservoirs, and mussel-mediated changes in the water quality (i.e., dissolved nutrients, phytoplankton, zooplankton) of such reservoirs as Lake Powell and Lake Mead will likely impact food web structure or trophic linkages in the downstream riverine ecosystem. Concerns over increased algal blooms in the reservoirs are real, since blooms of some species have already occurred before the quagga mussel invasion (*Pyramichlamys dissecta* and *Cylindrospermopsis* r*aciborski*), and *Microcystis,* which now regularly blooms in some shallow, warm regions of the Great Lakes, is a component of phytoplankton communities in these reservoirs (St. Amand and others, 2009). Most certainly, productivity will shift from the pelagic to the benthic region, and an increase in biomass of many benthic invertebrates will likely result. Because bottom habitat drives the food web in some Colorado River reservoirs (Umek and others, 2009), this shift may benefit many bottom-feeding fish species, including some of the natives (i.e., razorback sucker, *Xyrauchen texanus*). On the other hand, the threadfin shad (*Dorosoma petenense*), a pelagic planktivore and a forage base for some sport fish, may be adversely affected much like the alewife was affected in the Great Lakes.

Currently, the quagga mussel population is expanding in the Colorado River system, but eventually the population will stabilize as abundances reach equilibrium with the surrounding environment. During this process, both acute and chronic ecological impacts will be realized as ecosystem components respond at different rates, leading to outcomes that can be both interactive and cumulative over time (Strayer and others, 2006). It may take many years, but eventually the ecosystem will reach a new, different steady state. Monitoring of key ecosystem parameters during this process is essential in understanding interactions that form the basis for a new paradigm of resource management.

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Economic Values for National Park System Resources Within the Colorado River Watershed

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Abstract

This paper provides a literature review of economic valuation studies of recreational use and other ecosystem services provided by National Park Service (NPS) resources in the Colorado River watershed. These parks are nationally important recreation and conservation resources and in 2005 had a total of about 10.5 million recreational visits to reservoir and river sites. Existing economic valuation studies can be grouped into two main areas: (1) estimates of direct recreational use values and (2) passive use values, including existence and bequest motives. With respect to recreation values, one study, now more than 20 years old was conducted for floating in Glen and Grand Canyons and for fishing in the Lees Ferry river section. No survey-based estimates of visitor by willingness to pay (WTP) have been conducted for Lake Mead, Curecanti, Dinosaur, or Canyonlands park units, which support over 80 percent of water-related visits to NPS Colorado River Basin park units. With respect to passive use values, a second study measured national values for improved conditions to Grand Canyon endangered fish and associated river ecosystems through modified flow regimes. The available economic valuation estimates for these resources are not comprehensive, do not consider Tribal perspectives, and are generally dated.

Introduction

For purposes of planning and participation in water-resource allocation decisions, the National Park Service (NPS) needs to know the economic values of the resources it manages within park units along the Colorado River system (including major tributaries). At present, the NPS does not have recent or comprehensive values to represent their current water-related activities within these Colorado River park units. The purpose of this paper is to summarize economic estimates relevant to these resources in the Colorado River Basin.

The NPS units located along the Colorado River and its tributaries (table 1) accounted for nearly 20 million recreational visits in 2005. Of these, more than 10 million visits were directly linked to water-based activities and, thus, were dependent to some extent on water levels within the Colorado River system.

Existing economic studies of NPS Colorado River park units and related economics literature can be grouped into two main areas: (1) estimates of direct recreational use values (net economic benefits as measured willingness to pay (WTP) over and above trip costs) and (2) passive use values, which include the benefits individuals derive from simply knowing that a unique natural environment or species exists even if the individual does not visit or see the resource. These are sometimes called existence and bequest values.

The impacts of water levels on recreational values occur through two influences: water levels influence the quality of the recreational trip and accordingly the WTP per trip, and water levels affect visitor participation. For example, river-sections use drops to zero at very low (impassable or quite dangerous) flow levels and to near zero in extreme floods. Participation and trip quality are generally optimized at intermediate flow levels. By contrast, use on some reservoirs increases continuously with reservoir elevations and is maximized at full pool. By identifying the relation of participation and value to water levels, it is possible to estimate the marginal value of water associated with recreational use. This typically is expressed in terms of dollars per acre-foot (af) of storage on reservoirs and dollars per cubic foot per second (ft^3/s) or per acre-foot per year on rivers.

Recreational visitation to NPS units within the Colorado River system is associated with significant economic values. These values generally are described within two distinct accounting frameworks: net economic value and regional economic impact. The first measure of value, net economic value, describes both the direct use value associated with park visitation and passive use value within the context of a benefit/cost framework. The second framework, regional economic impact analysis, describes the impact of visitor spending on a defined local or regional economic area. The focus in this paper is on net economic values.

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Table 1. National Park Service Colorado River units and associated visitation characteristics.

[NP, national park; NRA, national recreation area; NM, national monument]

Park unit	Waters	Type of water	Total 2005 visitation[a]	Colorado River water-related 2005 visitation[a]
Arches NP	Borders Colorado River	River	781,670	negligible
Black Canyon of Gunnison NP	Gunnison River	River	180,814	46
Canyonlands NP	Colorado and Green Rivers	River	393,381	11,508
Curecanti NRA	Blue Mesa, Morrow Point, and Crystal Reservoirs	Reservoir	882,768	882,768
Dinosaur NM	Green and Yampa River	River	360,584	12,802
Glen Canyon NRA	Lake Powell Colorado River	Reservoir River	1,863,055 45,671	1,863,055 45,671
Grand Canyon NP	Colorado River	River	4,401,522	22,000[c]
Lake Mead NRA	Lake Mead Colorado below Hoover Dam Lake Mojave	Reservoir River Reservoir	7,692,438	7,692,438
Rocky Mountain NP	Headwaters of Colorado River	River	2,798,368	negligible

[a] Total 2005 recreational visitation from the National Park Service Public Use Statistics Office (*http://www2.nature.nps.gov/mpur/index.cfm*).

[b] The NRA units (Curecanti, Glen Canyon, and Lake Mead) are assumed to be entirely water-based recreation.

[c] Total float use of Grand Canyon has been relatively stable at 20,000 to 24,000 visits in recent years (Grand Canyon National Park Management Plan Final Environmental Impact Statement).

Direct Recreational Value Estimates

To date, the number of published estimates of the value of recreational visits to National Park System units nationwide is somewhat limited. Kaval and Loomis (2003) identified 11 studies that provided 49 activity-specific net economic value per activity day estimates. The activities included sightseeing, boating, picnicking, hiking, and wildlife viewing. Updating the Kaval and Loomis (2003) average estimates from 1996 dollars to 2005 dollars indicates an average value per day across all 49 observations of $53.88. The updated average that Kaval and Loomis report for the Southwest region national parks is $28.16.

To date, two major economic studies related to NPS-related uses in the Colorado River corridor have been conducted, both in the context of the Glen Canyon Dam studies. These studies had a fairly narrow geographic scope (just the river corridor through Glen Canyon and Grand Canyon). Both of these earlier studies focused on identifying marginal values, in the sense of measuring the change in value associated with moving from the base case or no action alternative in the environmental impact statement (EIS) planning process for Glen Canyon Dam to some specific alternative. By having these marginal values, it was possible in the EIS process to compare the tradeoffs of alternative uses, including recreation and power generation values.

The first Glen Canyon Dam economic study focused on recreational use and was undertaken by Bishop and others (1987). The second study focused on passive uses and will be discussed in a following section. The Bishop study was conducted as part of the Glen Canyon Environmental Studies efforts during 1984 and 1985. The overall goal of the Bishop study was to evaluate the impacts of alternative flow release patterns from Glen Canyon Dam on whitewater boating, day-use rafting, and fishing on the Colorado River below the dam. The authors of the 1987 study conducted a several-phase investigation in order to address their goal. First, user surveys were conducted to identify the attributes of fishing and floating trips that provided value to users. A second, more comprehensive contingent valuation (CV) survey of river users addressed potential changes in resource values associated with alternative flow release patterns. While Bishop and others (1987) found no statistically significant relation between flow levels and values associated with day-use floating below Glen Canyon Dam, they found a strong link between flows and both fishing and whitewater boating values. The study found that for whitewater rafters, relatively constant flows between 20,000 and 25,000 ft^3/s yielded the highest satisfaction and associated values. For anglers, a similarly constant flow regime in the 10,000 ft^3/s range yielded improved recreational trip values over existing flow regimes (Bishop and others, 1987). As an example of the range in values, the net economic value per trip (WTP over and above trip costs) for commercial whitewater boaters was estimated at $176 per trip ($319 in 2005 dollars) at a 5,000 ft^3/s flow level and rose to a maximum value of $602 per trip ($1,093 in 2005 dollars) at higher flows.

With respect to the significance of recreation use values in the Glen Canyon Dam operations context, the influence of flows on recreational values is primarily through the effect on the quality of the trip. There is excess demand for river recreation below Glen Canyon Dam (use is basically always at the permitted capacity in the main season). This limits the potential magnitude of changes in use values in response to changing flow regimes. By contrast, the nonuse value effects are quite large relative to the foregone power revenues for the alternatives examined and have allocative significance, as noted below.

For reservoir recreation within Colorado River Basin NPS units, Douglas and Johnson (2004) used 1997 survey responses for Lake Powell recreational visitors to estimate a travel cost model of WTP for trips to the reservoir. The authors estimated that per visit consumer surplus for Lake Powell visits ranged from $71 ($86 in 2005 dollars, based on a log-log model specification) to $159 ($174 in 2005 dollars, based on an inverse-price model specification).

Douglas and Harpman (2004) report dichotomous choice contingent valuation results for the same 1997 survey dataset as Douglas and Johnson (2004). The dichotomous choice question valued improvements in angler harvest, water quality (reduced beach closures relative to the 1991–1996 period), and archaeological site protection and restoration. The payment vehicle was the season pass. A current trip valuation question was not included. For the authors' preferred model, household benefits across the summer ranged from $396 (1997 dollars) to $1,100 per household per year. On a per visit basis, this implied a range in value of $9 to $39 per visit ($11 to $47 in 2005 dollars). It appears from the paper that this is just the incremental value of the improved trip. The value, particularly for the archaeological site scenario, may include passive use as well as recreational use value.

Martin and others (1980) estimated net economic value (NEV) per trip for anglers at Lake Mead. A zonal travel cost model was estimated for this warmwater fishery on data collected on anglers between July 1978 and June 1979. During this period, there were an estimated 1.3 million individual fishing days of use at Lake Mead, mostly targeting striped (*Morone saxatilis*) and largemouth bass (*Micropterus salmoides*). Estimated mean net benefits per individual fishing day were $45 to $61 ($126 to $174 in 2005 dollars), depending on the specification of the model. Martin and others (1980) also report angler expenditure per day with a mean value of $43 ($122 in 2005 dollars).

In 1998, visitors to two Colorado River NPS units (Glen Canyon National Recreation Area (NRA) and Grand Canyon National Park (NP)) were surveyed within the context of a study of visitor attitudes about the NPS Fee Demonstration Project. In addition to the survey questions related to the fee program, the surveys included a dichotomous choice WTP question designed to elicit per-trip NEV responses. These NEV responses were not part of the Fee Demonstration Program study objectives, and thus an analysis of the responses was not included within the study report (Duffield and others, 1999). A subsequent analysis of these responses indicates that for park visitors who said that visiting the units was the primary purpose of their trip away from home, visitors to Glen Canyon NRA have a median NEV per party trip of $109 ($157 in 2005 dollars). Visitors to Grand Canyon NP had a median NEV per party trip of $132 ($190 in 2005 dollars; table 2).

Just as there is an economic literature on in-stream flow values, there is a related literature on the effect of reservoir levels on recreation. Huszar and others (1999) developed and estimated a joint model of fish catch and recreation demand, both of which depend on water levels, to assess the losses and gains from water-level changes tied to events in the Humboldt River Basin of northern Nevada. Additionally, Eiswerth and others (2000) estimated recreation values for preventing a decline in water levels at, and even the total loss of, a large western lake that is drying up.

Table 2. Summary of literature and estimates of Colorado River National Park Service units direct recreational value estimates.

[NEV, net economic value; NPS, National Park Service; NRA, NRA, national recreation area; NP, national park; CV, contingent valuation]

Study	Description	NEV estimate	NEV estimate (2005 dollars)
Kaval and Loomis (2003)	Survey of literature – 11 studies providing 49 estimates of activity values within NPS units	$53.88 per visitor day (2005 dollars)	$54
Kaval and Loomis (2003)	Survey of literature – Estimates only for Southwest U.S. NPS units	$28.16 per visitor day (2005 dollars)	$28
Bishop and others (1987)	Study of values of Grand Canyon float boaters	$176–$602 per trip depending on river flow level (1985 dollars)	$319–$1,093
Douglas and Johnson (2004)	Travel cost study of Lake Powell recreationists	$70.84–$159.35 per visit consumer surplus (1997 dollars)	$86–$194
Martin and others (1980)	Study of Lake Mead recreation values	$44.63–$61.44 per angler day (1978–79 dollars)	$126–$174
Duffield and Neher (1999)[a]	Visitor survey of Glen Canyon NRA and Grand Canyon NP visitors. Estimates of per trip NEV based on dichotomous choice CV survey questions.	Glen Canyon NRA – $109 per party trip Grand Canyon NP – $132 per party trip (1988 dollars)	Glen Canyon – $157 Grand Canyon – $190
Douglas and Harpman (2004)	Dichotomous choice CV survey of improved trip quality scenarios (angler harvest, water quality)	$8.63–$38.92 per visit[b] (1997 dollars)	$11–$47

[a] Consumer surplus estimates were derived in an analysis subsequent to the preparation of the primary report on visitor attitudes regarding park fee increases.

[b] Not total value of current trip, but incremental values due to improvement.

Passive Value Estimates

Passive use values are an indication of the national significance of NPS resources. These values are associated with knowing that these resources are in a viable condition and with wanting future generations to also be able to enjoy this heritage.

These motives for nonuse values were first described by Weisbrod (1964) and Krutilla (1967) as existence and bequest values. Existence values can be derived from merely knowing that a given natural environment or population exists in a viable condition. For example, if there was a proposal to dam the Grand Canyon, many individuals could experience a real loss, even though they may have no expectation of ever personally visiting the river corridor through Grand Canyon. Other individuals might similarly suffer a loss if the grizzly bear were to become extinct in the Northern Rockies, even though those individuals may have no desire to directly encounter a grizzly. Bequest motives are derived from one's desire to provide for future benefit to children and others in future generations. There may be many possible motives for nonuse values, and these motives may or may not be mutually exclusive.

The methods used to estimate nonuse values are so-called stated preference methods (including contingent valuation and conjoint analysis (National Research Council, 2005)). Individuals are asked in a survey to indicate directly the value they place on nonuse services or resources. These methods are generally accepted and applied in policy analysis, as evidenced by their endorsement as a recommended method in regulatory guidelines. These include the Department of the Interior regulations for implementing the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (or CERCLA, at 43 CFR part 11) and in the U.S. Environmental Protection Agency's "Guidelines for Preparing Economic Analysis" (2000).

These methods have now been widely applied and reported in the published economics literature. When contingent valuation as a recommended approach was challenged in court (Ohio v. United States Department of Interior, 880 F.2d 432, 474 (D.C. Cir. 1989)), the court affirmed its usefulness for natural resource damage assessment. Additionally, in the context of the development of related regulations for implementation of the Oil Pollution Act of 1990 by the National Oceanic and Atmospheric Administration (NOAA), the use of contingent valuation was reviewed by a panel that included several Nobel laureates in economics. The panel endorsed the use of contingent valuation in a litigation setting, subject to the caveat that studies meet certain recommended guidelines (Arrow and others, 1993).

The National Research Council (2005, p. 6) offers the specific guidance that: "Economic valuation of changes in ecosystem services should be based on the comprehensive definition embodied in the TEV [total economic value] framework; both use and non-use values should be estimated."

Table 3 provides a summary of passive use studies relevant to water-related NPS resources, including lakes and rivers (in-stream flows and endangered fisheries). These selected studies are generally in the Southwest or intermountain West. All of the studies use stated preference methods, such as contingent valuation. Data are collected through surveys, generally of a resident population in a given geographic area. The choice of geographic area should correspond to the "market area" for the passive use service; that is to say, an area big enough to include most people expected to hold passive use values for the resource at issue. This area may be small for a county or city park of only local historical significance, but possibly national in scope for nationally significant resources such as Grand Canyon or Yellowstone.

Key characteristics of the studies summarized in table 3 include: (1) the resource service being valued, generally a change such as increased lake elevations or populations of an endangered fish, (2) the payment mechanism (e.g., increase in monthly water bill, increase in annual taxes, a one-time donation, etc.), (3) the population surveyed, and (4) the estimated values.

The previous estimates of passive use values for Colorado River park units have all been for Grand Canyon National Park resources including visibility, river flow-related habitat, and wilderness. The first such studies were focused on visibility impacts of the Navajo Generating Station and include Randall and Stoll (1983), Schulze and others (1983), and Hoehn (1991). These and other studies eventually led the U.S. Environmental Protection Agency on October 3, 1991, to issue a regulation requiring the Navajo Generating Station coal-fired powerplant to reduce sulfur emissions. In a 1990 study, the annual benefits of achieving 90-percent emission control was estimated to be between $130 and $150 million annually, compared to the estimated costs of this control of $89.6 million (1990 dollars). Deck (1997) describes both the benefit and cost studies that were the basis of this decision.

The only passive use study relating to water resources is the Welsh and others (1995) contingent valuation study undertaken as part of the Glen Canyon Dam EIS process. Harpman and others (1995) describe the importance of nonuse economic values as a policy analysis tool with specific reference to water-influenced resources in Grand Canyon.

In the Welsh and others (1995) study, contingent valuation methods were applied to estimate WTP to improve native vegetation, native fishes, game fish (such as trout), river recreation, and cultural sites in Glen Canyon NRA downstream from Glen Canyon Dam and in Grand Canyon NP (Welsh and others, 1995). This study utilized a population survey of two groups, Western U.S. households within the marketing area for Glen Canyon power and households in the entire United States. Respondents were asked questions of their WTP either increased electric power rates (Western U.S. sample) or higher taxes (national sample) to reduce flow fluctuations from Glen Canyon Dam to protect wildlife, beaches, and cultural sites. The study results (table 4) show that the "steady flow" scenario that was presented as being most beneficial to resource protection also had the highest associated values.

While the nonuse study for the Colorado River corridor in Grand Canyon NP (Welsh and others, 1995) was completed too late to be fully utilized in the 1995 EIS (U.S. Department of the Interior, 1995), the study findings did have an influence on the EIS outcome. The National Research Council panel that reviewed the Glen Canyon Environmental Studies (GCES) commented favorably on the study. Their report stated: "The GCES nonuse value studies are one of the most comprehensive efforts to date to measure nonuse values and apply the results to policy decisions. … While not completed in time to be reported in the final EIS, the nonuse value results are an important contribution of GCES and deserve full attention as decisions are made regarding dam operations" (National Research Council, 1996, p. 135).

The estimates of the Welsh and others (1995) contingent valuation study are conservative in that Welsh chose in his methodology to count only those "yes" respondents that also indicated they would "definitely yes" pay the stated amount. The use of only "definitely yes" responses has been shown in other CV validity studies to provide a valid estimate of actual WTP. Champ and others (1997) also found this result in assessing the nonuser social value of a program at Grand Canyon NP to remove compacted dirt roads on the North Rim to create a wilderness setting. A more recent study by Champ and her colleagues that is focused on riparian ecosystems (Duffield and others, 2006) also found that CV responses with a self-rated high certainty of actual contribution corresponded well with actual levels of cash donations. The application in this case was for purchases of in-stream flow rights on dewatered Montana streams, primarily to benefit riparian ecosystems, fishery species of special concern, and other wild fish.

Table 3. Empirical estimates of passive use values for water-related resources.

[WTP, willingness to pay]

Author	Survey year	Payment vehicle	Resource	Survey region	Value estimate
Lakes					
Sutherland and Walsh (1985)	1981	Annual payment into trust fund (per household)	Flathead Lake and River	Montana households	\$19.99 existence \$26.48 bequest
Loomis (1989)	1986–87	Monthly water bill increase (per household)	Mono Lake	California households and Mono Lake visitors	\$4.12–\$5.89 (households) \$9.97–\$12.15 (visitors)
Rivers					
Hanemann and others (1991)	1989	Annual household WTP (per household)	San Joaquin Valley	California	\$181
Duffield and Patterson (1991)	1990	One-time donation to trust fund (per person)	In-stream flows in Montana trout streams	Montana resident and nonresident fishing license holders	\$2.24–\$4.64 (residents) \$12.60–\$17.36 (nonresidents)
Welsh and others (1995)	1994	Increased electric power rates or increased taxes (per household)	Colorado River riparian ecosystem	Western U.S. households and all U.S. households	\$17.74–\$26.91 (U.S. sample) \$29.05–\$38.02 (Western sample)
Brown and Duffield (1995)	1988	Annual WTP into trust fund (per household)	Bitterroot, Bighole, Clark Fork, Gallatin and Smith Rivers	Phone directory listings for major Montana cities and Spokane, WA	\$6.70 (one river) \$12.43 (five rivers)
Berrens and others (1996)	1995	Annual donation to trust fund for 5 years (per household)	Middle Rio Grande, Gila, Pecos, Rio Grande, and San Juan Rivers	New Mexico residents	\$28.73–\$89.68
Loomis (1996)	1994–95	Additional taxes for 10 years (per household)	Elwah River system	Clallam County, WA; rest of Washington; and rest of U.S. households	Clallam \$59 Rest of Washington \$73 Rest of United States \$68
Berrens and others (1998)	1995–96	Annual payment into trust fund (per household)	Major rivers in New Mexico	New Mexico residents	\$74
Berrens and others (2000)	1995–96	Annual payment into trust fund (per household)	Gila, Pecos, Rio Grande, and San Juan Rivers	New Mexico residents	\$55
Duffield and others (2006)	2005	One-time donation to trust fund (per person)	Small Montana trout streams	Resident and nonresident Montana fishing license holders	\$5.73 (residents) \$31.07 (nonresidents)

Table 4. Welsh and others (1995) estimates of nonuse values for three Glen Canyon flow scenarios (2005 dollars).

Flow scenario	National sample		Western U.S. sample	
	Per household	Annual value (millions)	Per household	Annual value (millions)
Moderate fluctuations	$17.74	2,791	$29.05	79
Low fluctuations	$26.19	4,386	$28.25	80
Steady flow	$26.91	4,474	$38.02	107

Implications for Management

Table 5 presents a summary of the availability of existing data and estimates for the economic impacts of water flows and levels on Colorado River Basin park units. The basic finding is that existing studies are not adequate to support economic analysis, such as benefit-cost evaluation of alternative water allocations, for most units. There is a need to conduct additional economic research focused on water resource uses in the region.

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Table 5. Is sufficient information available to support benefit-cost evaluations?

[NEV, net economic value; NRA, national recreation area; NP, national park; NM, national monument; n/a, not applicable]

Park unit	Produce direct use total value estimates for water-based visitation?	Produce passive use estimates?	Estimate marginal impacts of water level on NEV?
Glen Canyon NRA			
Lake Powell	Yes	No	Yes
Colorado River (Glen-Lees)	No	No	No
Lake Mead NRA			
Lake Mead	No	No	No
Lake Mojave	No	No	No
Curecanti NRA	No	No	No
Grand Canyon NP			
Grand Canyon Float	Yes	Yes	Yes
Dinosaur NM			
Yampa and Green River	No	No	No
Canyonlands NP			
Cataract Canyon	No	No	No
Black Canyon NP	No	No	No
Arches NP and Rocky Mountain NP	n/a	n/a	n/a

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Confluence of Values: The Role of Science and Native Americans in the Glen Canyon Dam Adaptive Management Program

By Kurt E. Dongoske,[1] Loretta Jackson-Kelly,[2] and Charley Bulletts[3]

We don't see things as they are, we see things as we are.

—Anaïs Nin

Abstract

Grand Canyon and the Colorado River are important places on the landscape for many Native American Tribes. The Glen Canyon Dam Adaptive Management Program (GCDAMP) is designed to employ science as a means for gathering, analyzing, and disseminating information on the condition of resources. A Western science perspective dominates this program with recognition of Native American traditional perspectives as a valued component. Analogous to a confluence of rivers, Native American traditional perspectives were initially envisioned as enhancing the Western science approach by creating a more holistic understanding of this valued ecosystem; however, this integration has not been realized. Identified barriers to effective participation by Native American stakeholders are vast cultural differences that express themselves in complex sociocultural scenarios such as conflict resolution discourse and a lack of insight on how to incorporate Native American values into the program. Also explored is the use of "science" as a sociopolitical tool to validate authoritative roles that have had the unintended effect of further disenfranchising Native Americans through the promotion of colonialist attitudes. Solutions to these barriers are offered to advance a more effective and inclusive participation of Native American stakeholders in this program. Finally, drawing from the social sciences, a reflexive approach to the entire GCDAMP is advocated.

Introduction

Grand Canyon and the Colorado River are important places on the landscape that are central to the traditional values and histories of many Native American Tribes. During the development of the Glen Canyon Dam Environmental Impact Statement (GCDEIS), between 1991 and 1995, the Bureau of Reclamation acknowledged the importance of integrating Native American perspectives and values into the environmental analysis equation by providing five Tribes (Hopi, Hualapai, Navajo, Southern Paiute Consortium, and the Pueblo of Zuni) with cooperating agency status. The Glen Canyon Dam Adaptive Management Program (GCDAMP) was created in 1997 as a result of the Record of Decision for the GCDEIS and is designed to employ science as a means for gathering, analyzing, and disseminating information on the condition of natural and cultural resources to the appropriate managers.

A critical examination of the past 10 years of Tribal participation in the GCDAMP reveals a failure of the program to effectively integrate Native American perspectives. Our analysis and conclusions of the GCDAMP are based on direct participation as Tribal representatives. Our participation as Tribal stakeholder representatives began during the development of the Glen Canyon Dam Environmental Impact Statement and continues in the GCDAMP at the time this paper

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was written.[4] In this paper, we identify several barriers to effective participation by Native American stakeholders in the GCDAMP. Identified barriers include vast cultural differences among stakeholders that express themselves in complex sociocultural scenarios and a lack of insight by program managers and scientists on ways to affirm and respond to Tribal cultural concerns.

Also explored in this paper is the concept of Western science as it exists within the GCDAMP and how science is employed as a sociopolitical tool to validate authoritative roles within the program. The heavy reliance on Western science within the program is demonstrated to have had the unintended effect of promoting the disenfranchisement of the participating Tribes from the GCDAMP through the continuation of colonialist attitudes. Solutions to these identified barriers are proposed to advance a more effective and inclusive participation by the Native American stakeholders in this program. Finally, drawing from the social sciences, a reflexive approach to the entire GCDAMP is advocated.

Grand Canyon as Cultural Landscape

Grand Canyon and Colorado River are culturally important and represent astounding aspects of the landscape for most Americans and for individuals from across the globe. Grand Canyon has been classified as one of the seven natural wonders of the world and is also recognized as a World Heritage Site. Grand Canyon embodies a powerful and inspiring landscape that overwhelms the human senses.

In the late 1800s, Clarence Dutton (as quoted in Worster, 2001, p. 326) described Grand Canyon most appropriately when he stated that Grand Canyon is:

> More than simply two walls rising from the river, the Grand Canyon is a complex architectural wonderland some ten to twelve miles in breadth at its widest point. What nature has done here is precisely the work of an architect: chiseling, sculpting, cutting out large amphitheatres, naves, arches, and columns, leaving walls, spandrels, lintels. Hundreds of these mighty structures, miles in length, and thousands of feet in height, rear their majestic heads out of the abyss, displaying their richly-molded plinths and friezes, thrusting out their gables, wing-walls, buttresses, and pilasters, and recessed with alcoves and panels.

Nearly 100 years ago, in 1919, the beauty, diversity, and splendor of Grand Canyon and its importance as a natural and cultural jewel of the American people were formally recognized when Congress established Grand Canyon National Park. Since that time, many policies and programs have been developed to protect and preserve the diversity and unique qualities of Grand Canyon and the Colorado River ecosystem, and the GCDAMP associated with the operations of Glen Canyon Dam has become central to those efforts.

Although Grand Canyon and the Colorado River are recognized as culturally important to the greater American society, the Native American connection to this landscape existed and persevered for hundreds, if not thousands, of years before the arrival of Europeans (Schwartz, 1965; Euler and Taylor, 1966; Euler and Chandler, 1978; Schwartz and others, 1979, 1980, 1981). Native Americans relation to this vast place and their resultant knowledge of this landscape and environment are the product of a long temporal and intimate association, including having to depend upon and survive within this landscape. Given the brevity of this article, a full account of the cultural importance of Grand Canyon and the Colorado River to each of the Native American Tribes that participates in the GCDAMP is not possible. Presented below is a brief general summary of Grand Canyon's importance to participating Native Americans.

Grand Canyon and the Native American Tribes

According to multiple Tribal creation accounts, Grand Canyon is the place of origin and emergence into this the fourth world. The Colorado River, the canyons, the land, the middle of the river, and the springs, seeps, and tributaries to the river are essential to the well being, survival, and the collective and individual identities of many of the participating Tribes. As such, these Tribes are entrusted with the responsibility to care for the natural environment and the resources contained within Grand Canyon and their traditional homelands (Hualapai Tribe and Stevens, 1998; Austin and others, 2007).

[4] The primary author began his involvement in this program in 1991 as the Hopi Tribe's representative to the cooperating agencies in the development of the Glen Canyon Dam Environmental Impact Statement (GCDEIS). He continued to represent the Hopi Tribe as their representative to the Technical Work Group and as the alternate representative to the Adaptive Management Work Group during the development and implementation of the Adaptive Management Program from 1996 to 2003. Mr. Dongoske also served as the Chair of the Technical Work Group for 5 years and is currently the Pueblo of Zuni's representative to the Technical Work Group. Ms. Jackson-Kelly represented the Hualapai Tribe during the development of the GCDEIS and has been the Hualapai Tribe's representative to the Adaptive Management Work Group since 1997. Mr. Bulletts is the Southern Paiute Consortium's representative to the Technical Work Group and the Adaptive Management Work Group, a position that he has held since 2006. Our analysis is based on direct participant observation and data acquired over an 18-year period that extends from the development of the GCDEIS to the origination and implementation of the Adaptive Management Program.

For many members of these Tribes, the Colorado River and the components of the ecosystem are regarded as living entities infused with conscious spirit. All of these elements in and around the canyons are accorded powers of observation and awareness. The Colorado River itself is regarded as an important conscious living being that has feelings and is expressive of calmness and anger. The river can offer happiness, sadness, strength, life, sustenance, and the threat of death. According to many of the Tribal beliefs, if a land and its resources are not used in an appropriate manner, the Creator will become disappointed or angry and withhold food, health, and power from humans.

Even though the current reservation lands of some of the participating Tribes are located far from Grand Canyon, these Tribes still maintain very strong ties with Grand Canyon, the Colorado River, and the Little Colorado River because of their origin and migration narratives. Traditional narratives describe the locations of shrines and sacred areas and explain why Grand Canyon is sacred. The daily prayers of many Native Americans incorporate specific locations, including sacred areas, shrines, springs, and other places of religious significance within Grand Canyon (Hart, 1995; Stoffle and others, 1995; Hualapai Tribe and Stevens, 1998). The animals, birds, rocks, sand, minerals, and water in Grand Canyon all have special meaning to the Native American people.

An Evaluation of Tribal Involvement in the Glen Canyon Dam Adaptive Management Program

Today, the Hualapai, Hopi, Navajo, and Zuni Tribes and the Southern Paiute Consortium are recognized as legitimate participants in the GCDAMP and have representatives to the Adaptive Management Work Group (AMWG) and the Technical Work Group (TWG).[5] The importance of Tribal participation in the GCDAMP was recognized by the stakeholders during the development and adoption of the GCDAMP's strategic plan. The strategic plan contains a vision and mission statement that recognizes Grand Canyon as homeland to Native American Tribes and a special place that contains properties of traditional cultural importance to Native Americans. It also acknowledges the "trust" responsibility of the Federal government to these Native American Tribes and recognizes their sovereign status and management authority.

The strategic plan also delineates 12 goals for the natural and cultural resources that are the focus of the GCDAMP. Management goal 11 addresses the preservation, protection, and management of cultural resources for the inspiration and benefit of past, present, and future generations. The recognition of past generations in this management goal is the acknowledgment by the program of the active on-going dynamic spiritual relationship contemporary Native American people have with their ancestors. Goal 11 also recognizes that the spirits of these ancestors still inhabit specific places (e.g., archaeological sites) within Grand Canyon. Accompanying management objectives 11.2 and 11.3 recognize the importance of maintaining and protecting places and resources of traditional cultural importance to Native Americans and ensuring unrestricted access to these places by Native American religious practitioners.

Management goal 12 seeks to maintain a high-quality monitoring, research, and adaptive management program that incorporates meaningful Tribal participation. Management objective 12.5 seeks to attain and maintain effective Tribal consultation through the inclusion of Tribal values and perspectives in the GCDAMP. Management objective 12.6 seeks to attain meaningful Tribal participation in management activities, research, and long-term monitoring to meet the Tribal interests to ensure that Tribal values are incorporated into the scientific activities of the GCDAMP and that Tribal interpretations are considered. Both of these management objectives are directly linked to the vision and mission statement discussed previously.

Even though the GCDAMP recognizes the importance of integrating Tribal involvement, a review of the past 10 years of research and monitoring programs indicates a rapidly declining role for the Tribes. At the inception of the GCDAMP there was significantly more Tribal involvement in research and monitoring projects than there has been during the past 5 years. For example, the Southern Paiute Consortium conducted event-specific research on dam impacts and developed a place and resource monitoring program and a corresponding educational outreach program for Paiute elders and youth (Stoffle and others, 1995; Seibert and others, 2007). The Hualapai Tribe conducted research and monitoring of places and resources between 1996 and 2003, including ethnobotanical research associated with the high-flow experiment that took place in 1996.

A sincere effort by Grand Canyon Monitoring and Research Center (GCMRC) to integrate Tribal perspectives into its terrestrial ecosystem monitoring program was put into effect between 1999 and 2002. This collaborative effort was not successful, because integration assumed that Tribal perspectives could be integrated into a framework defined and directed by the tenets of Western science. Moreover, GCMRC's inclusive intent was seriously constrained by the scientific perspective, which relies on credible, objective (i.e., numeric) data intended for model generation and a clear lack of understanding of Tribal perspectives (Austin, 2007).

[5] The Havasupai Tribe was invited to participate in the development of the Glen Canyon Dam Environmental Impact Statement and to participate in the resultant Glen Canyon Dam Adaptive Management Program; however, they declined to actively participate on the basis of cultural and financial reasons.

Sociocultural Barriers to Effective Tribal Participation

Since GCMRC's failed attempt to integrate Tribal perspectives into the terrestrial ecosystem monitoring, there have been no further efforts at Tribal integration. The absence of a defined Tribal component in the current GCDAMP science program and a progressive decline in effective Tribal voices within the AMWG and TWG are attributable to a number of sociocultural factors. We submit several sociocultural barriers that exist within the GCDAMP that are actively limiting or marginalizing effective Tribal participation.

Cultural differences in communication present at the AMWG and TWG tables act as a barrier to effective integration. Here, Tribes are expected to communicate and act in the style of Western scientists and managers even though the Tribal representatives generally do not share the same cultural and (or) educational background of the majority of stakeholder representatives. Tribal representatives have expressed their discomfort with what they call the "bigger language of English" that dominates the TWG and AMWG meeting venues. Tribal representatives have articulated how they perceive non-Native Americans as expecting them to respond to words that are not normally employed by Tribal people. Some Tribal representatives have also expressed a feeling of condescension and intimidation associated with "bigger language of English" usage (Austin, 2007).

Strongly correlated with the discomfort associated with the use of the "bigger language of English" is the argumentative nature of many of the exchanges that take place during stakeholder meetings. Tribal representatives have expressed their discomfort with the volume and acerbity with which communication takes place and the propensity for interruptions that undermine one's ability and willingness to participate (Austin, 2007). Direct confrontation is considered impolite and inappropriate behavior within the cultural contexts of the participating Tribes and constrains the Tribal representatives' willingness to "speak up more" in meetings.

Another identified sociocultural barrier to effective Tribal participation is the uncertainty of managers on how to effectively respond to concerns and values expressed by Native American Tribal representatives. This was poignantly demonstrated when several of the Tribes expressed concern about the mechanical removal of nonnative fish at the confluence of the Colorado River and Little Colorado Rivers. These Tribes expressed their disapproval of taking of life that was associated with the planned removal and destruction of thousands of rainbow (*Oncorhynchus mykiss*) and brown trout (*Salmo trutta*) as an experiment to control their predation of native fishes.

The Tribes also expressed concern about the location where this experiment was going to take place because the confluence of the Colorado and Little Colorado Rivers represents fertility and life and is considered sacred. One Tribal representative expressed his concern that the proposed mechanical removal would produce an "aura of death" over this sacred place (Leigh Kuwanwisiwma, Hopi Cultural Preservation Office, oral commun., 2002). The solution offered by managers was to provide the Hualapai Tribe with the processed trout remains to be used as fertilizer in Hualapai gardens. No solution was offered regarding the concern about the location. The conflict of cultural values expressed by the Tribe's objection to the "taking of life" associated with the implementation of this experiment was never sufficiently addressed by the program. Today, the mechanical removal is implemented regularly at the confluence of the Colorado and Little Colorado Rivers as a management action without further consideration of those expressed core Tribal concerns and values.

An additional example of sociocultural barriers underscores the deficiency of the program in effectively dealing with Tribal issues. The 2000 Protocol Evaluation Panel (PEP) of the GCDAMP cultural resources program recommended the development of a Tribal consultation plan. The panel emphasized that this plan should entail more than just improved coordination; a Tribal consultation plan would require Federal agencies and Tribes to agree to a process for communicating, coordinating, resolving differences, acknowledging roles and responsibilities, and establishing government-to-government relationships (Doelle, 2000). The development of the Tribal consultation plan began in 2001 and has been through various iterations; however, after 7 years and 14 drafts the plan has yet to be finalized or implemented. The extended delay in finalizing the Tribal consultation plan is symptomatic of the program's ineffectiveness and lack of ability to meaningfully integrate Native Americans.

Science as a Sociopolitical Tool

At the foundation of the GCDAMP is the role science plays in elucidating the integrated nature of the Colorado River ecosystem and a core belief that the Western science perspective is the only legitimate form of knowing the ecosystem. At the heart of this science program is a positivist approach to understanding the ecosystem that visualizes an unproblematically objective world presumed knowable via epistemologically transparent schemes of explanation (Whiteley, 2002). This perception of the world is rooted in the core Judeo-Christian philosophical perception of man's relationship to nature.

The GCMRC is the science provider for the GCDAMP and in that role it is ascribed an authoritative voice in ascertaining the condition of the ecosystem. The GCMRC employs science as a means for evaluating the health of the Colorado River ecosystem and the efficacy of management paradigms. The GCMRC also employs the concept of science tautologically as a rhetorical device for validating its authoritative role and justification of budgetary decisions.

As Ogden (2008) demonstrates for the Everglades, science when it becomes institutionalized can take on a life of its own and often is applied to meet the bureaucratic mandates of an agency. In the GCDAMP, resource goals have been bureaucratized into a set of scientific research and monitoring activities that structure the ways in which the ecosystem is comprehended and acted on. Within this context, Native American perspectives of the ecosystem are de-legitimized and marginalized in favor of the continued promotion and acquisition of scientific knowledge that supports the science program's philosophical underpinnings, self-interest, and authority.

All of these factors have contributed to the dominance of the Judeo-Christian perception of the world within the science program, which has had the unintended consequence of promoting antiquated colonialist attitudes toward Native Americans. These attitudes are a peculiar paradoxical blend of romanticized perception of Native Americans as the "noble savage" in the Rousseauian sense and at the same time antithetically perceiving them as a conquered people removed from the landscape as the result of a history of American Western expansionism. This explanation is offered not as an indictment of the GCDAMP, but as a possible rationale for the contradictory way in which the Tribes have been unsuccessfully included in this program.

Steps to a Holistic Integration

The dominance of science in the GCDAMP, to the exclusion of other valid forms of knowing the world, is in part the inability of the program to recogize that the fundamental differences between the dominant Anglo-American culture and Native American cultures lie not only in the acquisition of knowledge but also in the broader world views about what can be known about the world, who has the right to know it, and what is the proper place of humans in relation to nature (Austin, 2007). For an effective adaptive management program, differences in perception of and relating to the ecosystem must be more than just acknowledged. These differences must be embraced by the program with openness toward meaningful integration through validation.

This holistic integration can be accomplished through embracing Native American traditional knowledge in its complex forms composed of distinctive political and social perspectives rooted in a shared history, distinct ethical and cosmological knowledge, and a local knowledge of the ecosystem (Austin, 2007). The intimate ecological knowledge that the Tribes possess about the Colorado River ecosystem provides the authority and significance for their understanding and relating to this important place. This ecological knowledge is embedded in hundreds of years of directly relating to and living within the ecosystem, knowledge which has been passed on from generation to generation. The efficacy of the transmission and reliability of traditional knowledge has been well documented for ecology (Berkes, 1999), history (Whiteley, 2002), and ritual (Cushing, 1896; Bunzel, 1932).

Tribal knowledge about ecosystems is increasingly recognized as equivalent to scientific knowledge and is increasingly valued. As Hobson (1992, p. 2) points out:

> Often overlooked is the fact that the survival of aboriginal peoples depended on their knowledge, their special relationship with the environment, and their ways of organizing themselves and their values. Traditional knowledge was passed on from generation to the next. Today, aboriginal peoples are aware that they must integrate traditional knowledge into the institutions that serve them; it is essential to their survival as a distinct people, and it is the key to reversing the cycle of dependency which has come to distinguish aboriginal communities.

Traditional knowledge about the ecosystem is based then on empirical observations that are accumulated over generations providing an important diachronic perspective. Embodied within this perspective is an intuitive component that is based on observing natural resource patterns and relationships that are interpreted and integrated through the ethical and moral values and cosmological knowledge of the culture.

Accomplishing the holistic integration of Native American traditional knowledge into the GCDAMP necessitates a paradigm shift in the current science program toward an openness and willingness to accept traditional knowledge of the ecosystem on an equal basis as Western science generated knowledge. The past tendency of scientists and managers has been to reject Tribal traditional knowledge as anecdotal, non-quantitative, without method, and unscientific. For this perspective to change, a corresponding recognition that effective integration involves the sharing of power and decisionmaking by managers is essential.

A critical part of this paradigm change involves the acknowledgment by the science program that inherent in any interpretation of data and the resultant development of explanations about the ecosystem are developed through biased cultural lenses of managers and scientists. Cultural bias permeates the GCDAMP; it affects how resource data are interpreted, how knowledge is generated and defined, and how power for decisionmaking is ascribed and shared. Too little attention has been paid within the GCDAMP to developing effective mechanisms for bringing in and incorporating the knowledge and interests of Native peoples. For example, conspicuously absent from most discussions of Colorado River ecosystem management, especially for a place that is widely perceived to be a wilderness, are the poignant historical narratives of displacement, depopulation, and suffering that describe how this place came to be without humans and how the affected populations should be integrated into processes that are based in large part on assumptions that they or their ancestors are irrelevant to the ecosystem today (Austin and others, 2007).

A recommended first step toward effecting this paradigm shift is the development of a stronger social science component to the program administered by the GCMRC. Currently, the cultural resource program administered by the GCMRC is focused on the present condition of archaeological sites located along the Colorado River corridor and how current climatic conditions adversely impact or contribute to their preservation. In addition, the cultural program includes a recreation component that seeks to monitor and improve the recreational experience associated with non-Indian users of the Colorado River through Grand Canyon. Noticeably lacking in this cultural program is the integrated contributions that the disciplines of anthropology, sociology, psychology, and Native American studies can bring to this program. The integration of these disciplines would afford the program the tools to work toward the development of a holistic integration of Native American perspectives and values into the GCDAMP. Through the application of the tools that these disciplines can bring to the GCDAMP, a process for respectfully addressing and resolving conflicts of cultural values that arise within the program can be developed. Additionally, this process would allow for these conflicts to be addressed in a timely manner and thereby hopefully reduce feelings of disenfranchisement by a stakeholder group and the potential for litigious responses.

Confluence of Values

As noted above, the confluence of the Colorado and Little Colorado Rivers is a very important and significant place to the participating Tribes because of its literal and symbolic representation of fertility and life. The confluence is employed here as a metaphor to represent the fertility of knowledge and the beneficial outcomes that would result from the merging of diverse paradigms (i.e., scientific and traditional knowledge) for knowing the Colorado River ecosystem. Submitted for consideration is the view that this confluence is represented by three intersecting and overlapping circles of a Venn diagram (fig. 1). One circle represents the management paradigm of the GCDAMP, another circle represents the scientific paradigm about the Colorado River ecosystem, and the third circle represents the traditional paradigm of the participating Tribes regarding the Colorado River ecosystem, including their moral, ethical, and cosmological perspectives. The portions of the circles that overlap and intersect represent the successful merging of these three paradigms within the GCDAMP.

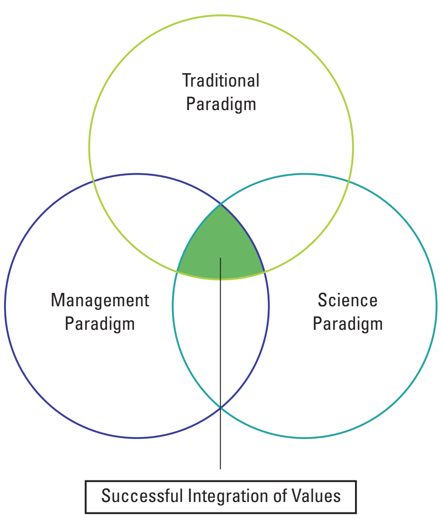

Figure 1. Confluence of values within the Glen Canyon Dam Adaptive Management Program.

This image of the confluence of values depicts a successful program of collaboration that recognizes, accepts, and seeks to integrate the diverse perspectives that scientific knowledge, Tribal traditions, and management represent. The future of working collaboratively with Native Americans within the GCDAMP rests on an honest understanding and appreciation of the diverse perspectives that have been presented above and a willingness to develop good faith communication channels between scientists, managers, and Native peoples that will only benefit the GCDAMP. When done correctly, the intersection of these competing paradigms provides an avenue for multiple views of the Colorado River ecosystem that can only enhance our understanding and appreciation of this important place.

Reflexive Approach to the GCDAMP

Finally, drawing from the social sciences, we advocate for a reflexive component of the GCDAMP. Reflexivity is an act of self-reference where examination or action "bends back on," refers to, and affects the entity instigating the action or examination. In brief, reflexivity refers to circular relationships between cause and effect. A reflexive relationship is bidirectional; with both the cause and the effect affecting one another in a situation that renders both functions causes and effects. Reflexivity is related to the concept of feedback and positive feedback in particular.

As applied to the GCDAMP, we believe there is utility in examining the internal social dynamics of the program and the interaction among participating groups. Specifically, we believe that it is important to examine and understand the power and gender relationships that exist within the AMWG and TWG and how these affect discourses among the stakeholders and the recommendations they generate. Moreover, a reflexive analysis should examine the dynamics of cultural differences that are operant within this program, some of which have been presented above. Through the examination of these cultural differences, a clearer understanding of the role the dominant cultural bias plays within the program and how that bias impacts and directs the program's perspective on ecosystem resources and data can be achieved. To this end, we encourage the planning and implementation of the GCDAMP effectiveness workshop, but that it should be expanded to include this reflexive component.

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The Promise and Peril of Collaboration in the Colorado River Basin

Remarks prepared for delivery by Kirk Emerson[1]

Thank you, John and Ted, for the opportunity to wrap up this morning after 2 1/2 days of packed information exchange on Colorado River science. I was invited to talk about collaboration and conflict resolution and chose a title for these remarks that should suggest that I will not be giving you a one-sided perspective on collaborative engagement; there can be a dark side to these efforts as well. However, I will suggest we can be smarter in how we do more and better collaboration in the future.

By way of background, I have lived in Tucson since 1995 when I came to the University of Arizona and started working on collaborative resource management as a political scientist. I had an earlier career in environmental planning at the county level back in Pennsylvania. My academic work soon got sidelined, however, for a good 10 years while I served as director for a Federal program called the U.S. Institute for Environmental Conflict Resolution at the Morris K. Udall Foundation in Tucson. The mission of the Institute is to assist Federal and nonfederal parties in working together to solve tough environmental and natural resource problems.[2] Sometimes that is done by mediating disputes referred by Federal court or administrative tribunals, but most often it is in helping public agencies and stakeholders reach common ground when developing policy, planning, siting, reviewing proposed actions, and managing environmental, natural resource, and public lands issues.

As a Federal program, the Institute has a national reach and works on issues all over the country from the Everglades to the coral reefs of the northwestern Hawaiian Islands. We have worked on a number of issues in the Southwest, all of which had important science components, many of which emerged from or became part of adaptive management efforts. For example, collaborative monitoring for forest management in New Mexico; recovery planning for the desert tortoise in the Southwest; sage grouse habitat conservation in the Northern and Central Plains States; Sonoran pronghorn protection on the Barry M. Goldwater bombing range; and many watershed and river basin restoration efforts involving multiple Federal, State, and Tribal actors in addition to nongovernmental stakeholders, the most ambitious of which is the recovery planning going on right now in the vast Missouri River Basin.

So I have had many opportunities to observe directly and indirectly a number of cooperative resource management efforts from the inside, but not as an insider, rather as a third party, bringing an outside perspective to the deliberations without a dog in the fight, other than to help people try to make more informed and equitable decisions together.

In the interest of full disclosure, I should note that my professional experience in the Colorado River Basin is limited, although I confess to some firsthand observation of beach improvements this spring when I spent 18 days on the river with the Grand Canyon Field Institute's annual rafting trip. This is also my first opportunity to attend a science conference about the Colorado River, and I am indeed overwhelmed with the breadth and diversity of the research being conducted.

My invitation to speak today included a charge to draw from the previous sessions and make connections for you between these presentations and the potential and challenges for collaboration as you move forward. I did my best to attend all the plenary sessions and at least sample sessions in each of the concurrent panel tracks. But my observations are certainly limited to what I was able to take in as well as what struck me as relevant to future collaborative science and management in the river basin. I will start with a few observations on what emerged for me from this symposium and then turn to a few comments about the promise and perils of collaboration, highlighting the need to:

- acknowledge just how hard it is to "do" collaboration and recognize some of the challenges that demand attention,
- pay more attention to basic principles of collaboration as well as challenge some of our long-standing assumptions, and
- refresh and adapt management programs, not just the science, over time.

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What I have witnessed over the past few days in the sessions, along the corridors, and during meals and receptions is an amazing engagement of scholars and managers in the science of the Colorado River and its implications for management. At the same time, there are some signals of tough challenges ahead, primarily generated from outside the river basin, that will test the adaptive capacity of the physical and management systems even more.

First, as I understand it, this is the first science and resource management symposium of its kind encompassing the entire Colorado River Basin. What a feat! That alone certainly took cooperation among the basin programs and a lot of coordination. I would not underestimate the significance of this event itself. Could it have happened 10 years ago? Here is a quick thought experiment: Imagine what such a conference might have looked like in 1998 and consider the progress in your own fields of research since then. What would you have been presenting? What kinds of management questions would you have been asking? Who would you have been collaborating with on your research? How many young researchers and managers have been recruited to this work since then? Perhaps there are some of you who would be discouraged when answering these questions. But I would wager that most of you could personally attest to the progress that has been made in generating usable knowledge about the Colorado River. A substantial investment in research and resource management has been made over the past 10 years. Could it have been more? Well, of course so, that answer is always yes. But I hope you will just take a moment to acknowledge this symposium as a real-time benchmark in and of itself.

I would also observe that there has, no doubt, been an increase in collaborative science on the Colorado River over the past 10 years. I do not have the data for this myself, but it could be easily measured if it has not been already. The size and diversity of the research teams presenting over the last few days is impressive. Not just the number of scholars but the multi-institutional cooperation and support. Is this too an accomplishment of the investments made by your adaptive management programs? Have they leveraged more additional funding from multiple agencies and private sources? This is another benchmark for your collective accomplishments.

Several speakers talked about the need for better integration and coordination across the subbasin programs—to exchange information, compare findings, collaborate on data collection, etc. Perhaps this theme is a function of the novelty of this meeting, but it is impressive to me that people are talking about the need for integration, when so many other areas are competing for resources and defending turf. While I would generally agree with John Shields and Gerald Zimmerman that the diversity among the subbasins probably warrants continuing distinct programs, perhaps this interest in integration is another expression of the kind of "collaborative capital" that is being built throughout the Colorado River's science and management community.

I would like to think so, because another theme arising from this conference is that there are significant external changes coming to the natural system that were not anticipated 10 years ago, and they will require all the collaborative capital you can muster. Chief among these challenges, of course, are invasive species and climate change. Brad Udall's presentation on the warming trends in the Southwest and the likely reductions in rainfall and runoff was chilling, as were the data and projections on the quagga mussels by Thomas Nalepa from National Oceanic and Atmospheric Administration (NOAA).

These threats are no longer hypothetical and they will undoubtedly upset the management agreements and priorities for science that have previously been set along the Colorado River.

But such unexpected pressures, of course, are what adaptive management regimes are meant to handle, are they not? They are intended to be flexible and responsive to new discoveries, surprises, unanticipated outcomes, and changing conditions. Adaptive management is not just about trying to reduce uncertainty, it is also about sharing future risk, that is, building enough trust and transparency to allow for public experimentation and adjustment. Adaptive management is about enabling public decisionmaking that can tolerate some degree of failure in order to learn and adjust accordingly to prevent greater failure in the future.

Another recurring theme that suggests future challenges ahead was the mention of "incremental adaptation" by Kameran Onley when she referenced the National Academy of Sciences report and what I assume is a related phenomenon Kathy Jacobs brought up—"stationarity"—discussed in a recent "Science" article. While I cannot attest myself to the prevalence of incrementalism in any of the basin programs, it strikes me as a rather inevitable outcome of adaptive management approaches generally, which may not be a bad thing when you only have to effect change around the margins. But what about when bold, decisive action is demanded? Or when you have to adapt to 40 percent less annual runoff or manage the full-scale invasion of quagga mussels in Lake Powell or generate more hydropower to meet new renewable energy portfolio standards? It is likely that in the not too distant future more boldness will be required by Colorado River managers and water users. Will the gains of adaptive management inform those decisions? Will the collaborative capacity be strong enough and ready to shape those decisions?

This leads me to another emerging theme from this symposium—the call for setting priorities. We heard it from John Schmidt when he suggested we need to address the cost/benefit of certain management objectives. We heard it again during the late Tuesday afternoon discussion and again this morning with a call for more trade-off analysis; we need to prioritize the restoration and recovery goals for different river reaches for a variety of reasons, not the least of which are the financial constraints that are likely to set in when our economic crisis translates into future budget cuts at Federal and State levels. Setting priorities means, of course, setting up for difficult decisions, acknowledging there will be shared pain and few gains. This is when collaboration really gets tough. It is no longer about sharing a growing pie of research dollars or sizable funding for management options, it becomes a negotiation over minimizing losses, with fewer, not more, options to consider. We cannot have it all, as John Schmidt reminds us. There will be losers. And when there are losers, there will be conflict. Pretty grim picture, right?

So the conclusion I draw from these general observations is that more and stronger collaboration will be needed in the next 10 years, and more conflict anticipation, management, and resolution. It is not really about whether collaboration is good or bad, but rather how we can improve the way we work together. How can we optimize the effectiveness of our partnerships and our adaptive management programs? And as Dennis Kubly noted, these challenges are opportunities as well.

We are all aware of the promises "process" advocates like me make about collaboration. It reduces the risks of protracted disputes, and it helps leverage shared resources. It enables trust to emerge and social capital to be built. It creates certainty and leads to better, more informed decisions. And after all, adaptive management depends on such decisions.

What we do not acknowledge adequately is just how difficult it is to do collaboration well and the jeopardy that can occur when attempts at collaboration go south. Here are just a few of the many perils of collaboration and, by extension, adaptive management processes that I have seen over the years (perhaps you will recognize some of these from your own experiences here in Colorado River programs or elsewhere):

- All parties needed at the table are not willing to participate, yet the collaborative process moves forward nonetheless; or parties participated in bad faith or did not abide by the group norms or ground rules;

- Decisionmakers come and go, and agency commitments are not honored and as a consequence undermine the parties' confidence in the group's legitimacy or efficacy;

- Cultural differences among the parties disadvantage some and privilege others, and the norms of the process do not recognize or adapt to these differences, leading to declining participation by some groups who feel increasingly marginalized;

- Difficult personalities dominate the process and suck the energy right out of a group. Membership falls off as people prefer to avoid conflict; or

- Unresolved differences continue to fester and express themselves in renewed power struggles or end runs.

So how do we avoid all these pitfalls and make collaboration work? Let me give you two kinds of answers, with the caveat that there is no silver bullet; there is no getting around that collaboration is not for the faint of heart, particularly when you have high conflict and low trust among the stakeholders and across agencies as well.

The first answer lies in returning to first principles; that is, the best practices of conflict resolution and collaborative problem solving. In November of 2005, these were actually codified in a policy memo on environmental conflict resolution issued by the Office of Management and Budget (OMB) and the President's Council on Environmental Quality (CEQ) (Office of Management and Budget and Council on Environmental Quality, 2005). Eight principles were cited as important to ensuring the effective use of environmental conflict resolution (ECR). They apply in my view to collaborative resource management as well. I will briefly list them:

1. Informed commitment – assuring your agency or constituency is fully aware of the issues on the table, the sideboards to the discussions, and nature of the commitment and the decisionmaking authority of the group;

2. Balanced and equitable representation – of all affected and engaged interests;

3. Group autonomy – to design its own ground rules, set its own agenda;

4. Informed process – where access to all available information is assured;

5. Accountability – where participants acknowledge and work with the tension between representing their agency or organization and also hold themselves accountable to the group's shared mission and goals;

6. Openness – transparency in decisionmaking not only for the group's benefit and to build trust but for the benefit of larger public engagement and confidence in the group's legitimacy;

7. Timeliness – in decisions and actions within the regulatory management constraints; and

8. A focus on implementation – such that any proposals and recommendations are feasible and most important fundable.

These principles were gleaned from some 30 years of practitioner experience as well as negotiated by a Federal interagency working group. The policy memo, I might add, directs all Federal agencies to increase the effective use of ECR and collaborative problem solving and report annually to OMB and CEQ on their progress. A synthesis of these annual reports, now in the third year of reporting, is available as well. Hopefully this will be one of the policies the new Obama administration builds on.

The other answer to the question of how to optimize collaboration can be found in empirical research. Frankly, there is not a lot of research that directly links certain process attributes or practices to collaborative outcomes. In a recent study of 52 ECR cases, however, co-sponsored by U.S. Department of the Interior (DOI), U.S. Environmental Protection Agency (EPA), U.S. Forest Service, Federal Highway Administration (FHWA), some State partners, and the Udall Foundation with funding from Hewlett Foundation, over 500 participants responded to post-process surveys. The study found that reaching high-quality agreements and building social capital were optimized when the appropriate parties are at the table and effectively engaged; when high-quality, relevant information is accessible to all the parties; when parties have the capacity to engage; and when the facilitator or mediator employs the appropriate skills and practices. Among all those factors, effective engagement makes the strongest contribution to positive outcomes on the basis of the multilevel modeling analysis conducted in the study (Emerson and others, 2009).

This research does not speak, however, to another concern about the performance of adaptive management programs that may be relevant in particular to the Glen Canyon Dam Adaptive Management Program. That is, how to enhance or reinvigorate or remediate a longstanding collaboration. There are a growing number of older (I won't say aging) adaptive management programs started in the 1990s that face a new set of problems associated with the perils of institutionalization. Chief among these are such problems as:

- Process fatigue,
- Free riders,
- Party bail out or exercise of other options outside of the collaboration to meet their needs,
- Weakened commitment to the process or the originating mission of the group and adversarial attitudes and behaviors re-emerge,
- Once-flexible dynamics turn into formalized meetings where there is little room for real deliberation, and
- Lack of measurable progress toward improved environmental conditions discourages many involved.

These are difficult challenges. Do they invalidate adaptive management programs and their performance? The jury frankly is still out on the effectiveness of large-scale ecosystem restoration programs like those in the Everglades, the Bay Delta, the Chesapeake Bay, and the Columbia River as well as Glen Canyon Dam (Gerlak and Heikkila, 2006).

My own view is that adaptive management approaches are essential—in fact, there is no other approach at hand that can begin to deal with the complexity of these natural systems or the corresponding management challenges. With respect to retrofitting or retooling longstanding collaborative processes, I think it can be done, it is being done, and often requires assertion of new leadership, the help of outside consultation, and considerable work by the parties in assessing their individual and shared commitments.

That said, we have yet to master the adaptive management of the adaptive management program itself. And I think we have fallen victim to our own mythologizing about collaborative action and its precursors. We need to challenge some of our underlying assumptions.

For example, we talk reassuringly about the voluntary nature of these collaborations, reifying agency and self-determination. First of all, there is often some level of coercion of reluctant parties to participate, even if it is against their self-interest. But more importantly, as John Duffield will probably tell you, people respond to cues—positive and negative incentives—as we make choices. In fact, using the power of the State as an incentive to get people to the table may not be so bad. Would we have the shortage agreement on lower Colorado River Basin flows today had not the Secretary of the Interior set deadlines for the States to negotiate a plan? Acknowledging the value of external deadlines and clear consequences set by legal authority (the exertion of leadership) might be a very good thing for incentivizing joint action.

Another sacred cow of collaboration is the consensus decision rule. Unanimity can certainly be hoped for, but it is rare indeed and usually occurs only when the level of disagreement among parties is low and the stakes of the particular decision are not very high. There are many ways to set more sensitive decision rules that neither set the bar too high nor lower it to voting rules that regularly disempower minority views.

These and other assumptions about collaboration could benefit from further empirical research and theoretical challenge.

Let me conclude by underscoring what Dennis Kubly said on Tuesday about neglecting the human dimension of adaptive management. If we do not pay more attention to the psychological, the social, the cultural, the political, and the institutional dimensions of adaptive management, we risk losing the ability to translate the biophysical science we have generated into the target management options on the ground. There are now several collaborative resource management programs working in the basin consuming energy and resources, providing outputs, and interacting with the environment. The productivity of these collaborative programs may well be as important as the productivity of a given fishery or stretch of riparian habitat, as we depend on them both for the protection of ecological services. So I encourage you to reflect on, indeed, to research ways to optimize your collective productivity and explore how to adaptively manage your adaptive management programs and build more collaborative capital. Indeed, if the forecasts are correct, you will need to draw on it in the not too distant future.

Congratulations on a terrific conference. It bodes well for future progress along the Colorado River. Thank you very much, and I will be glad to take a few questions.

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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.

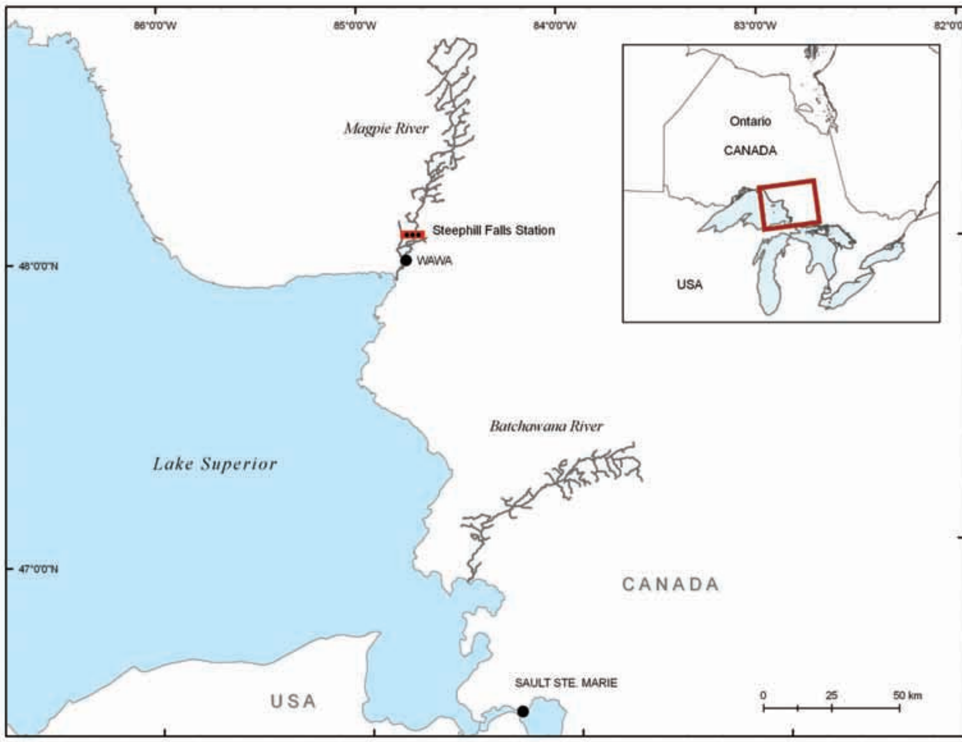

Figure 1. Map showing location of the Magpie and Batchawana Rivers relative to Lake Superior and Sault Ste. Marie, Ontario.

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.

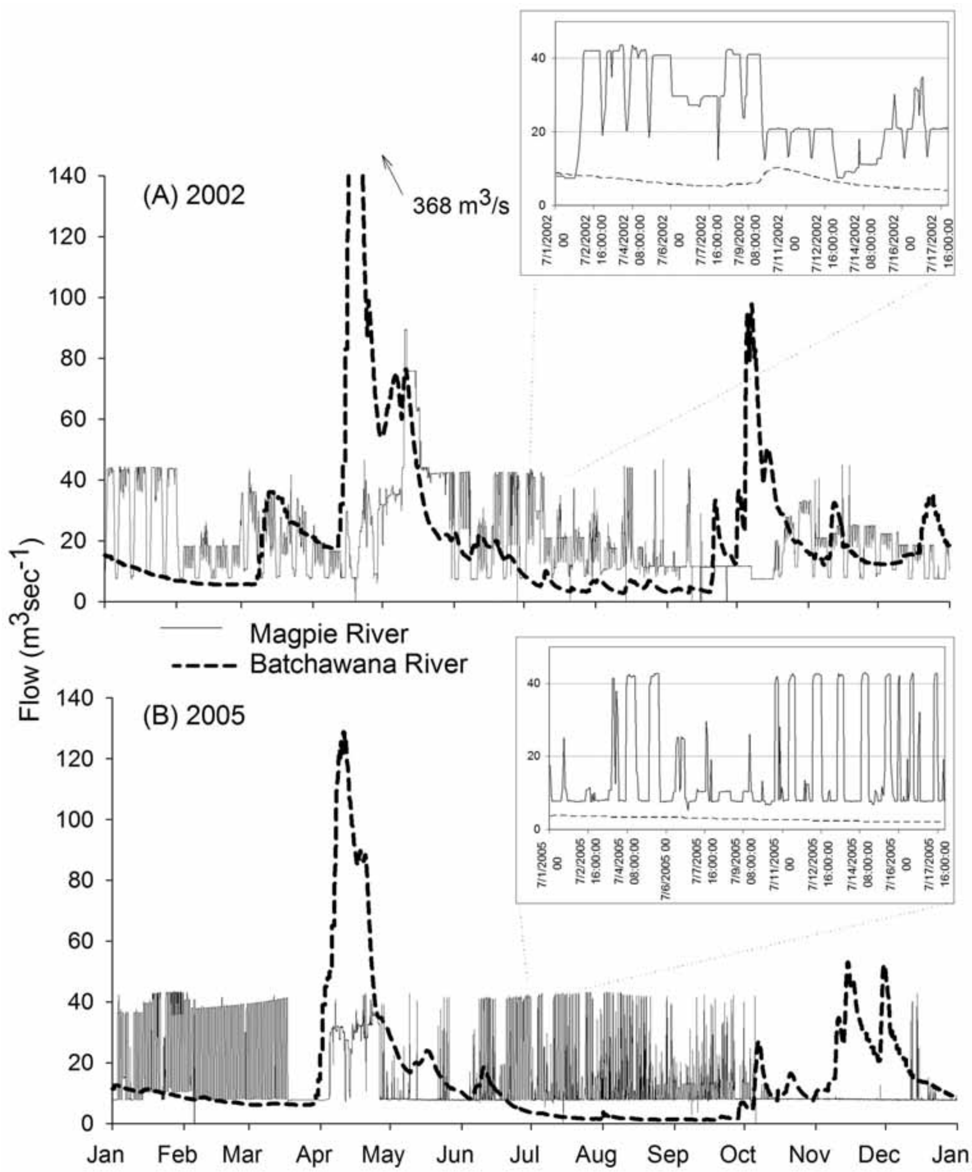

Figure 2. Annual hydrograph of the Magpie (solid lines) and Batchawana (dotted lines) Rivers, (A) in 2002, before ramping change on the Magpie, and (B) in 2005, after unlimited ramping on the Magpie River. Data for the Magpie River from the Steephill Falls waterpower facility (courtesy Brookfield Renewable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

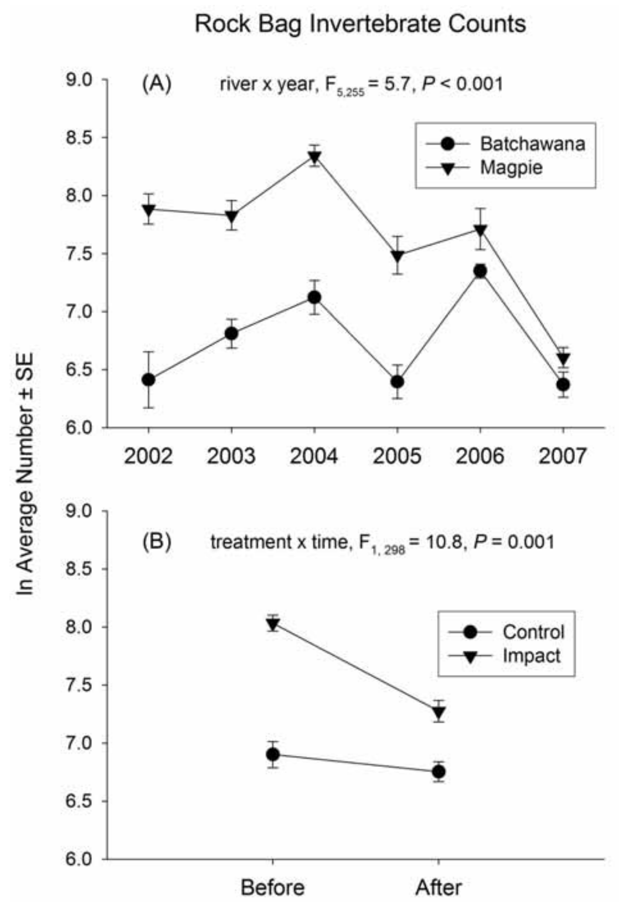

Figure 3. Average abundance (log + 1 transformed) of invertebrates per rock bag ± standard error (SE) plotted as (A) average across sites for each year, and (B) as the before-after-control-impact plot.

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

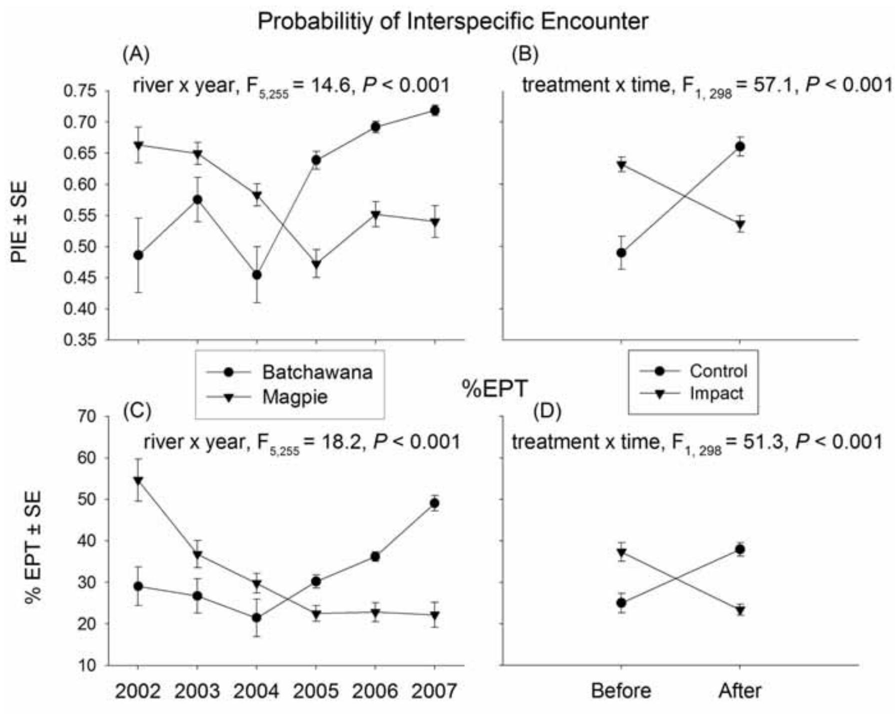

Figure 4. Average diversity (PIE) of invertebrates per rock bag ± standard error (SE) plotted as (A) average across sites for each year, and (B) as the before-after-control-impact plot. Average % EPT invertebrates per rock bag ± standard error (SE) plotted as (C) average across sites for each year, and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

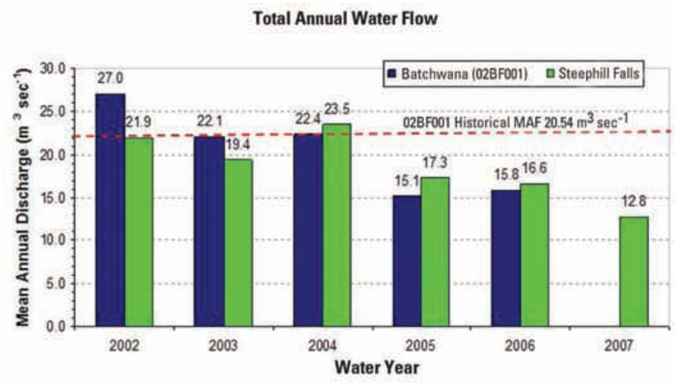

Figure 5. Total annual flow ($m^3{\cdot}s^{-1}$) on the Magpie and Batchawana Rivers both before and after unlimited ramping was implemented in 2005. The red dotted line indicates the mean annual flow for the Batchawana River as calculated from historical water survey of Canada data.

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.

Acknowledgments

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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

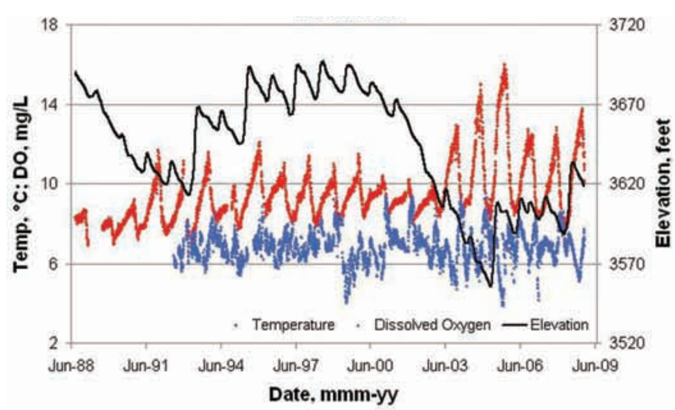

Figure 1. Daily water temperature and dissolved oxygen concentration below Glen Canyon Dam with Lake Powell water-surface elevations, 1988–2008 (adapted from Vernieu and others, 2005).

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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,

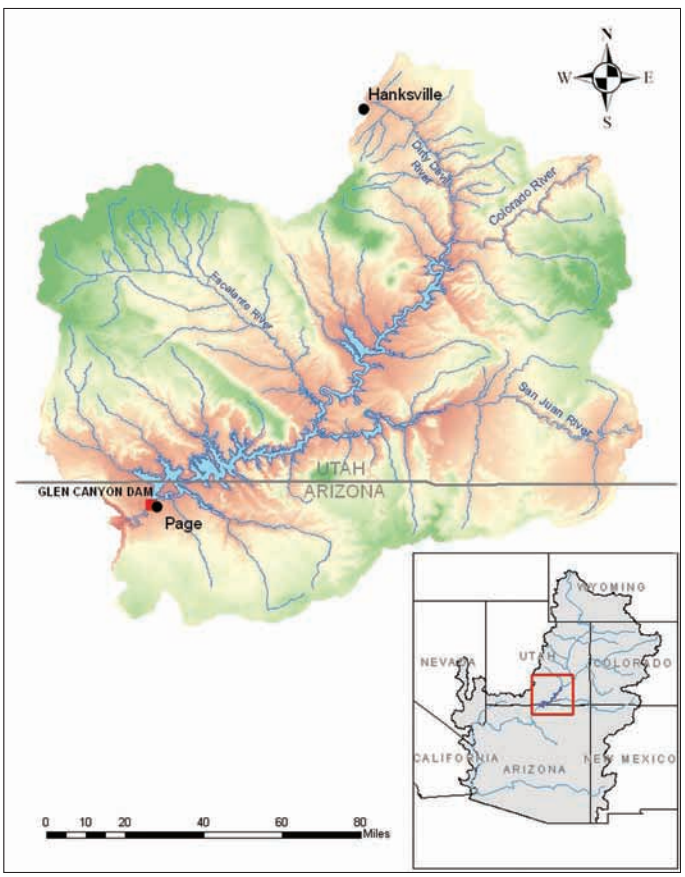

Figure 2. Lake Powell and immediate watershed showing location of Glen Canyon Dam; inset shows the location of Lake Powell in reference to the Colorado River Basin

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).

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).

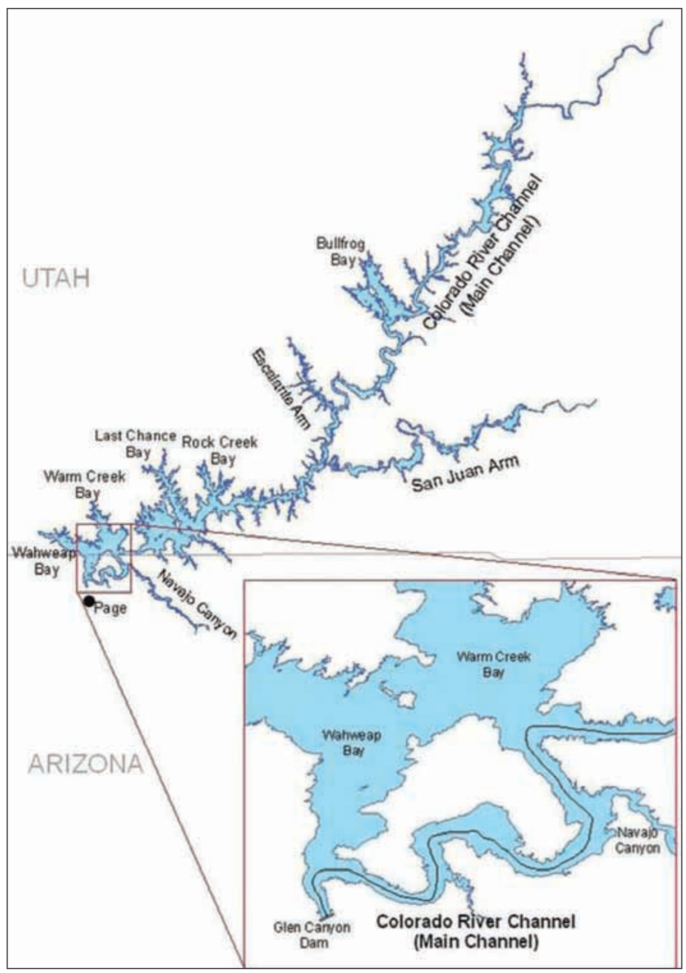

Figure 3. Lake Powell channels and bays.

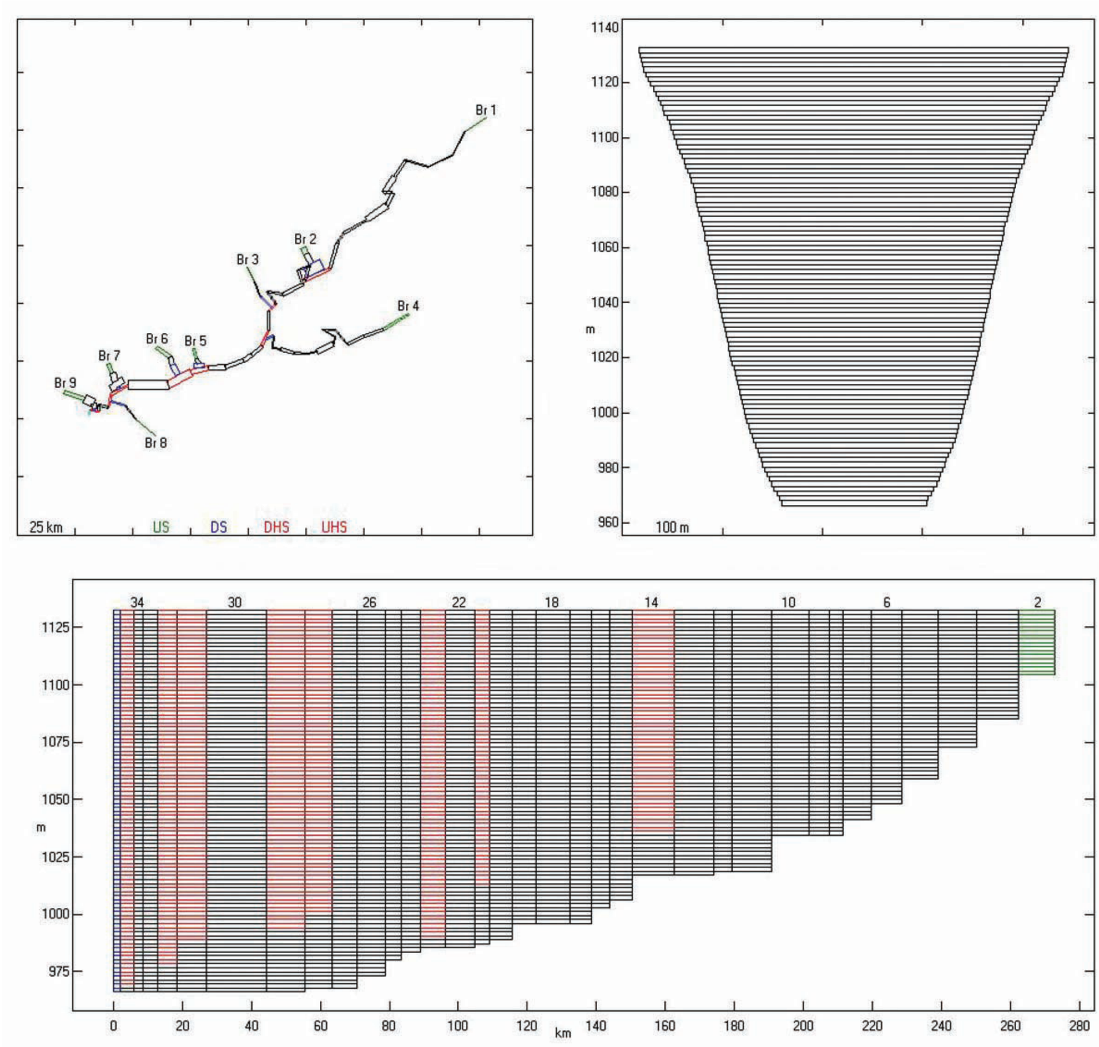

Figure 4. Lake Powell computational grid displaying plan, side, and profile views of the grid.

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.

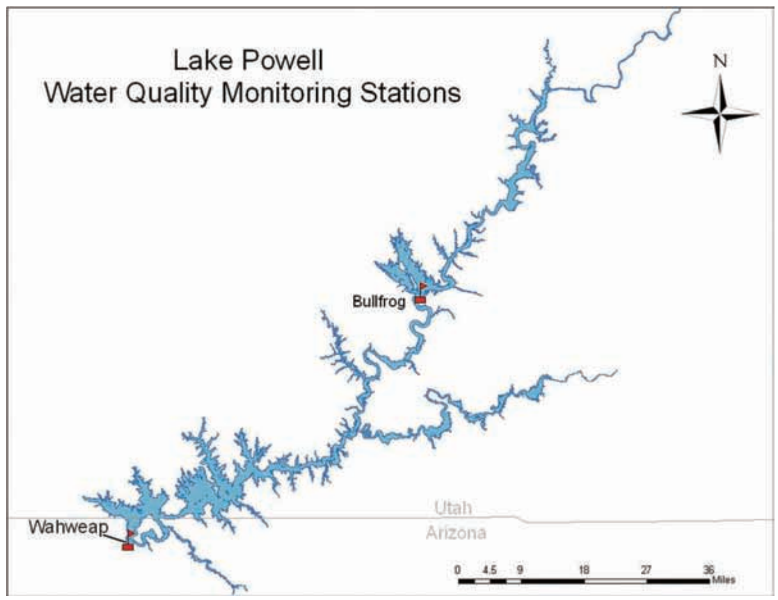

Figure 5. Lake Powell showing Wahweap and Bullfrog monitoring locations.

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

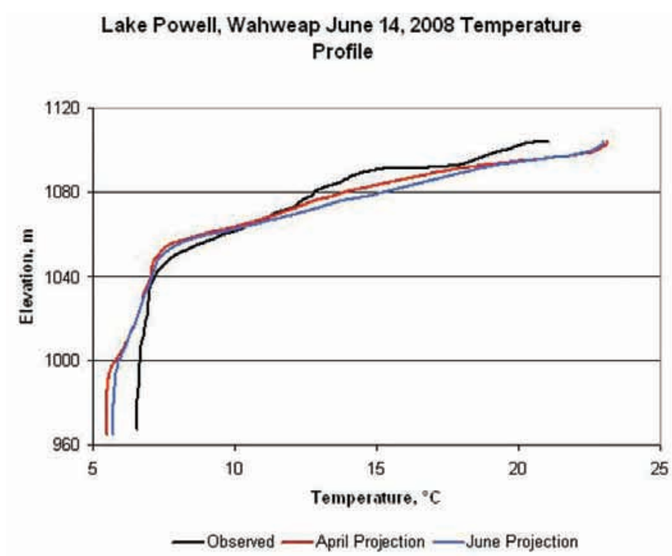

Figure 6. Wahweap, Lake Powell, June 14, 2008, temperature profile comparing projection simulation and observed temperatures.

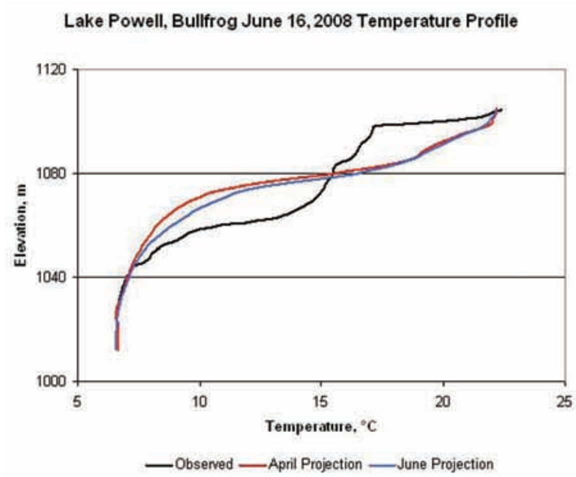

Figure 7. Bullfrog, Lake Powell, June 16, 2008, temperature profile comparing projection simulation and observed temperatures.

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

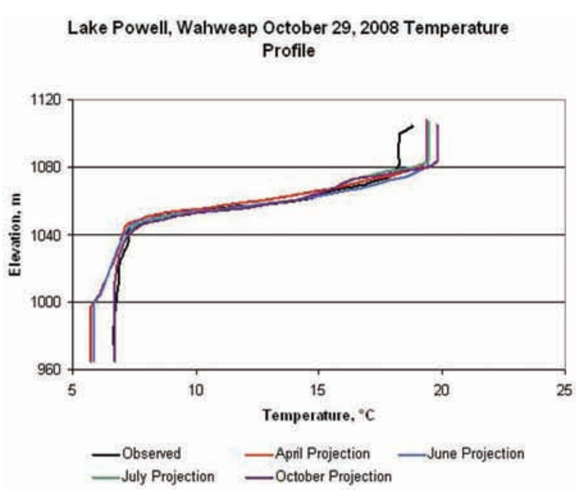

Figure 8. Wahweap, Lake Powell, October 29, 2008, temperature profile comparing projection simulation and observed temperatures.

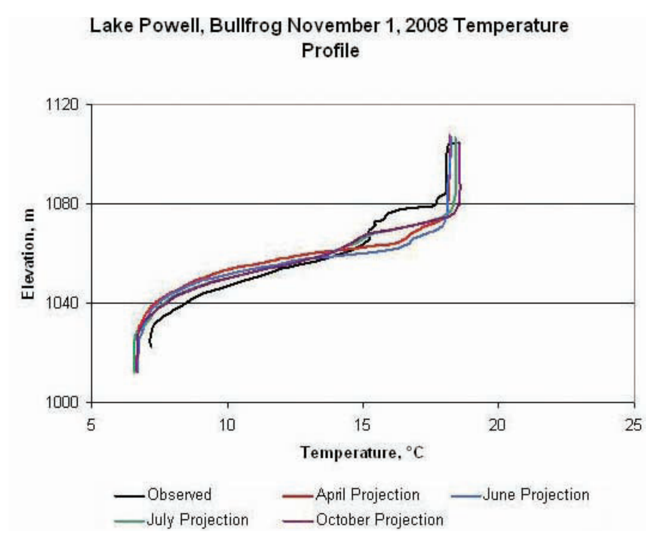

Figure 9. Bullfrog, Lake Powell, November 1, 2008, temperature profile comparing projection simulation and observed temperatures.

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.

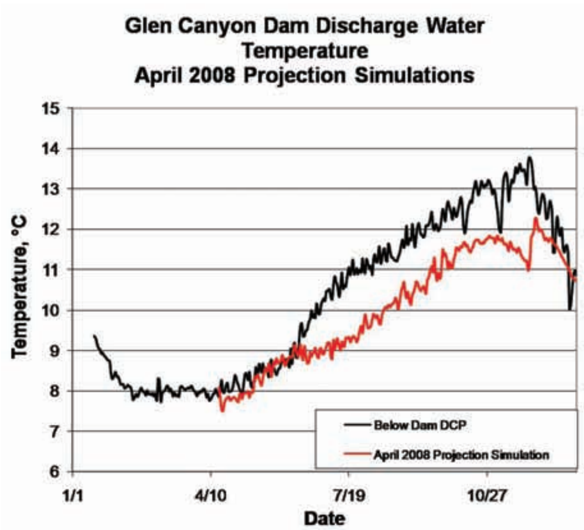

Figure 10. Glen Canyon Dam discharge water temperature, April 2008 projection simulation temperatures compared to Below Dam DCP temperatures.

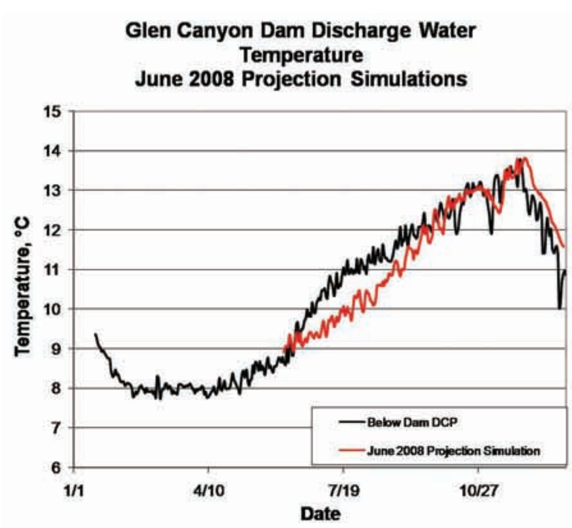

Figure 11. Glen Canyon Dam discharge water temperature, June 2008 projection simulation compared to Below Dam DCP temperatures.

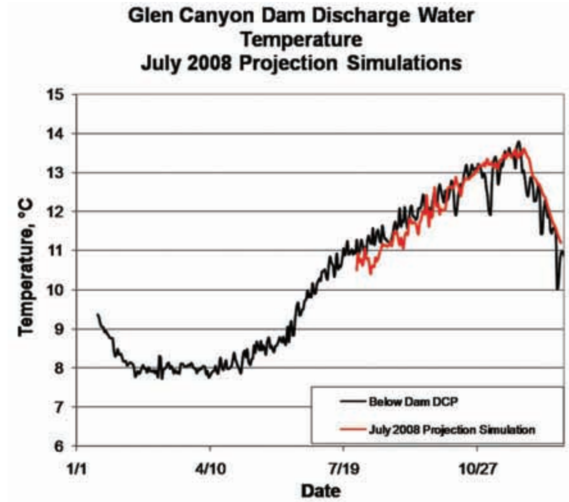

Figure 12. Glen Canyon Dam discharge water temperature, July 2008 projection simulation compared to Below Dam DCP temperatures.

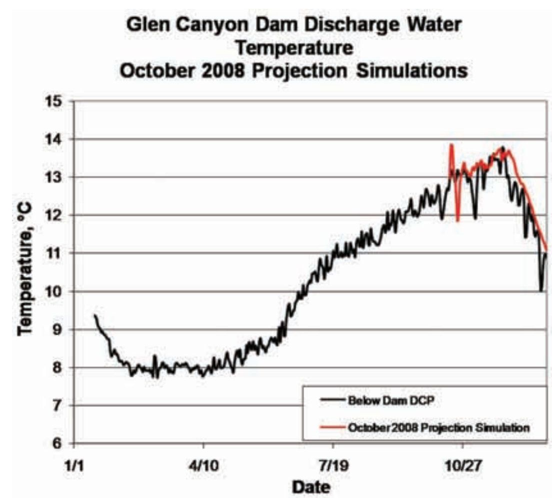

Figure 13. Glen Canyon Dam discharge water temperature, October 2008 projection simulation compared to Below Dam DCP temperatures.

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

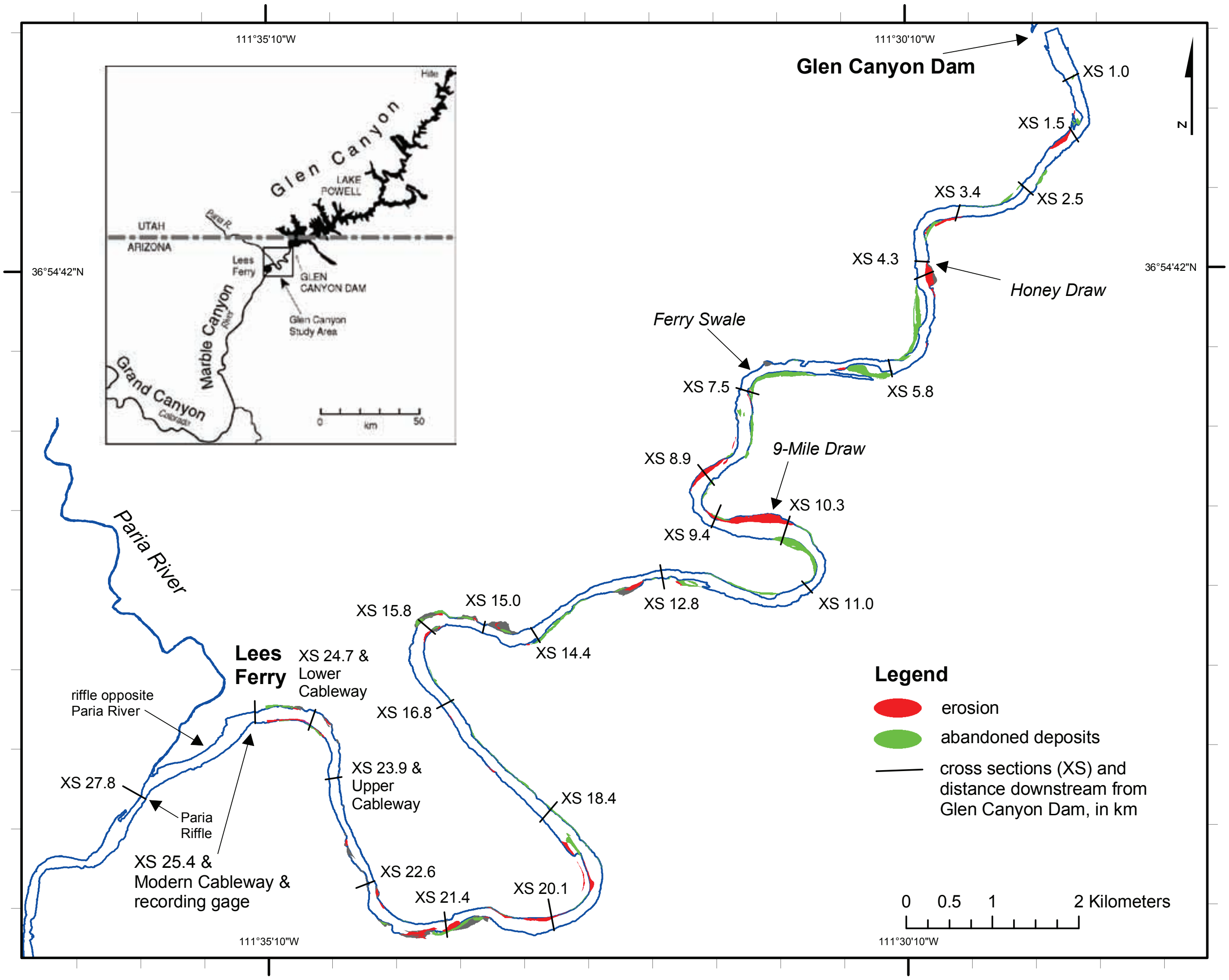

Figure 1. The Colorado River in Glen Canyon downstream from Glen Canyon Dam showing locations of monitoring cross sections and areas of erosion and deposition of channel-side deposits.

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,

(A)

(B)

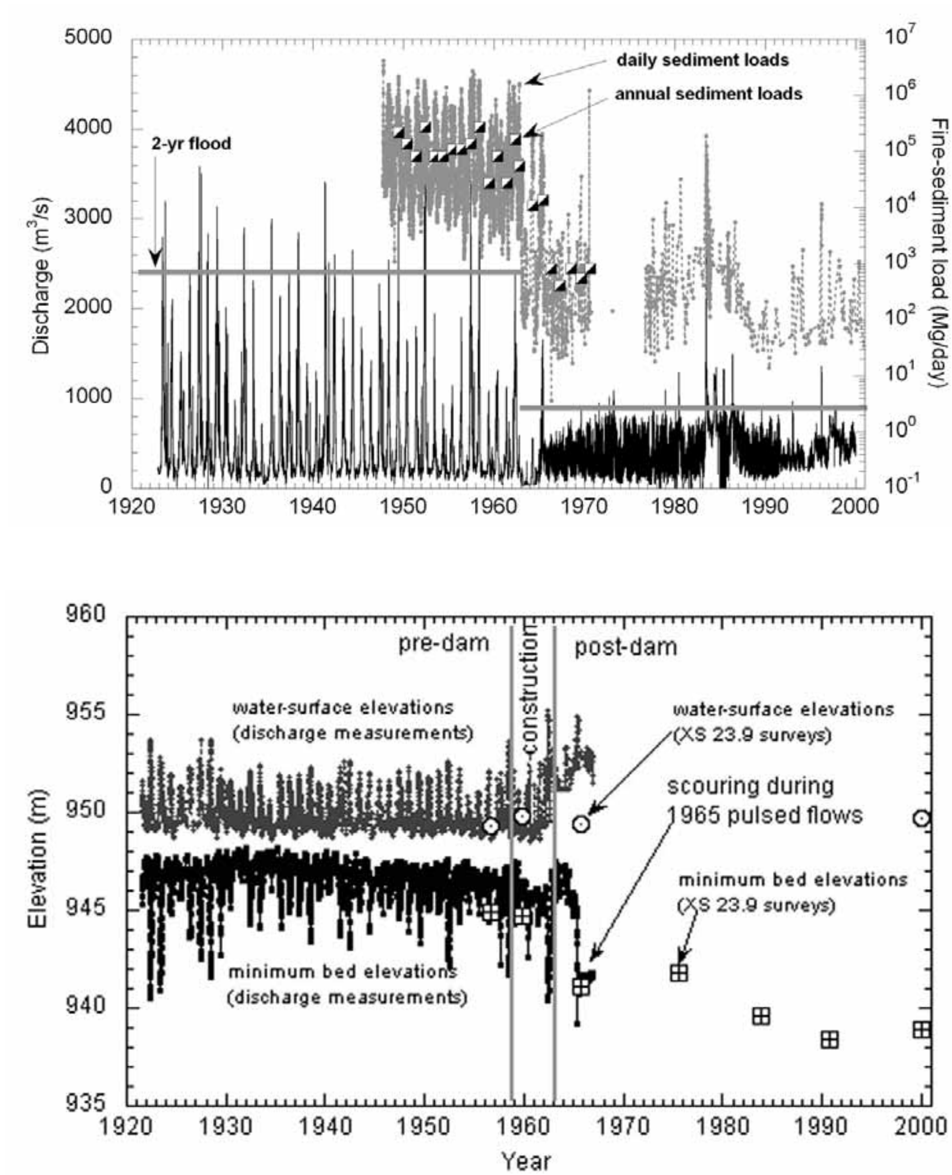

Figure 2. (A) Instantaneous discharge of the Colorado River at Lees Ferry, Arizona, 1921 to 2000, and measured sediment load for the same location, 1947 to 2000. The gray points connected by the dashed line are the computed loads for each day that sediment concentration was measured. The black and white 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

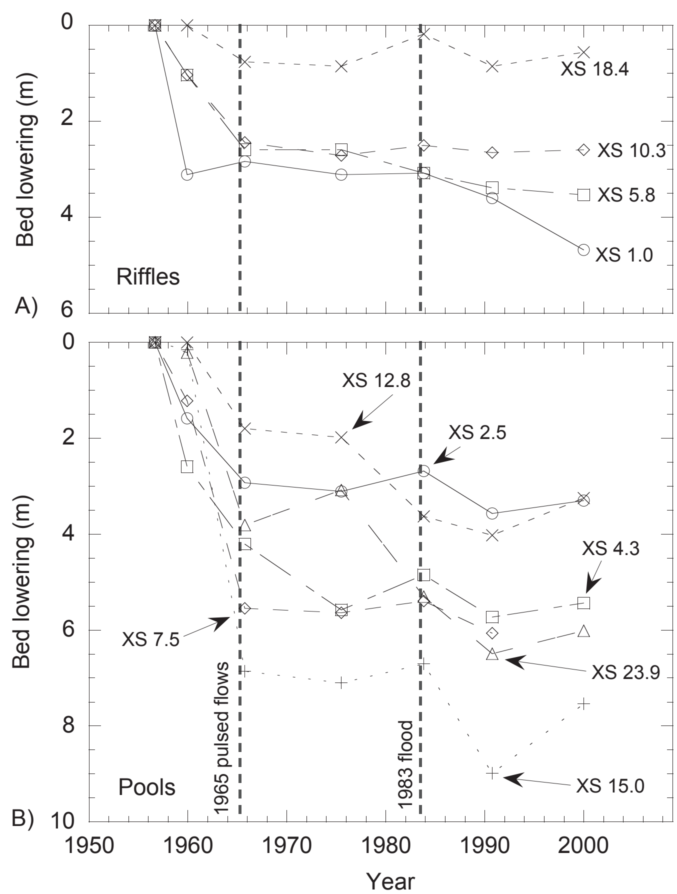

Figure 3. Magnitude of decrease in minimum bed elevation from 1956 to 2000 at the 10 monitoring cross sections grouped by (A) riffles and (B) pools. Each cross section is labeled by distance downstream from Glen Canyon Dam, and the times of the 1965 pulsed flows and the 1983 flood are indicated.

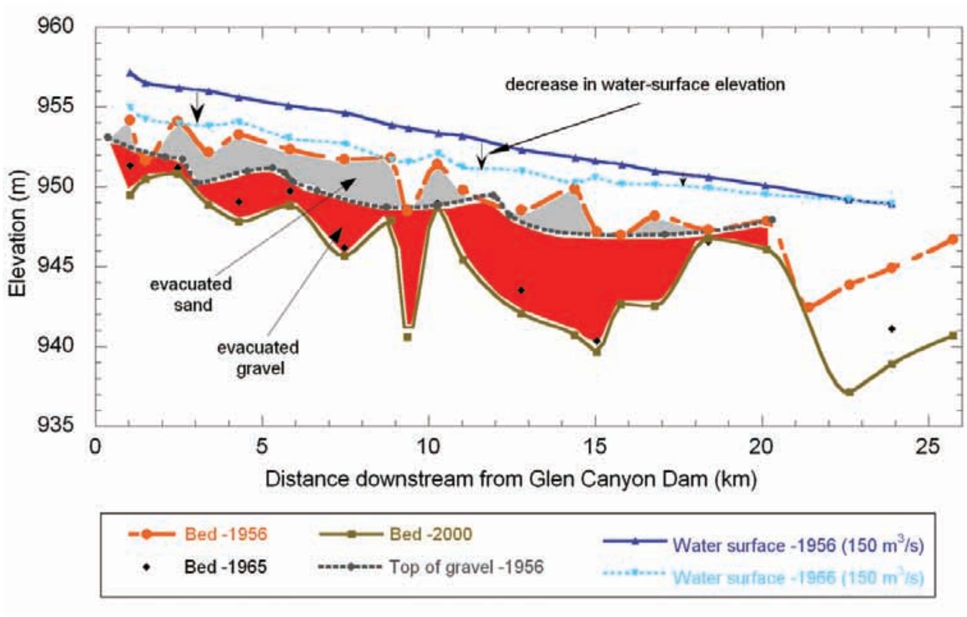

Figure 4. Longitudinal profile showing minimum bed elevation for each of the Bureau of Reclamation surveys and elevation of the top of the gravel layer determined by bore-hole and jet-probe measurements made in 1956. Water-surface profiles for a low-flow discharge of 150 m^3/s are also shown. For the 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.

(A)

(B)

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 m^3/s at the time of the 1952 photograph and 141 m^3/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.

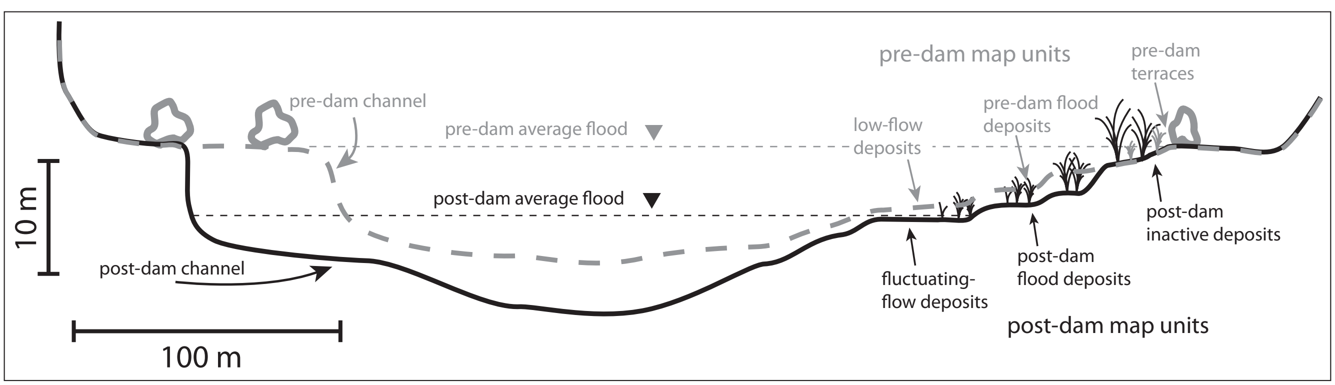

Figure 7. Relative elevations of mapped deposits in relation to the pre- and post-dam Colorado River channel. The approximate stages of the pre- and post-dam average (2-year recurrence interval) high flows of 2,407 m^3/s and 892 m^3/s, respectively, are also shown. The post-dam average high flow is approximately 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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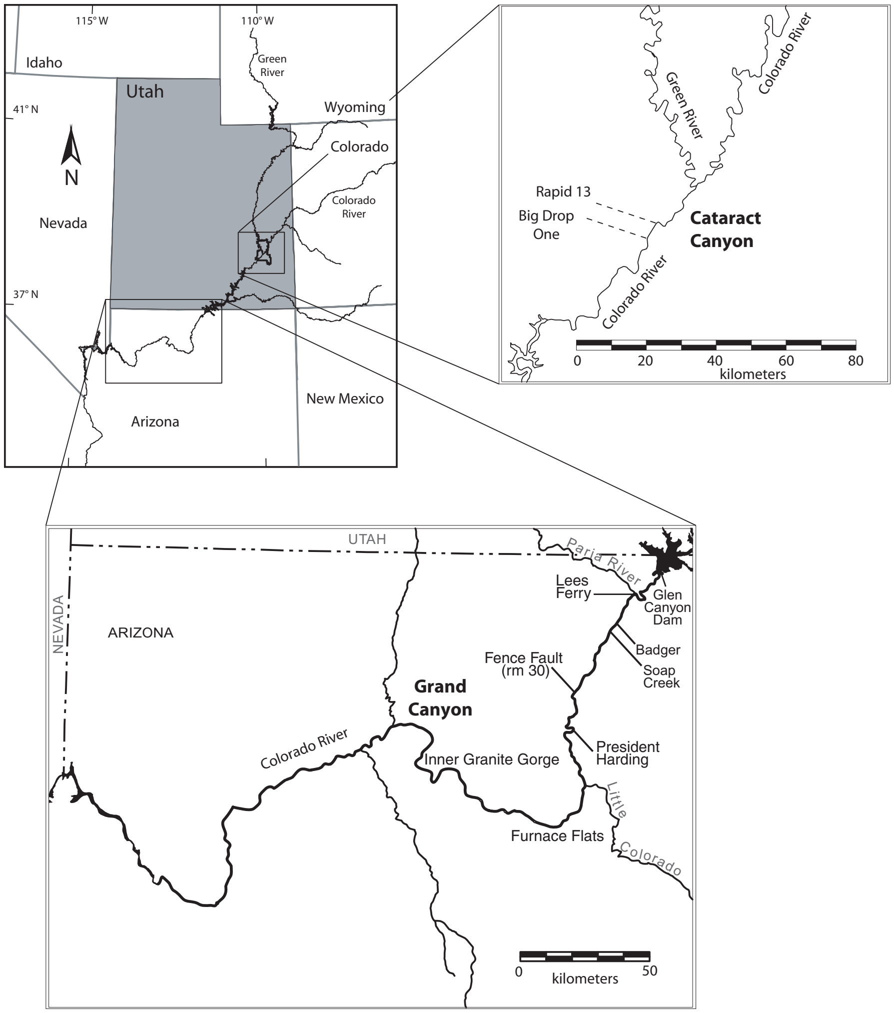

Figure 1. The Colorado River in the Southwestern United States. Water velocity was measured at locations in Cataract Canyon in Utah and Grand Canyon in northern Arizona.

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

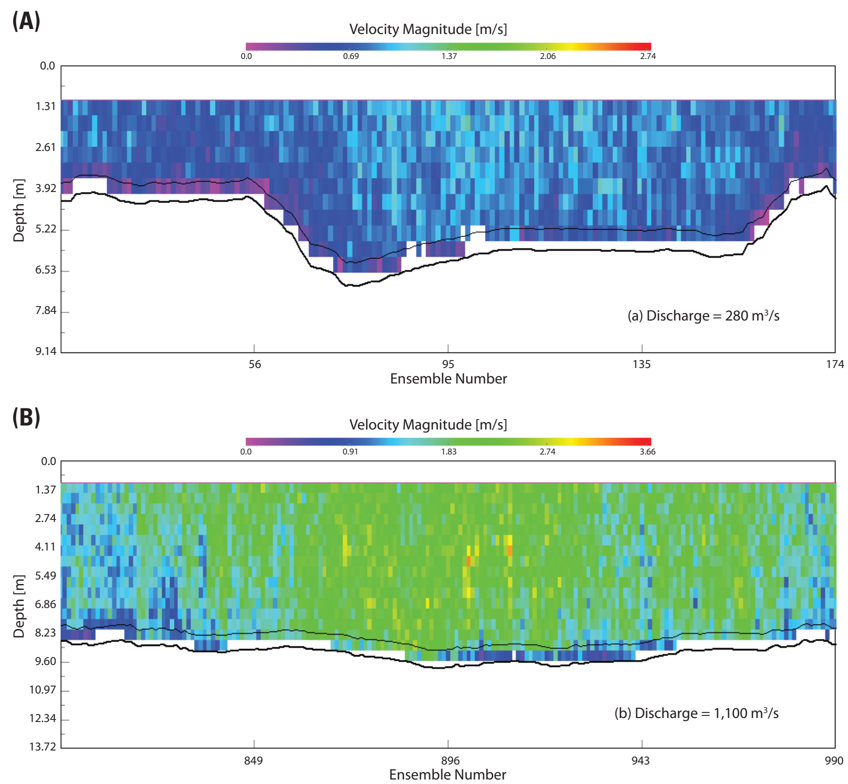

Figure 2. Flow velocities as measured with an acoustic Doppler current profiler are shown in a pool section of the Colorado River in Grand Canyon near river mile 30 for (A) low discharge of about 280 m^3/s and (B) larger discharge of about 1,100 m^3/s. The term "ensemble" refers to serial measurements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

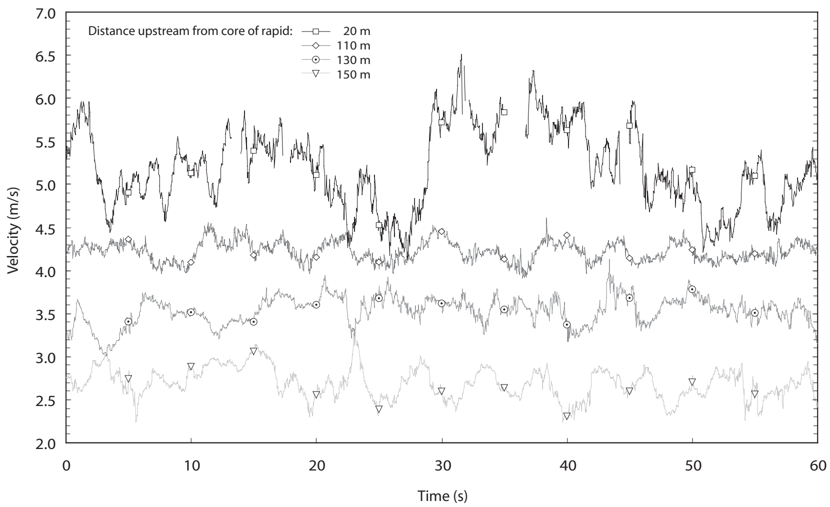

Figure 3. Turbulent flow velocities measured at the top of Big Drop One Rapid on the Colorado River in Cataract Canyon.

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 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.

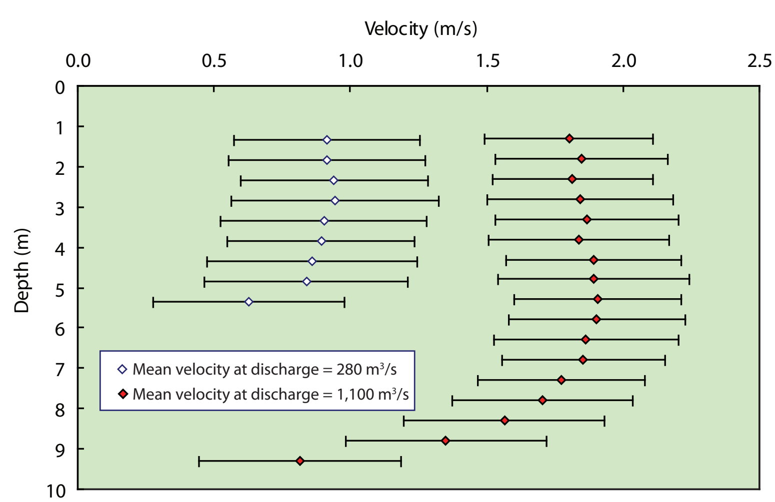

Figure 4. Average flow velocity in a pool section of the Colorado River in Grand Canyon near river mile 30 during low discharge and the controlled release of March 2008. The error bars represent the total range of turbulent fluctuations of velocity measured in the flow. Mean velocity in the pools almost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

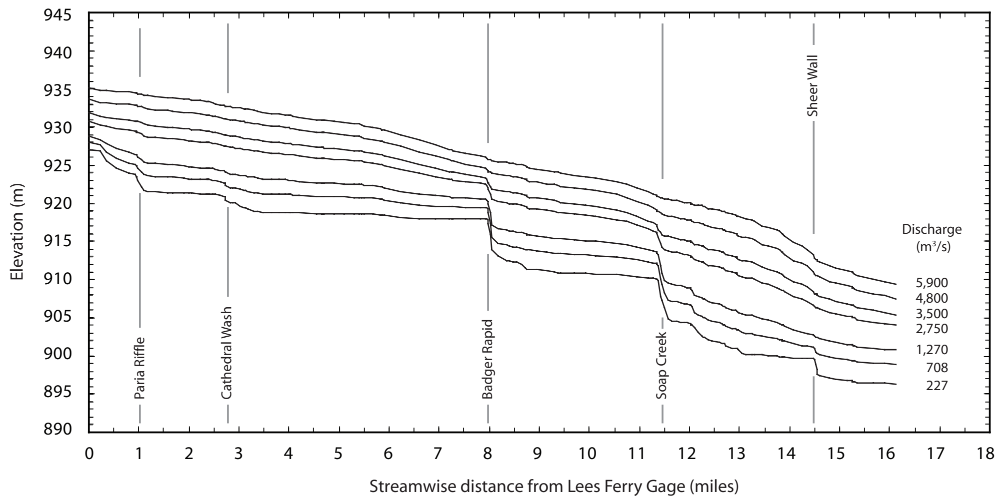

Figure 5. Predicted water-surface elevation profiles of the Colorado River downstream from Lees Ferry for discharges ranging from 227 m^3/s to 5,900 m^3/s. Badger Rapid and Soap Creek Rapids, both large rapids, completely drown out at a discharge of 4,800 m^3/s.

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

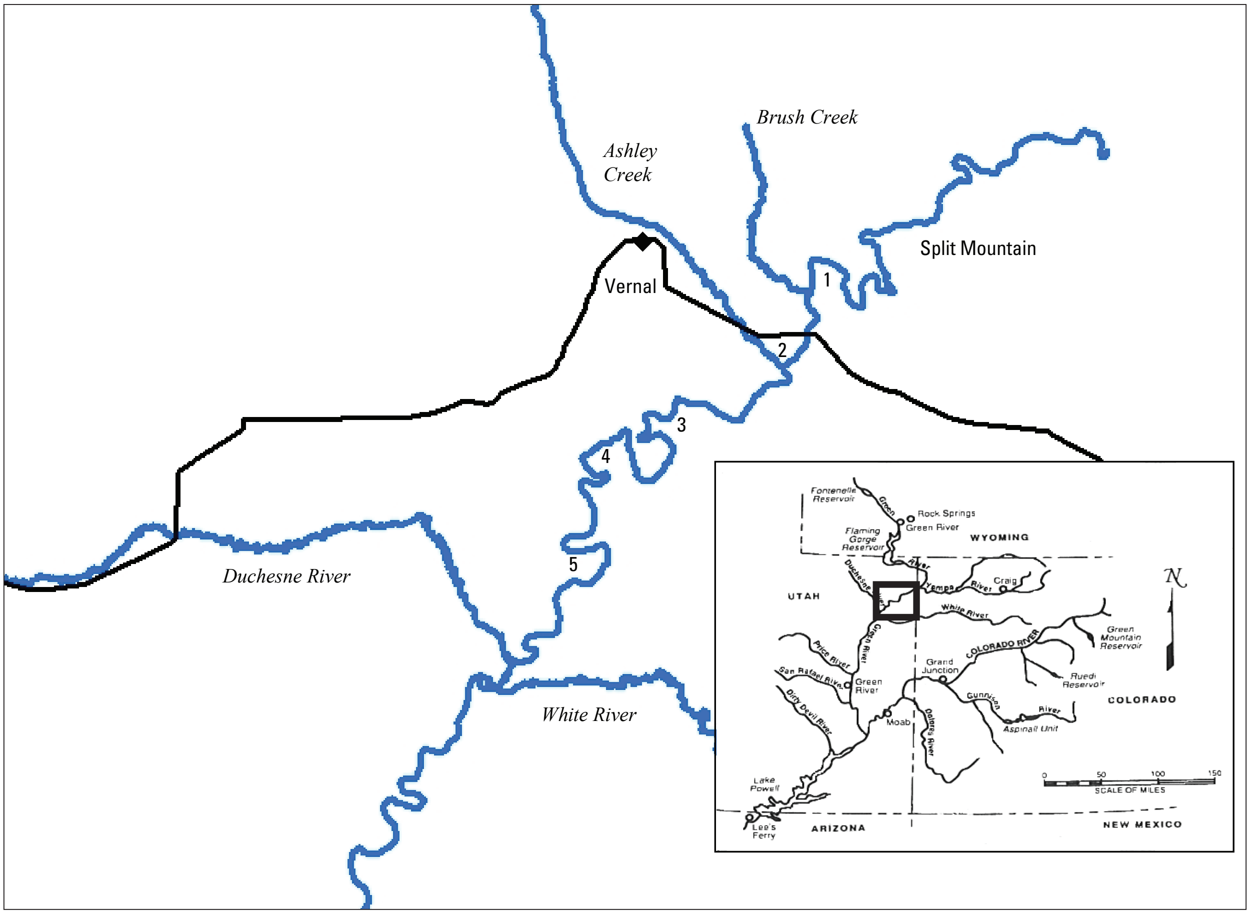

Figure 1. The middle Green River study area and flood-plain sites. 1 = Thunder Ranch, 2 = Stewart Lake, 3 = Bonanza Bridge, 4 = Stirrup, and 5 = Leota. Box in inset shows the extent of the project area within the larger regional area.

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

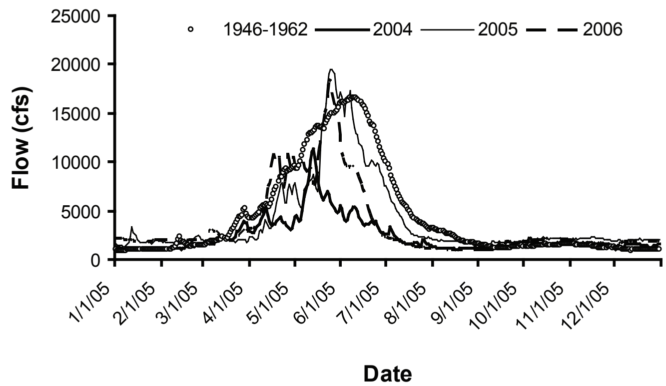

Figure 2. Mean daily average flows for the Green River near Jensen, Utah (station 09261000) for the study period, 2004–2006. Mean daily average flows for the period 1946–1962 (pre-Flaming Gorge Dam) are shown for comparison.

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]

		Thunder Ranch		Stewart Lake		Stirrup		Leota	
2005 release	River flow (ft^3/s)	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.

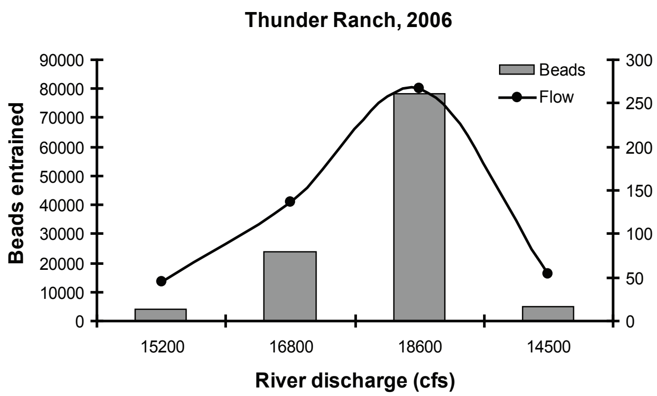

Figure 4. Bead and flow entrainment at Thunder Ranch in 2006 at four flows sampled: three on the ascending limb/peak and the last on the descending limb of the spring hydrograph.

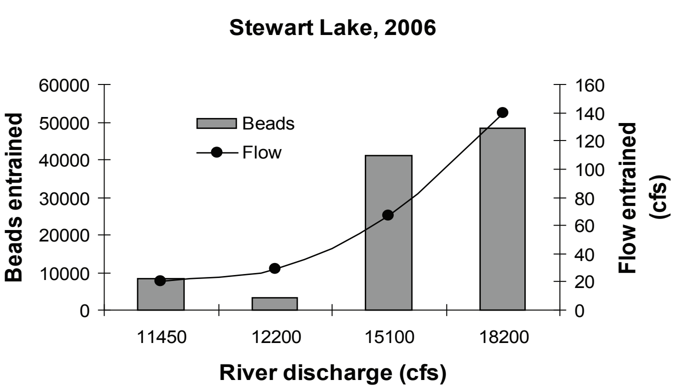

Figure 5. Bead and flow entrainment at Stewart Lake in 2006 at four flows sampled, all on the ascending limb or peak of the spring hydrograph.

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.

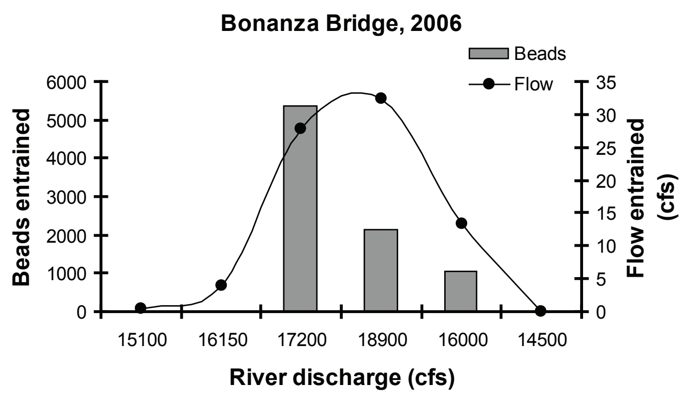

Figure 6. Bead and flow entrainment at Bonanza Bridge in 2006 at three flows sampled: two on the ascending limb/peak and the last on the descending limb of the spring hydrograph.

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).

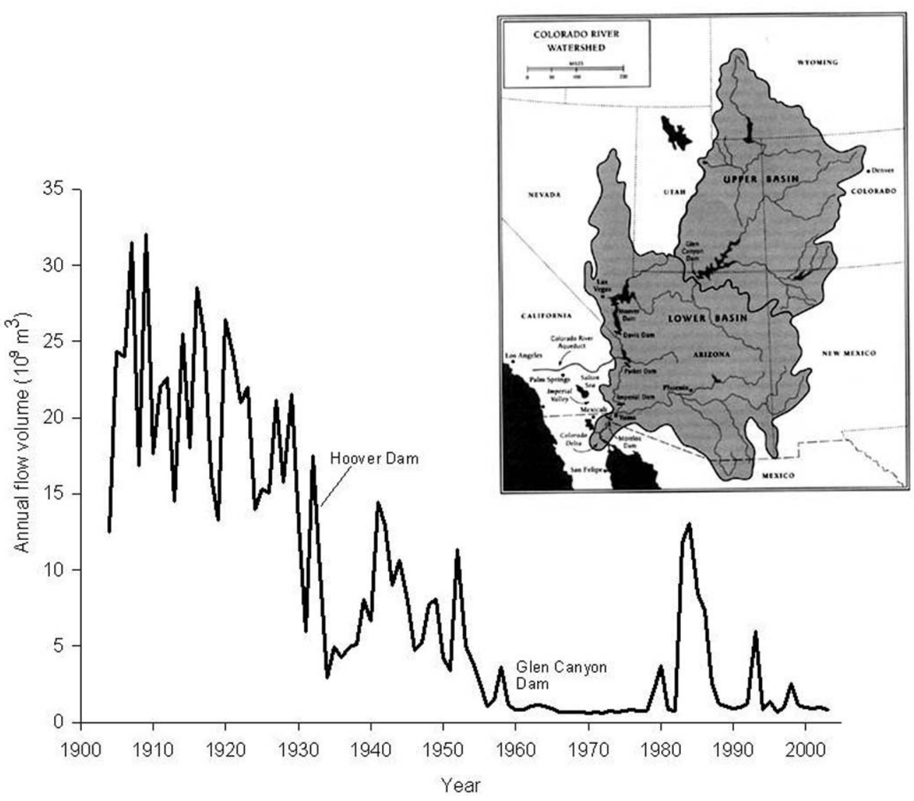

Figure 1. Colorado River annual flow volume (10^9 m^3) below Yuma Main Canal at Yuma, Arizona, (station 09521100), for years 1904–2003. Years where the hydrograph is flat depict the annual flow of 1.8 × 10^9 m^3 required by the 1944 Mexican water treaty.

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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.

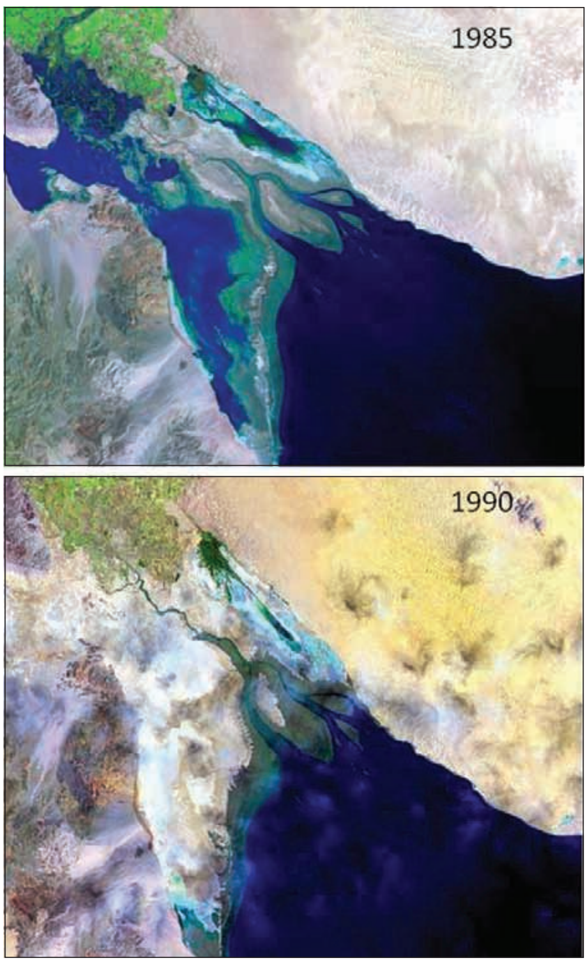

Figure 2. Landsat images of the Colorado River delta, illustrating the extensive wetland habitat created by the Colorado River. These photographs of the Colorado River mouth and the upper Gulf of California were taken during May of 1985 and 1990. In 1984, an abnormal snowmelt triggered a release of river 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.

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 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?

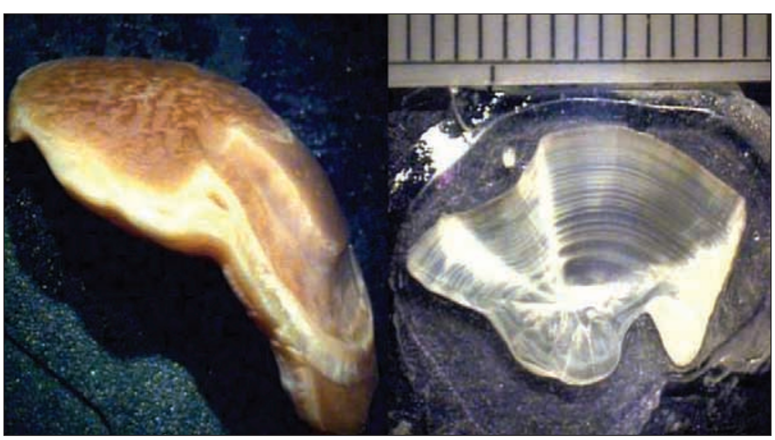

Figure 4. A totoaba otolith (≈5,000 years old) from an aboriginal shell midden along the coast of Sonora, Mexico. Annual rings can be seen in cross section. The oxygen isotopes embedded in the calcium carbonate of the growth increments document when these fish are residing in estuarine habitat created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

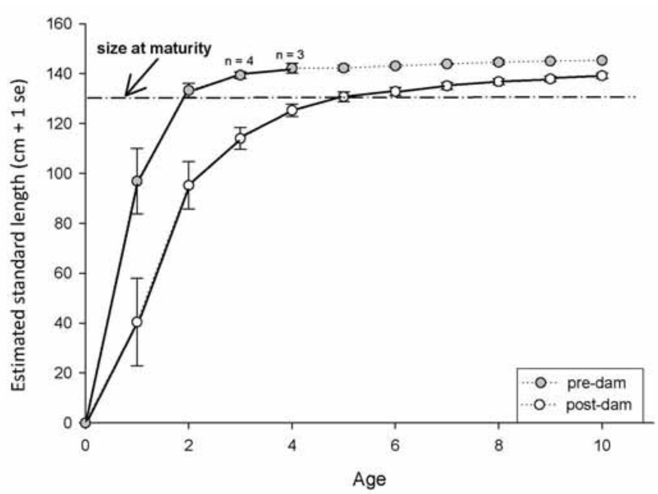

Figure 5. The growth curve and body size of pre-dam and post-dam totoaba, showing slowed growth and delayed maturation of totoaba that live in the absence of river water. The dotted line indicates where sample size equals two (from Rowell and others, 2008a).

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

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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.

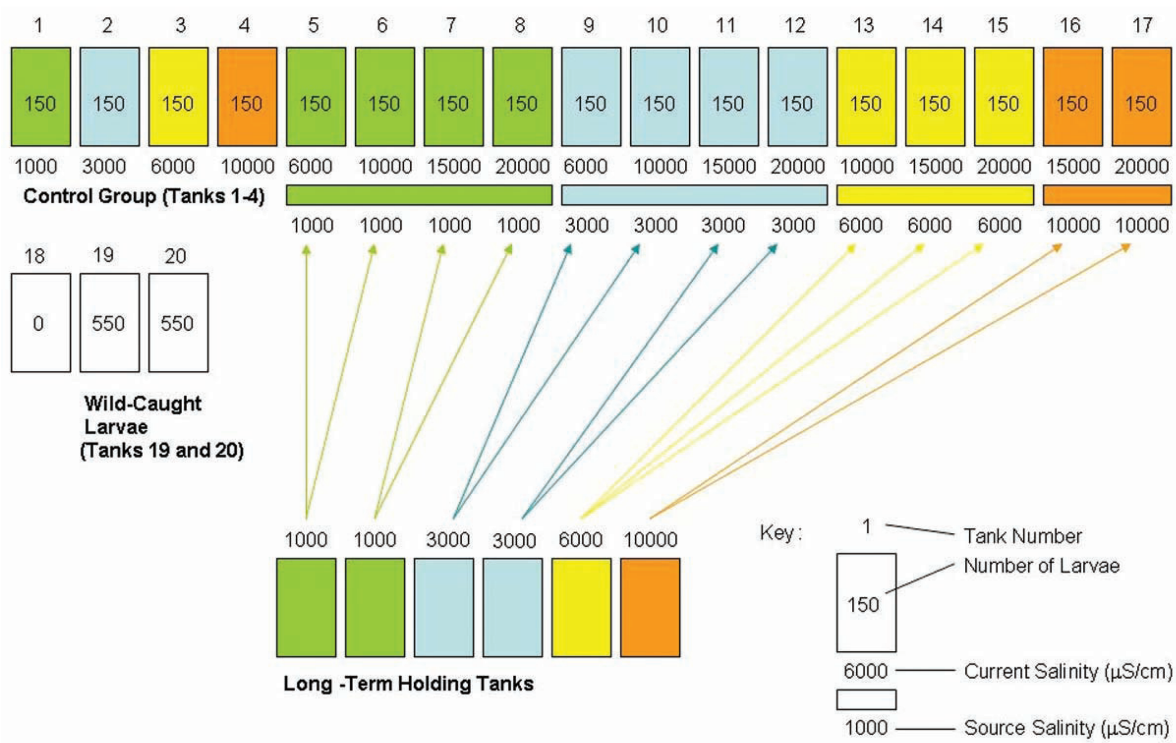

Figure 4. First trial tank setup (prepared by Dr. Mike Horn with modifications by James Stolberg).

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.

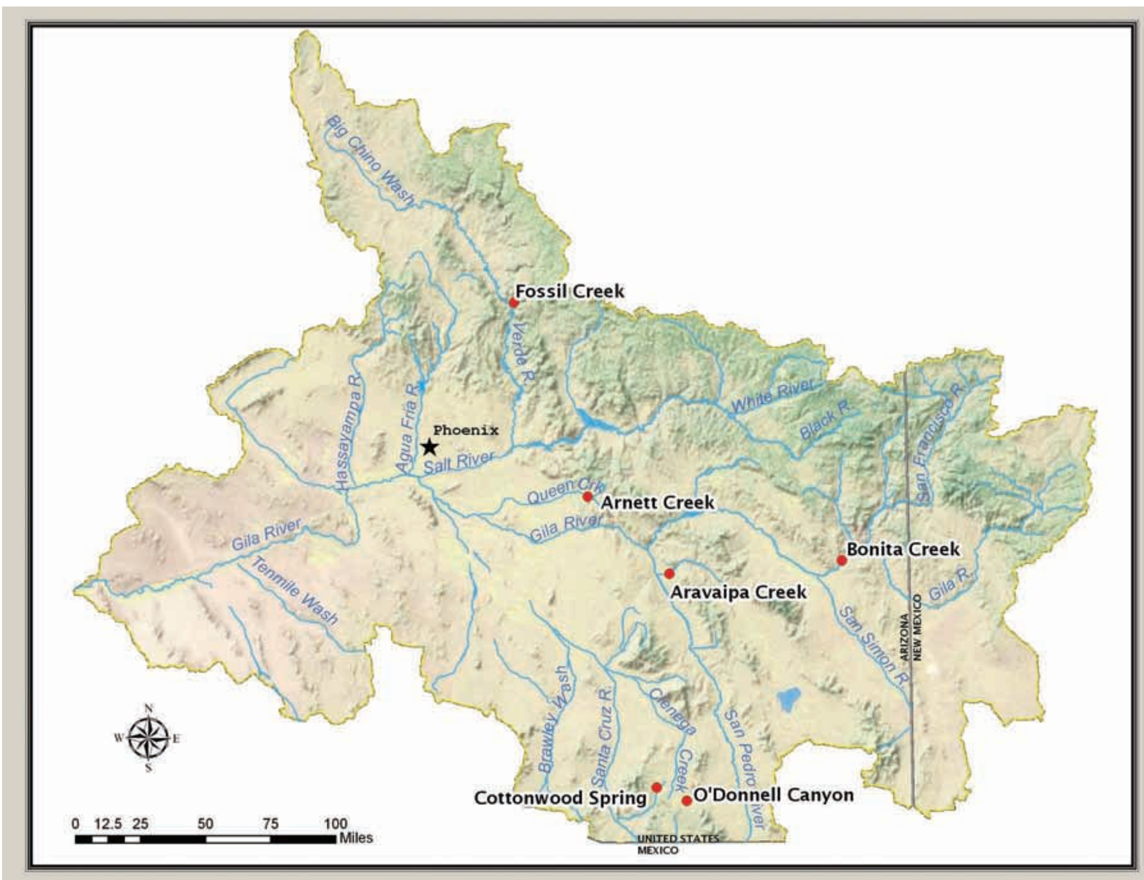

Figure 1. Boundaries of the Gila River Basin, Arizona and New Mexico (exclusive of Sonora), showing major streams and locations of fish barrier projects discussed in the text.

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.

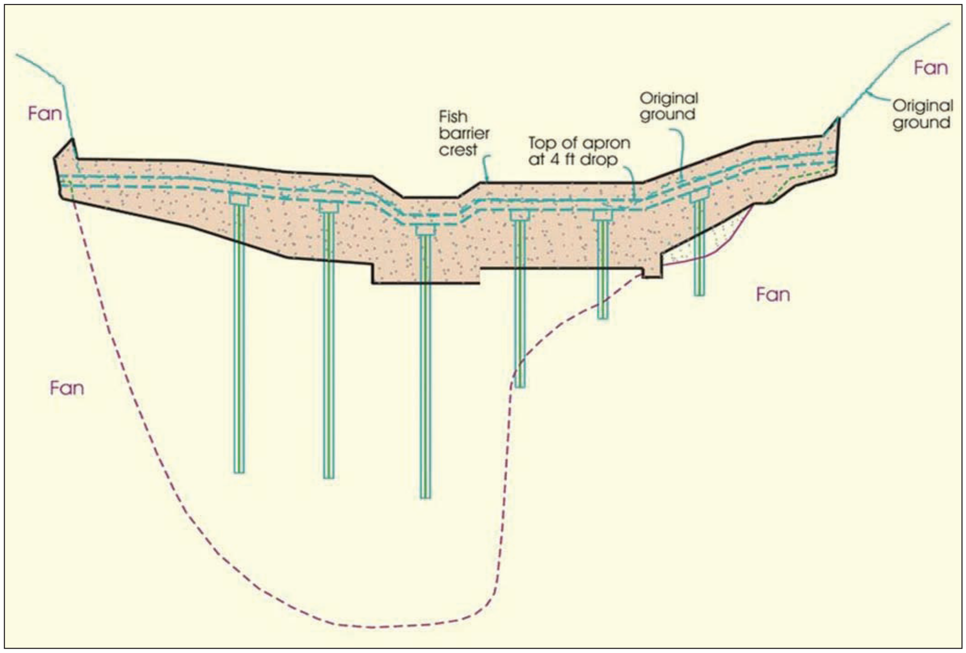

Figure 2. Plan view of the lower fish barrier on Aravaipa Creek, Pinal County, Arizona, showing the placement of concrete piles to stabilize the structure. "Fan" refers to fanglomerate, an accumulation of cemented coarse materials in an alluvial fan.

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,	Sonora sucker					Sonora sucker	Yes
	Roundtail chub					Roundtail chub	Yes
unpub. data, 2009	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.,	Mosquitofish*			1999	1	Sonora sucker	-
unpub. report, 2008				1999	23	Longfin dace	No
				2007	100	Longfin dace	Yes
Bonita Creek	Desert sucker	2008	2008	2008	1	Desert sucker	-
Robinson and others,	Sonora sucker			2008	201	Sonora sucker	Undetermined
unpub. report, 2009	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),	Desert sucker					Desert sucker	Yes
U.S. Fish and Wildlife Service, oral	Longfin dace					Longfin dace	Yes
comm., 2009							
Fossil Creek	Desert sucker	2004	2004	2004	354	Desert sucker	Yes
Weedman and others,	Sonora sucker			2004	204	Sonora sucker	Yes
unpub. report, 2005	Roundtail chub			2004	250	Chub spp.	Yes
Marsh and others,	Headwater chub			2004	906	Speckled dace	Yes
unpub. report, 2009	Speckled dace			2004, 2007	318	Longfin dace	Undetermined
Robinson, A.T.,	Longfin dace			2007–2008	2128	Loach minnow	Undetermined
unpub. report, 2009	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	Gila chub			2001	~20	Gila chub	Yes
Land Management,	Gila topminnow			-	0	Gila topminnow	Yes
oral comm., 2009	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 nonnatives have reinvaded the stream to date (table 1), and only relatively minor barrier maintenance has yet been required.

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

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.

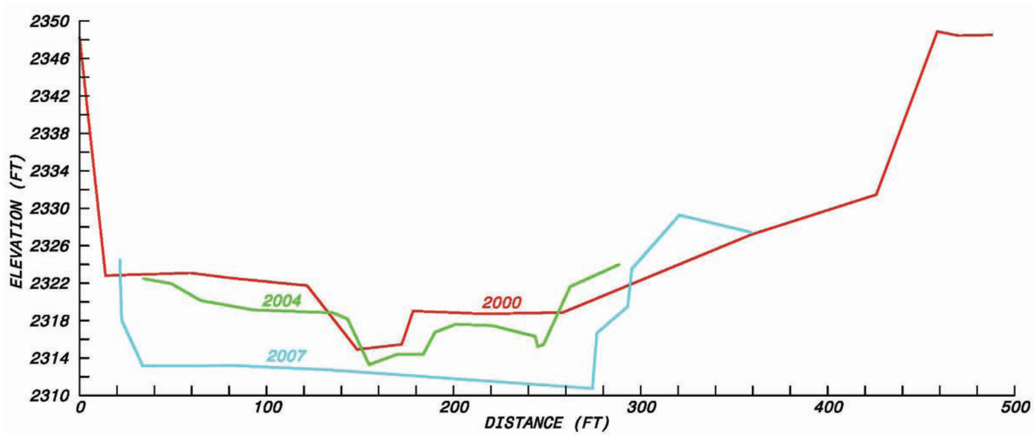

Figure 7. Surveyed cross sections of the stream channel on Aravaipa Creek immediately downstream from the lower fish barrier, showing channel configurations immediately pre-construction (2000), 3 years post-construction (2004), and 1 year following passage of an estimated 50-year flood event (2007).

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.

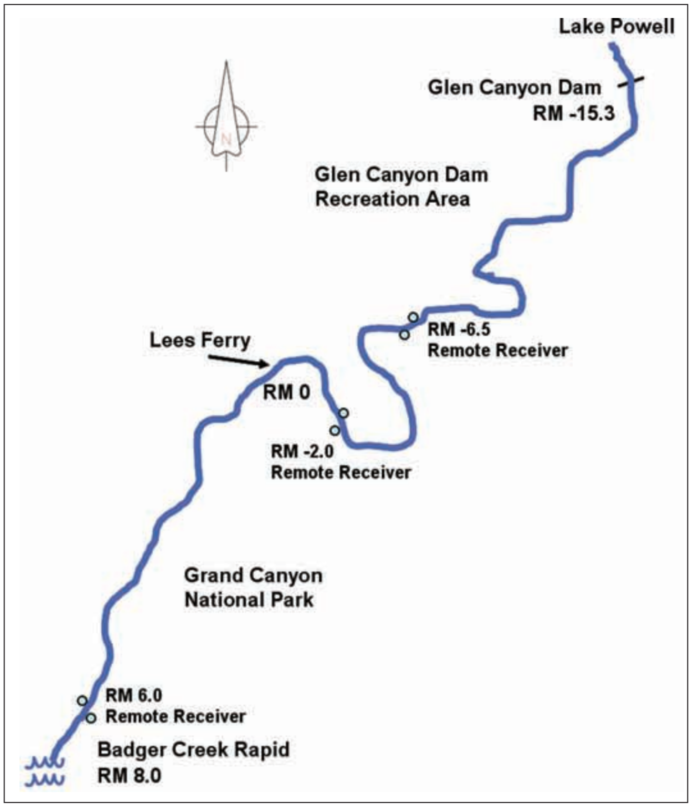

Figure 1. The study area in the Lees Ferry area from Glen Canyon Dam to Badger Creek Rapid in Glen Canyon Dam National Recreation Area and Grand Canyon National Park near Page, AZ. Dots indicate the placement of remote receivers to detect passing acoustic tagged rainbow trout. River mile (RM) is used 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).

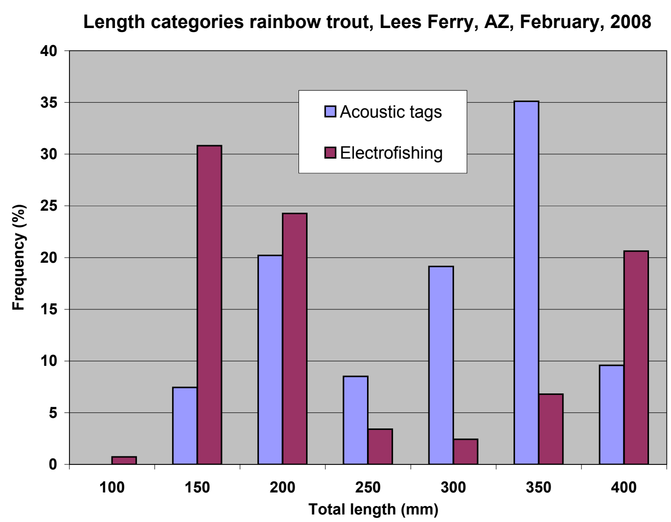

Figure 2. Length frequency of rainbow trout sampled with electrofishing and those that were implanted with acoustic tags in the Lees Ferry area during the March 2008 high-flow experiment. Fish less than 157 mm were too small to carry the acoustic tag, and fish larger than 400 mm were not susceptible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

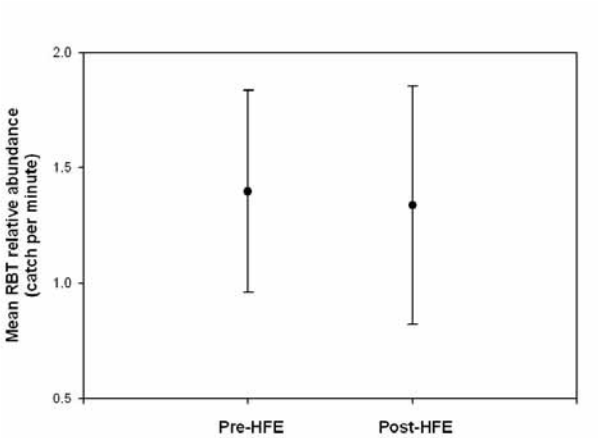

Figure 3. Mean relative abundance (catch per minute of electrofishing) of all size classes of rainbow trout (RBT) captured with electrofishing during pre- (February 28–March 1, 2008) and post-high flow experiment (HFE; March 18–20, 2008) sampling in the Lees Ferry area of the Colorado River, AZ. Bars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

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.

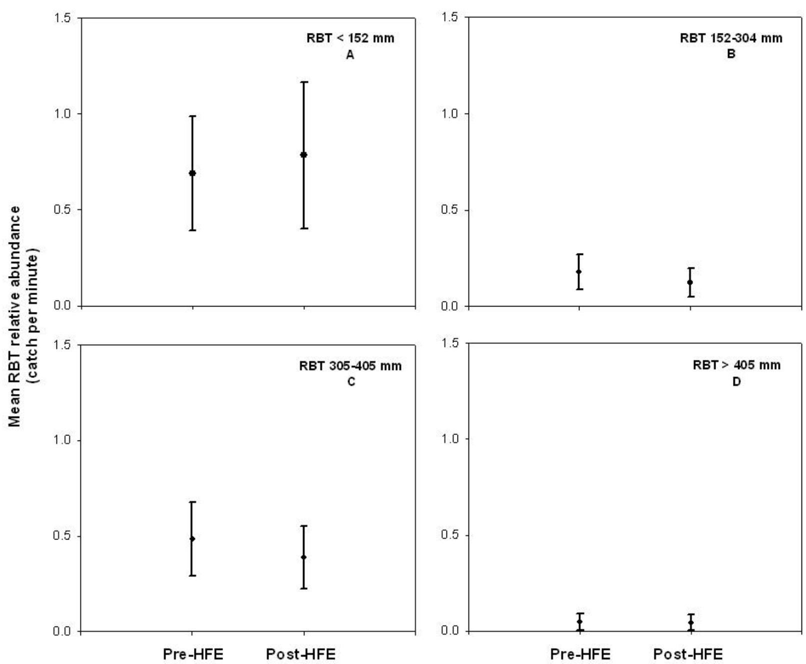

Figure 4. Size-stratified mean relative abundance (catch per minute of electrofishing) of rainbow trout (A) <152 mm total length (TL), (B) 152–304 mm TL, (C) 305–405 mm TL, and (D) >405 mm TL captured with electrofishing during pre- (February 28–March 1, 2008) and post-high flow experiment (HFE; March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

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

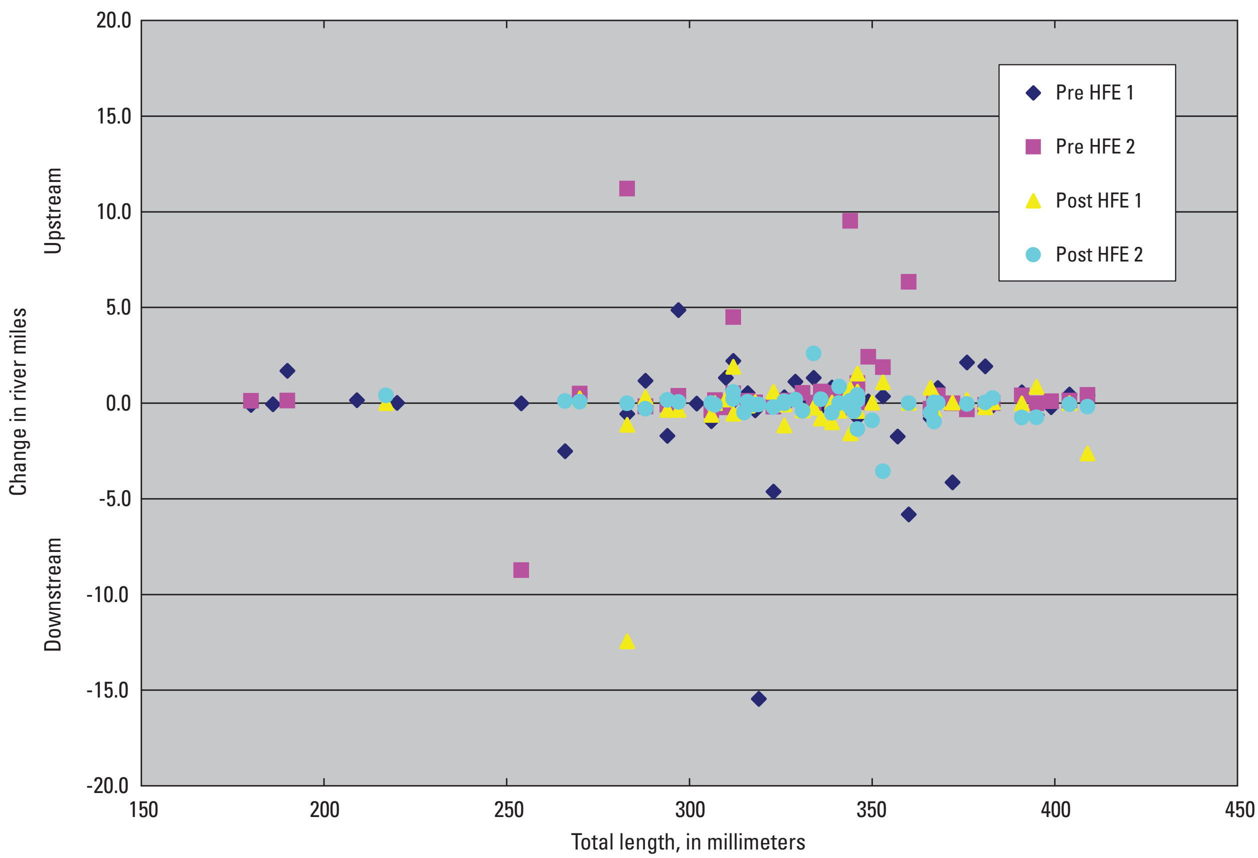

Figure 5. Scatter plot showing individual acoustic tagged rainbow trout movement in the Lees Ferry reach during the two tracking events before (Pre-HFE 1 and 2) and two tracking events after (Post-HFE 1 and 2) the March 2008 high-flow experiment (HFE). Individual tagged fish movement was highly variable 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.

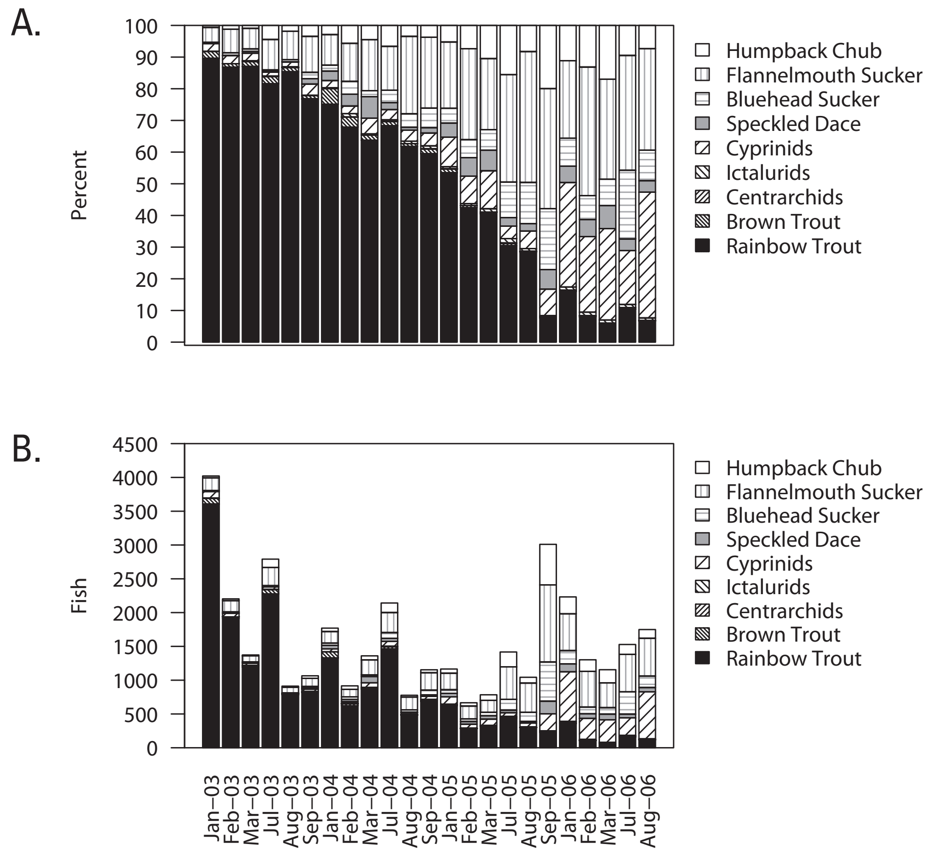

Figure 1. (A) Percent composition and (B) number of fish by species captured with electrofishing in the Little Colorado River inflow reach among months, 2003–2006.

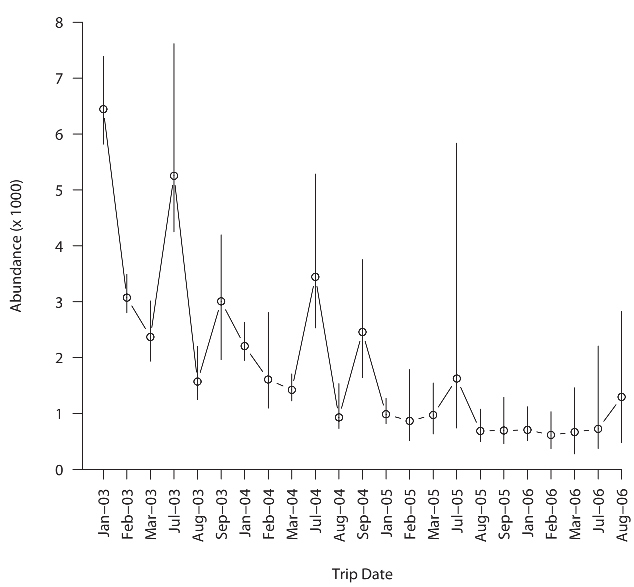

Figure 2. Estimated abundance of rainbow trout in Little Colorado River inflow reach among months, 2003–2006. Error bars represent 95-percent Bayesian credible intervals.

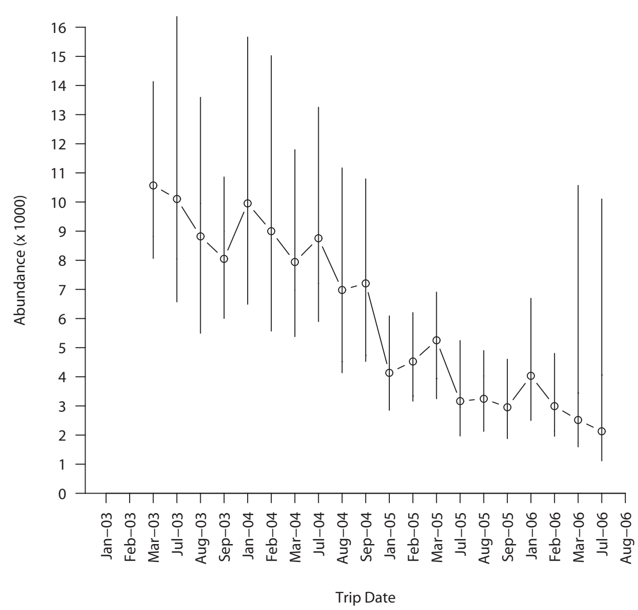

Figure 3. Estimated abundance of rainbow trout in the control reach among months, 2003–2006. Error bars represent 95-percent profile likelihood confidence intervals.

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.

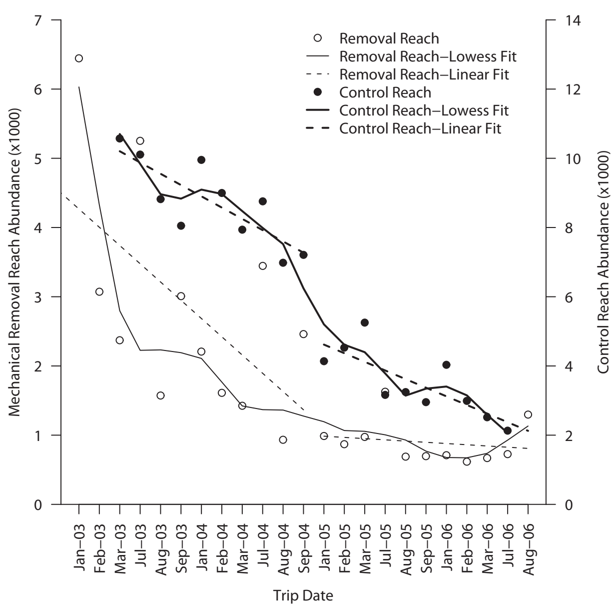

Figure 4. Estimated rainbow trout abundance in both the Little Colorado River inflow and control reaches at the beginning of each trip during 2003–2006. The solid lines represent the locally weighted polynomial regressions (Lowess) fit to each time series. The dashed lines represent linear regressions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

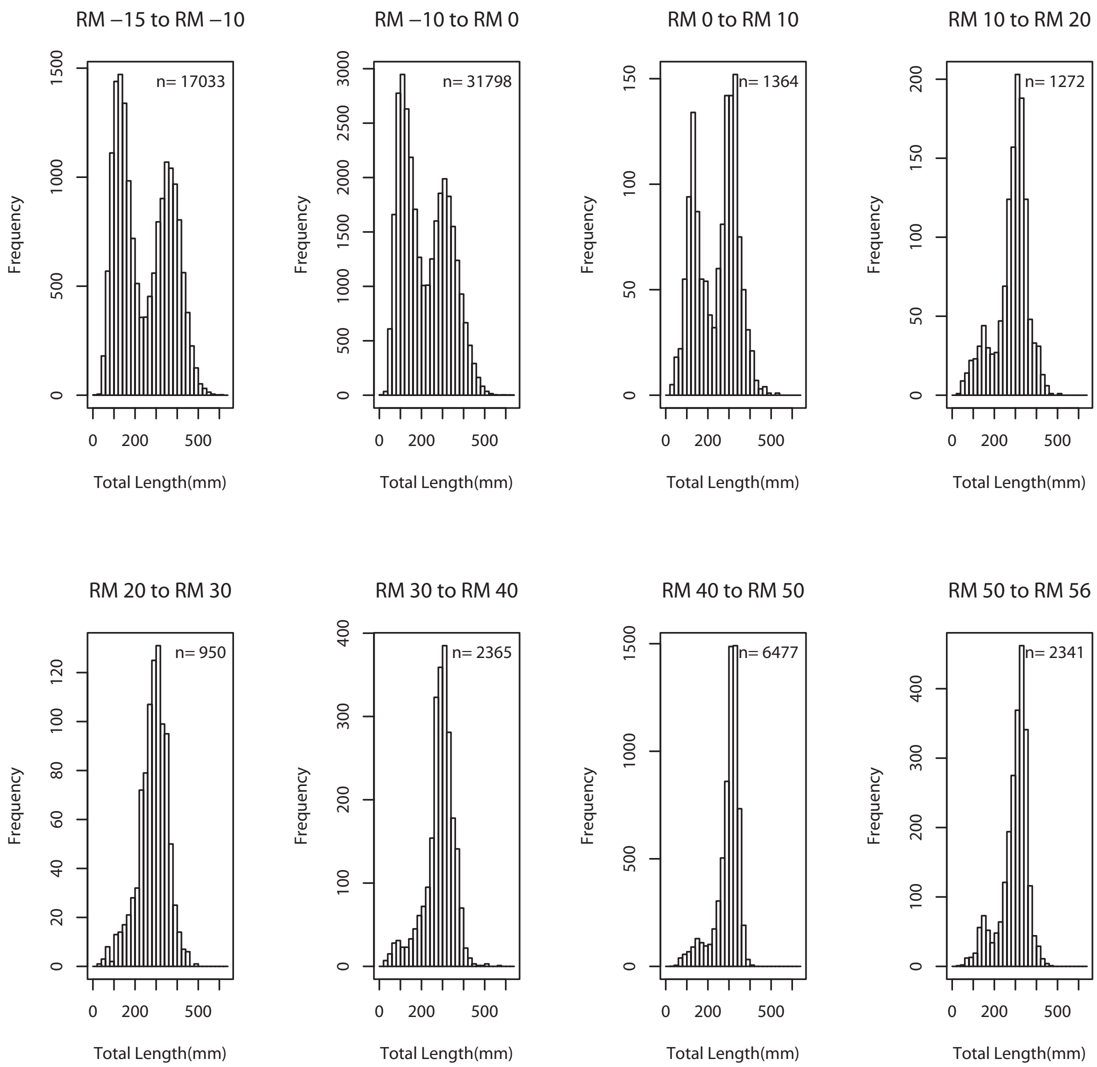

Figure 5. Length frequency distributions of rainbow trout captured during 1991–2004 using electrofishing in the Colorado River from river mile –15 to river mile 56. Each panel represents captures of fish within the identified river segment.

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.

Acknowledgments

We would like to acknowledge the fine work of Arizona Game and Fish staff under the leadership of Clay Nelson for conducting field operations for this study during 2005–2006. Additionally, we thank the many Colorado River boatman, technicians, and biologists whose tireless efforts allowed this study to come to fruition. This study was authorized by the Glen Canyon Dam Adaptive Management Program.

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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

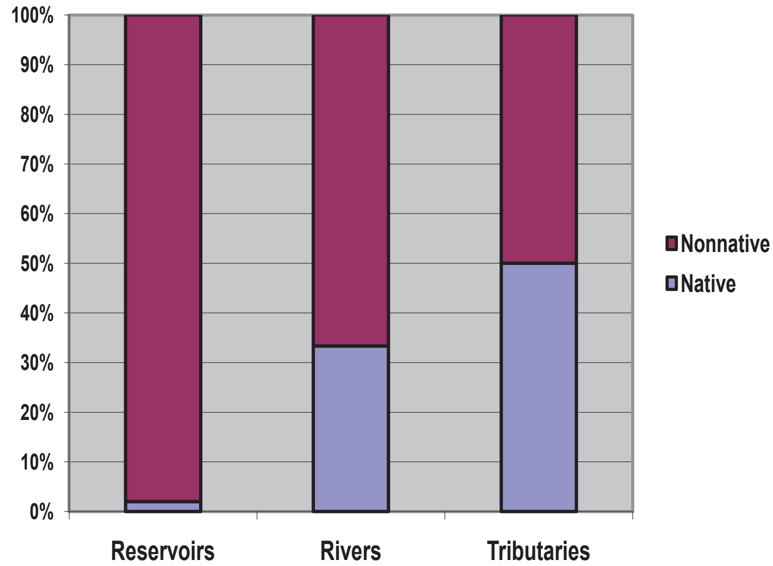


Figure 1. Ratio of number of native and nonnative fish species in Colorado River park reservoirs, rivers, and tributaries.

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.

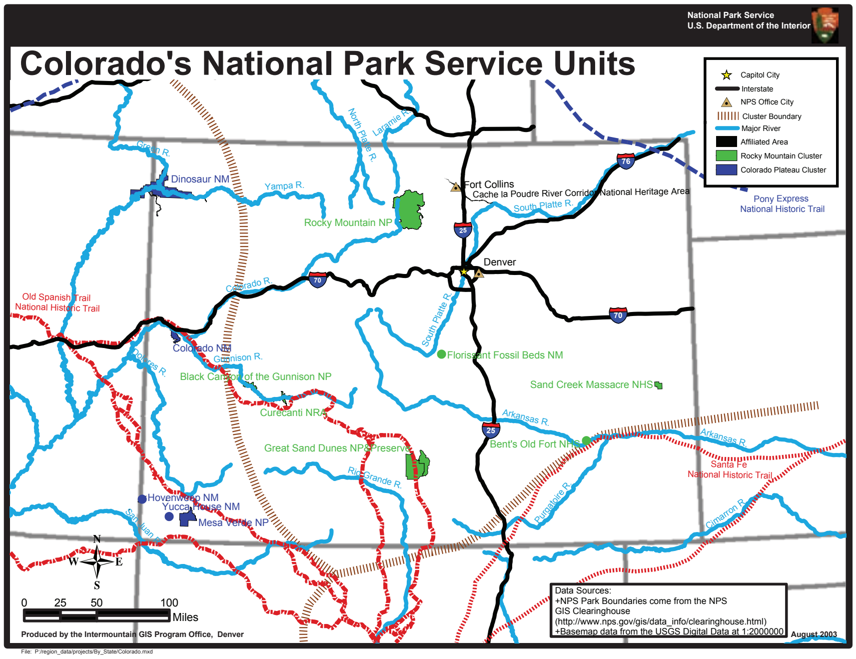
Curecanti National Recreation Area

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)



(b)

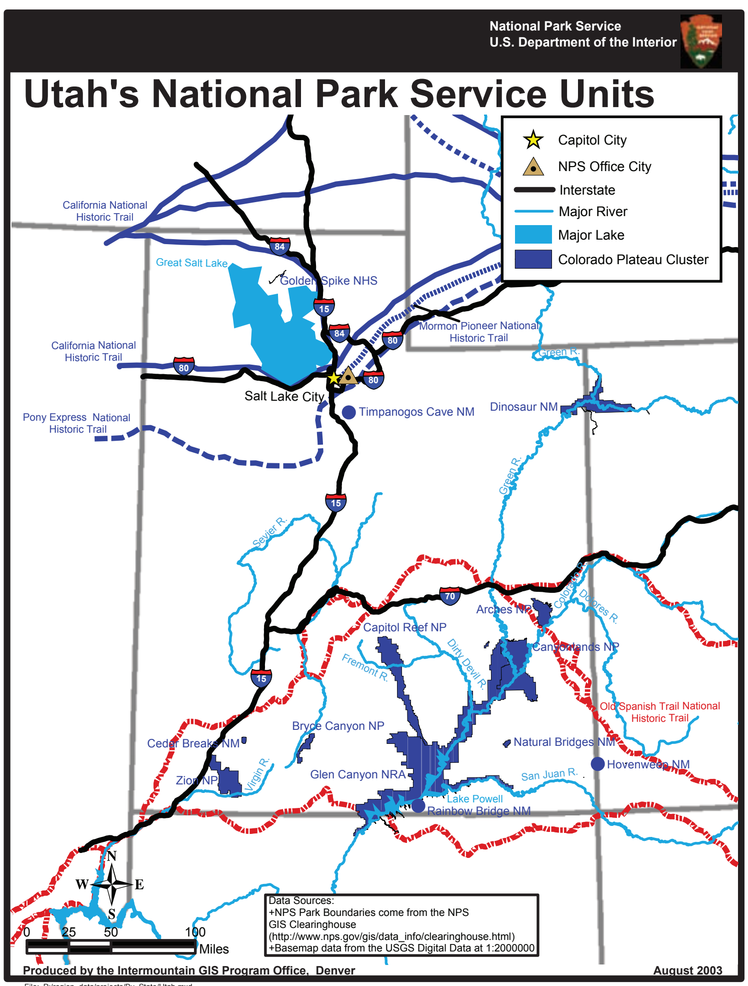


Figure 2. National Park units along the Colorado River system in (a) Colorado, (b) Utah, and (c) Arizona/Nevada.

(c)

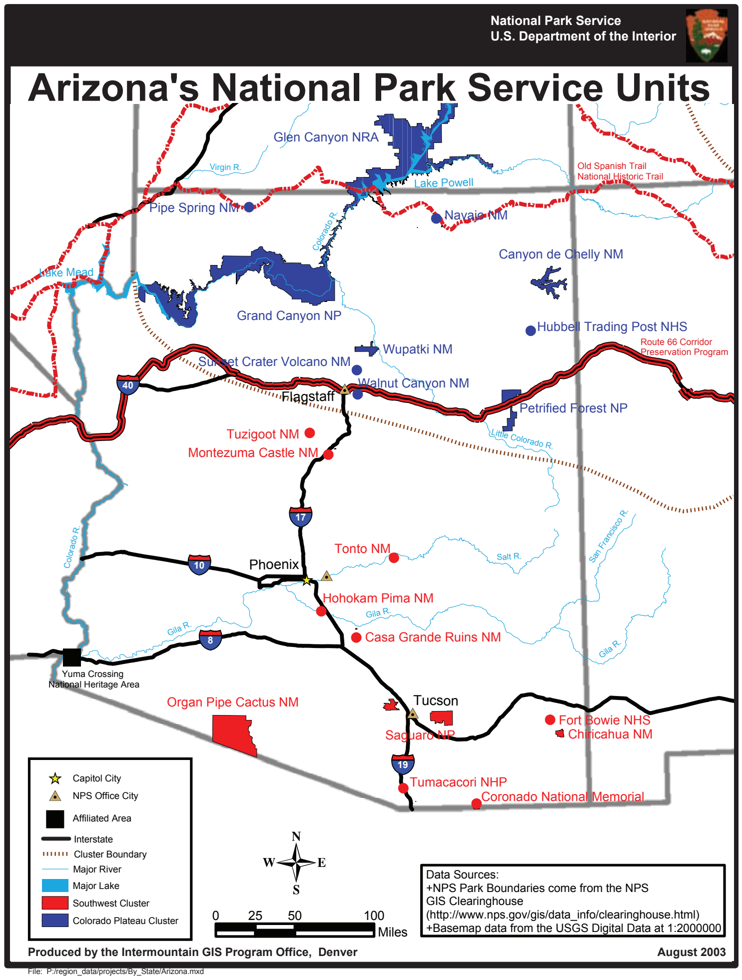


Figure 2. (continued) National Park Units along the Colorado River system in (a) Colorado, (b) Utah, and (c) Arizona/Nevada.

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

By Allen W. Calvert,[1] Susan C. Broderick,[2] and Theresa M. Olson[1]

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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[2] Bureau of Reclamation, PO Box 25007, Denver, CO 80225.

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

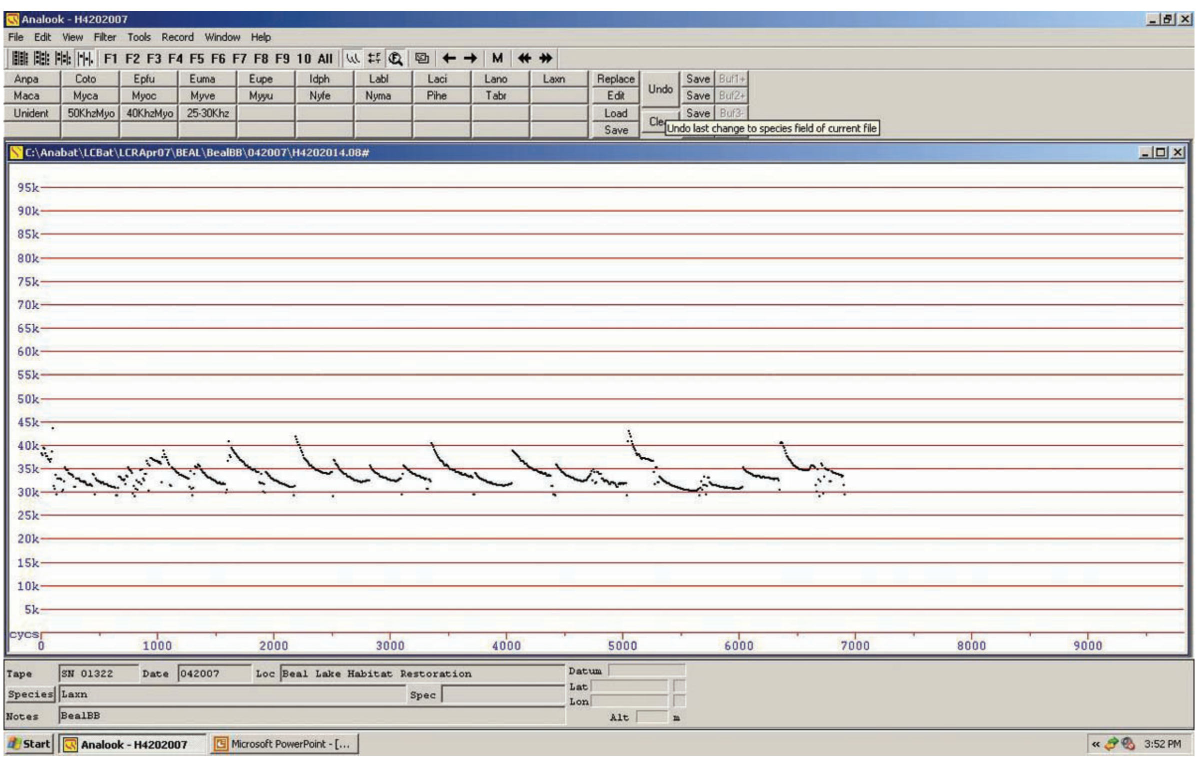

Figure 6. A screen shot of an Anabat™ call file displayed on the analyzing software.

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

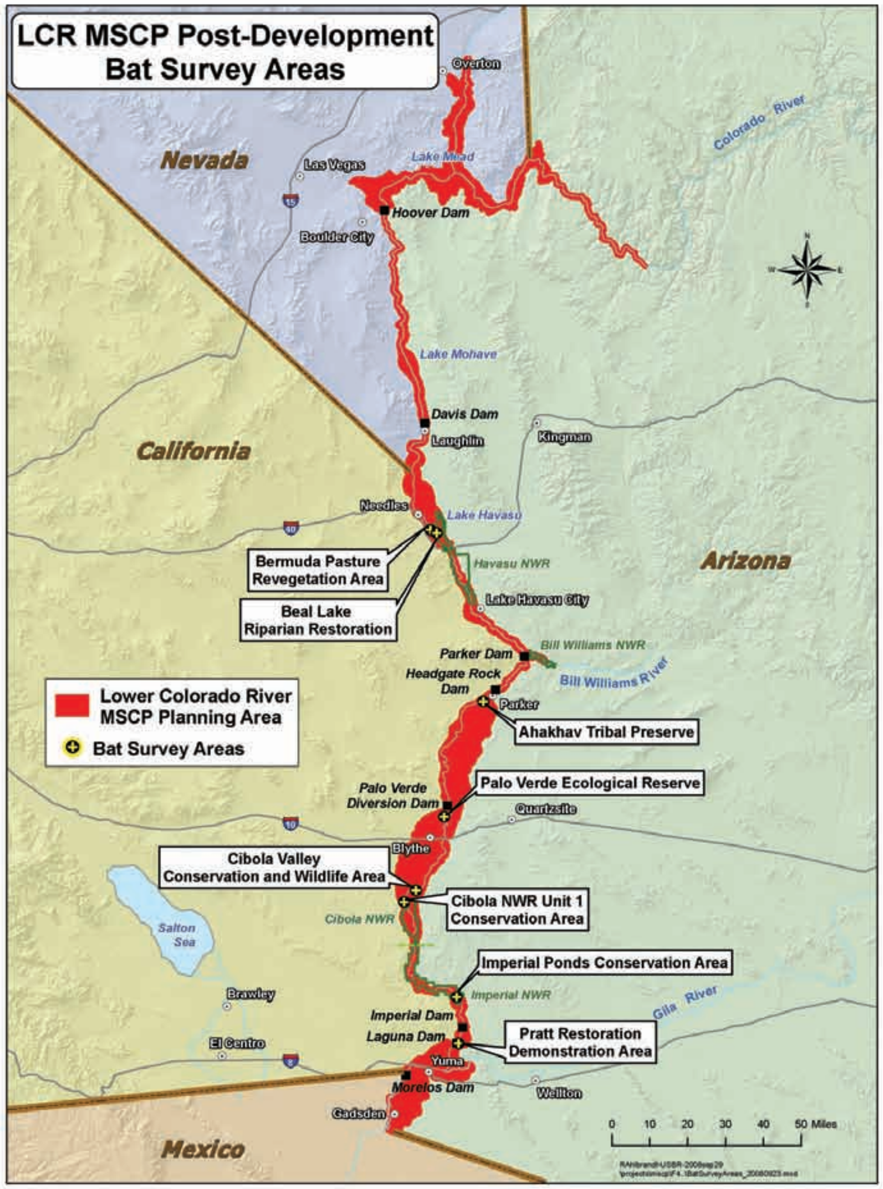

Figure 7. Bat monitoring locations at habitat creation areas along the lower Colorago River. Note that the survey area *Bermuda Pasture Revegetation Area* was not included in acoustic surveys.

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.

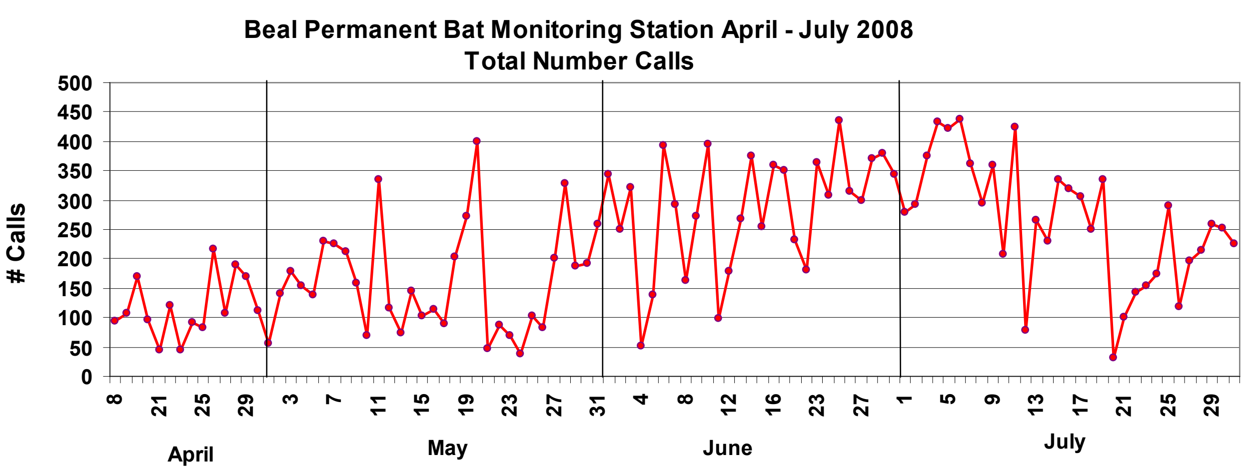

Figure 14. Variation in total bat activity at the Beal Restoration site from April to July 2008.

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.

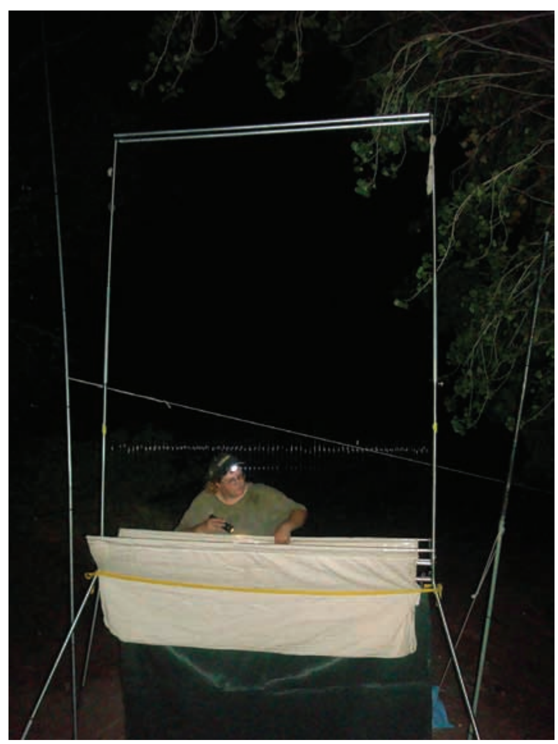

Figure 16. A harp trap set within a narrow opening at Cibola National Wildlife Refuge.

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.

Acknowledgments

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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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[4] Arizona Game and Fish Department, Research Branch, 1600 N. Page Springs Road, Cornville, AZ 86325.

[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

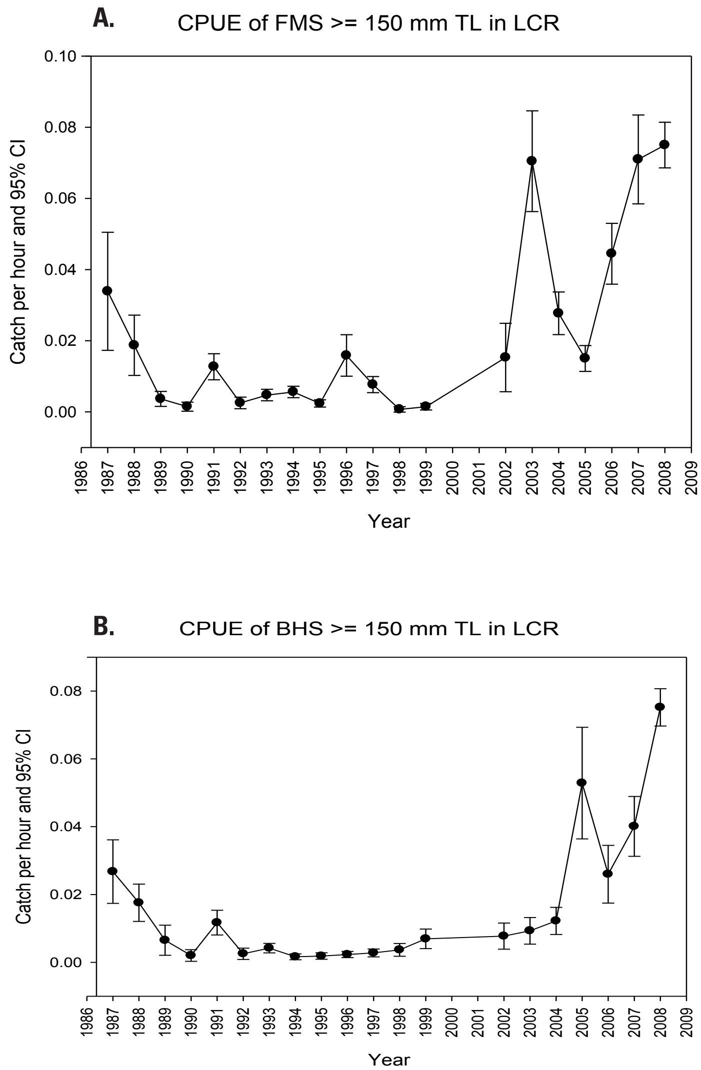

Figure 2. Mean catch per hour (CPUE) of (A) flannelmouth sucker (FMS) ≥150 millimeters (mm) total length (TL) and (B) bluehead sucker (BHS) ≥150 mm TL in the Little Colorado River (LCR) during Arizona Game and Fish Department lower 1,200-meter monitoring, 1987–2008.

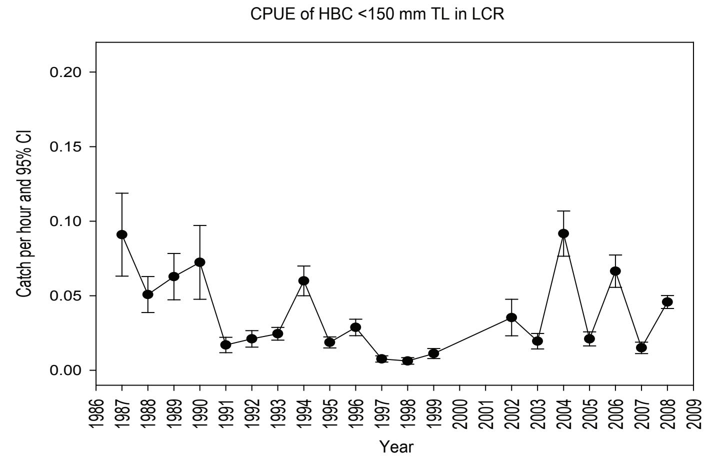

Figure 3. Mean catch per hour (CPUE) of humpback chub (HBC) <150 millimeters (mm) total length (TL) in the Little Colorado River (LCR) during Arizona Game and Fish Department lower 1,200-meter monitoring, 1987–2008.

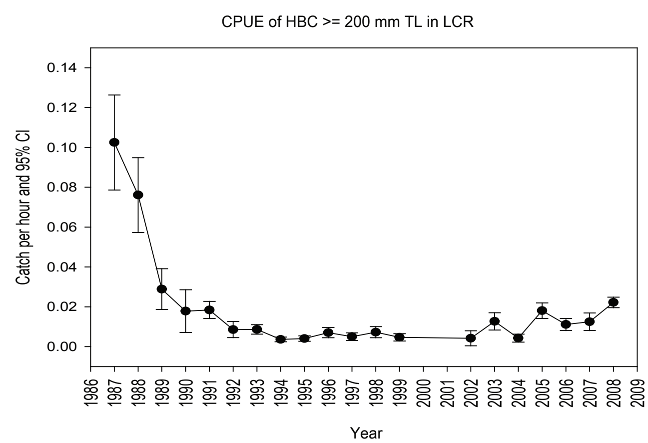

Figure 4. Mean catch per hour (CPUE) of humpback chub (HBC) ≥200 millimeters (mm) total length (TL) in the Little Colorado River (LCR) during Arizona Game and Fish Department lower 1,200-meter monitoring, 1987–2008.

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.

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

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

where:

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

$V[N^*]$ 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.

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

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

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.

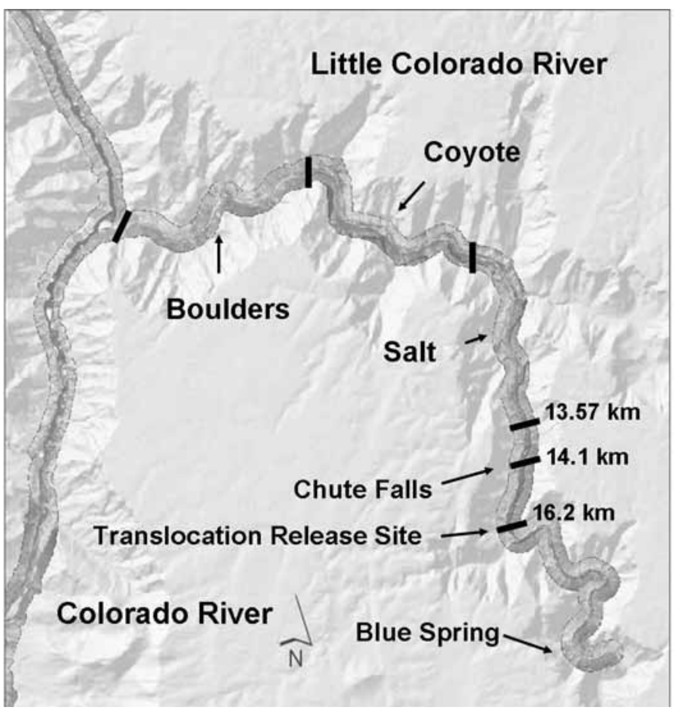

Figure 3. Study sites in Little Colorado River showing: (1) Salt, Coyote, and Boulders reaches (study areas of spring and fall mark-recapture efforts), (2) release site of translocated humpback chub at 16.2 km, and (3) the two reaches (lower and upper) of the Chute Falls mark-recapture efforts. (Note: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

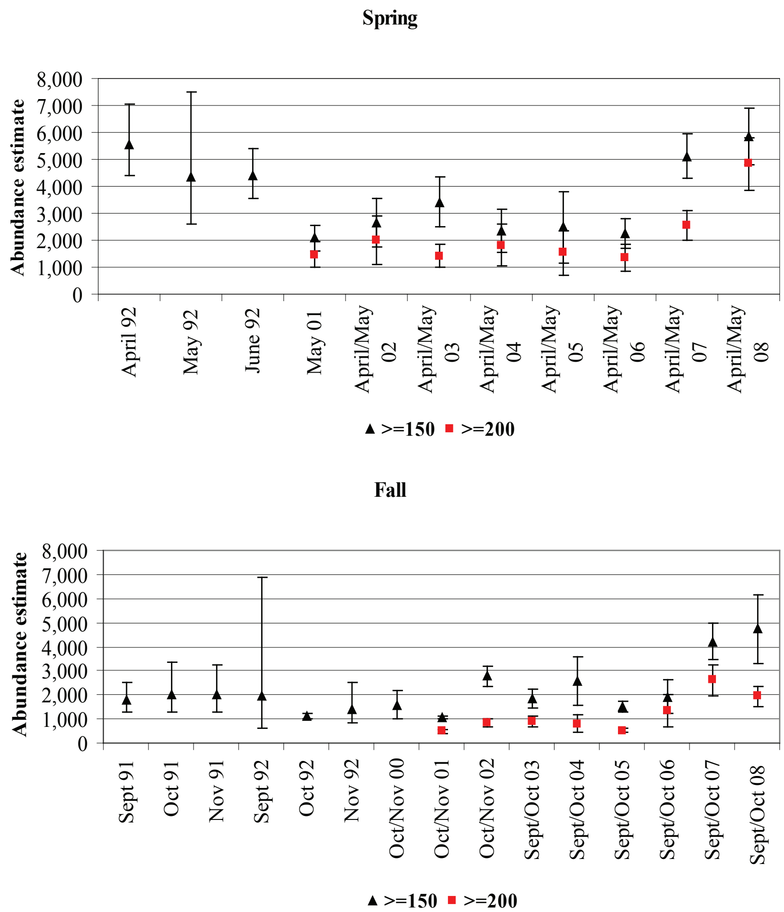

Figure 5. Spring and fall abundance estimates (with 95 percent confidence level intervals) of humpback chub ≥150 mm, and ≥200 mm in the lower 13.57 km of the Little Colorado River. All pre-2000 estimates are from Douglas and Marsh (1996). All other estimates from Van Haverbeke and Stone (2009).

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

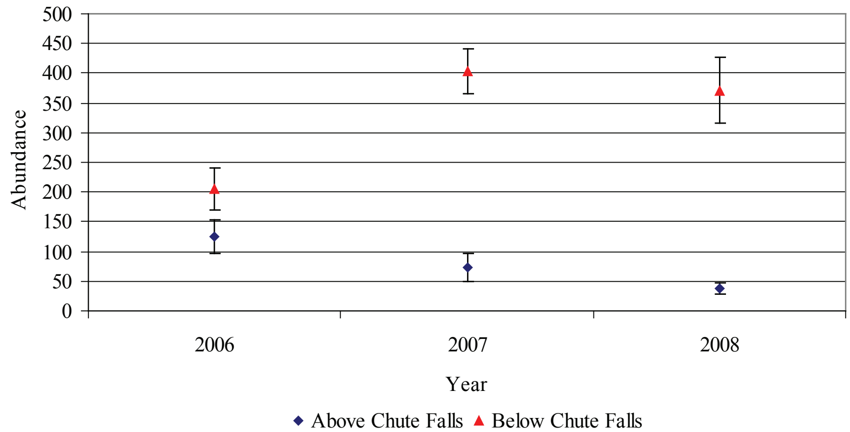

Figure 7. Abundance estimates of humpback chub ≥200 mm immediately below Chute Falls (13.57 to 14.1 km) and above Chute Falls (14.1 to 18.2 km), Little Colorado River.

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

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).

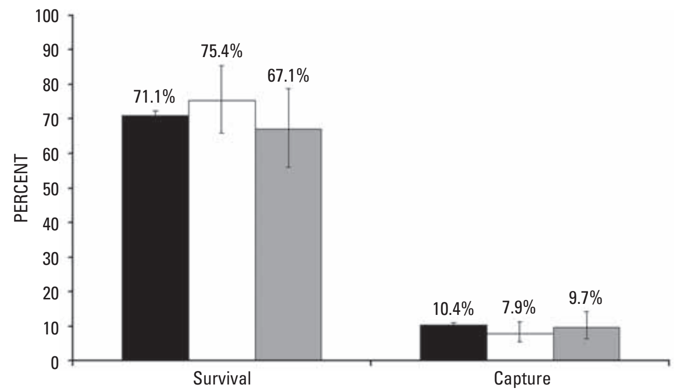

Figure 3. PIT-tagged wild (black), repatriated adults more than 450 mm TL (open), and repatriated adults more than 500 mm TL (grey) annual survival and capture in Lake Mohave.

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

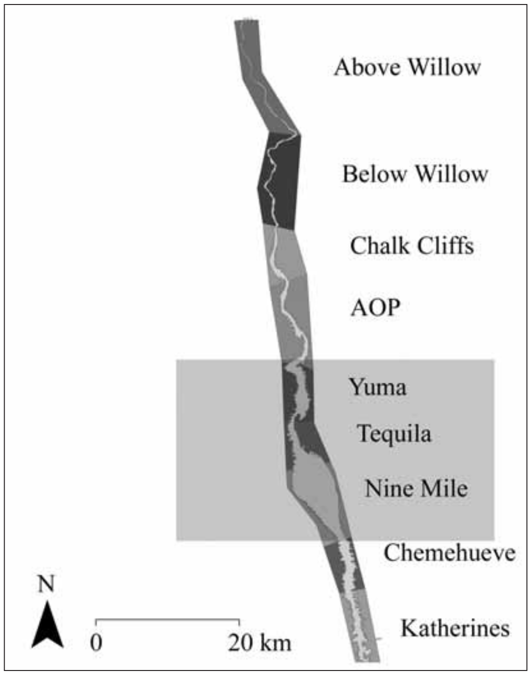

Figure 4. Lake Mohave and primary repatriate capture zones: Yuma, Tequila, and Nine Mile.

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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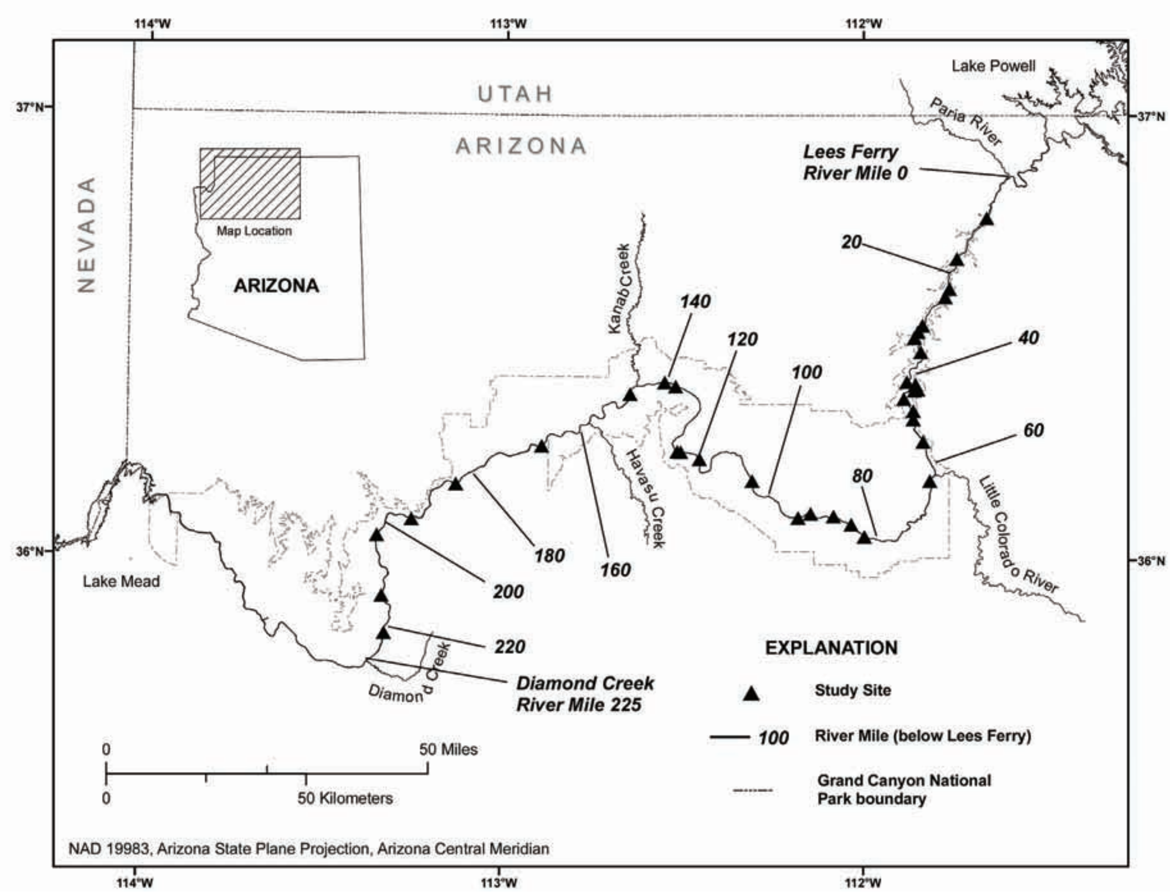

Figure 1. The Colorado River corridor between Glen Canyon Dam and the western boundary of Grand Canyon National Park. Study site locations are indicated with triangles. The use of river mile has a historical precedent and provides a reproducible method for describing locations along the Colorado River 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

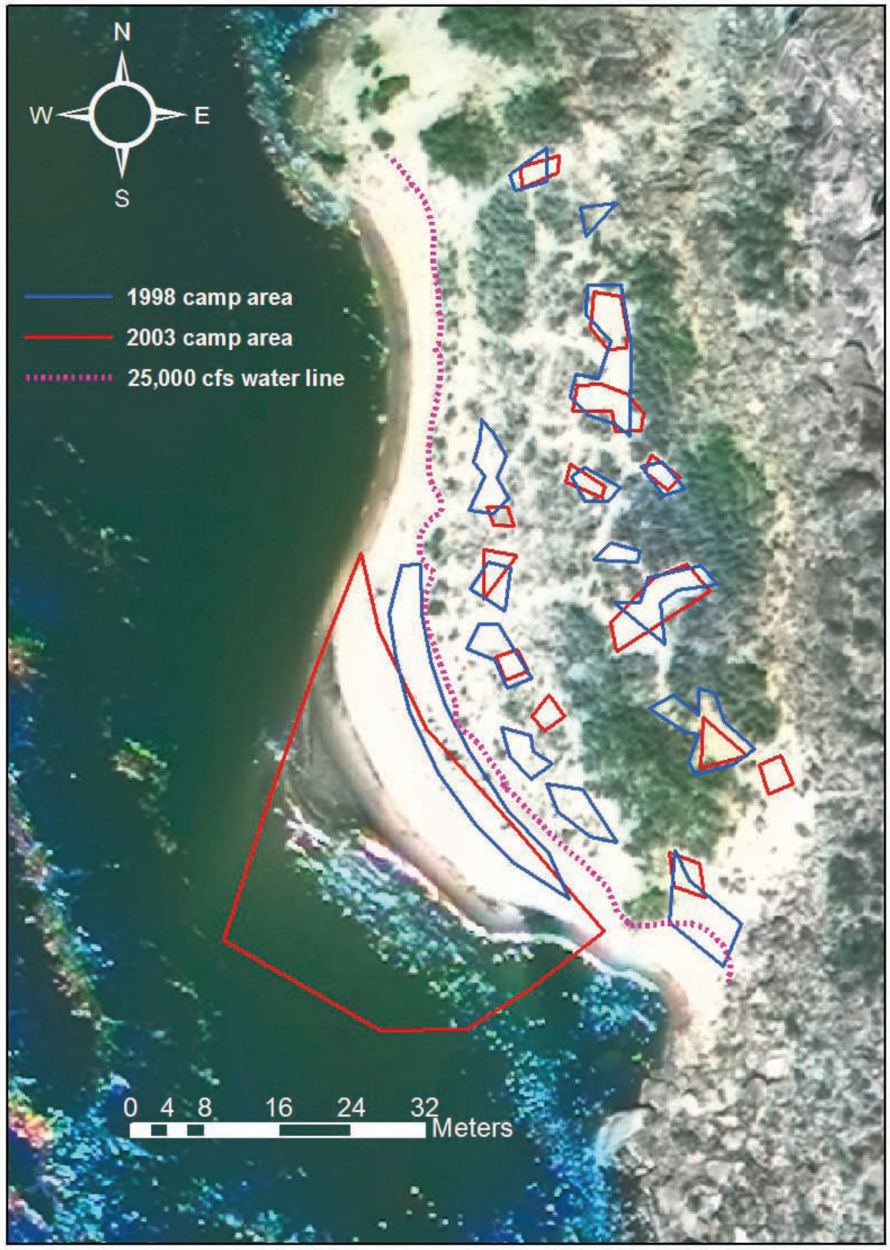

Figure 2. Aerial photograph taken in May 2002 of the 119.4-mile study site at a discharge of approximately 8,000 cubic feet per second (ft^3/s). The campsite area polygons surveyed in 1998 and 2006 are shown by blue and red lines, respectively. Also shown is the location of the 25,000 ft^3/s stage elevation 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).

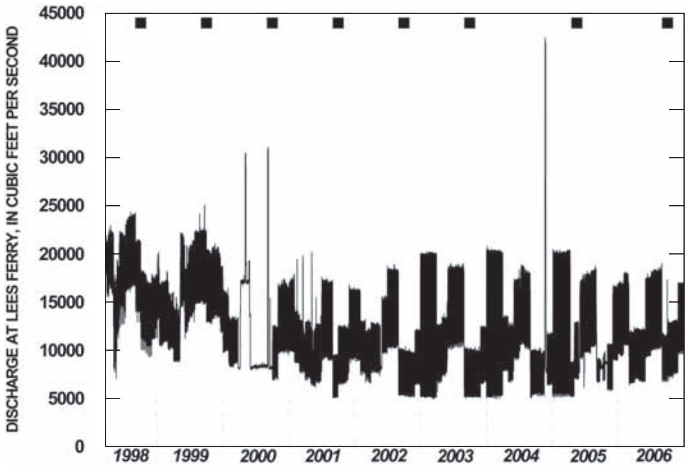

Figure 3. Daily mean discharge hydrograph from the USGS streamgaging station at the Colorado River near Lees Ferry (09380000) during the period of study. The squares indicate the survey date. Note the daily and seasonal fluctuations in flow volume, the May and September 31,000 cubic feet per second (ft^3/s)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

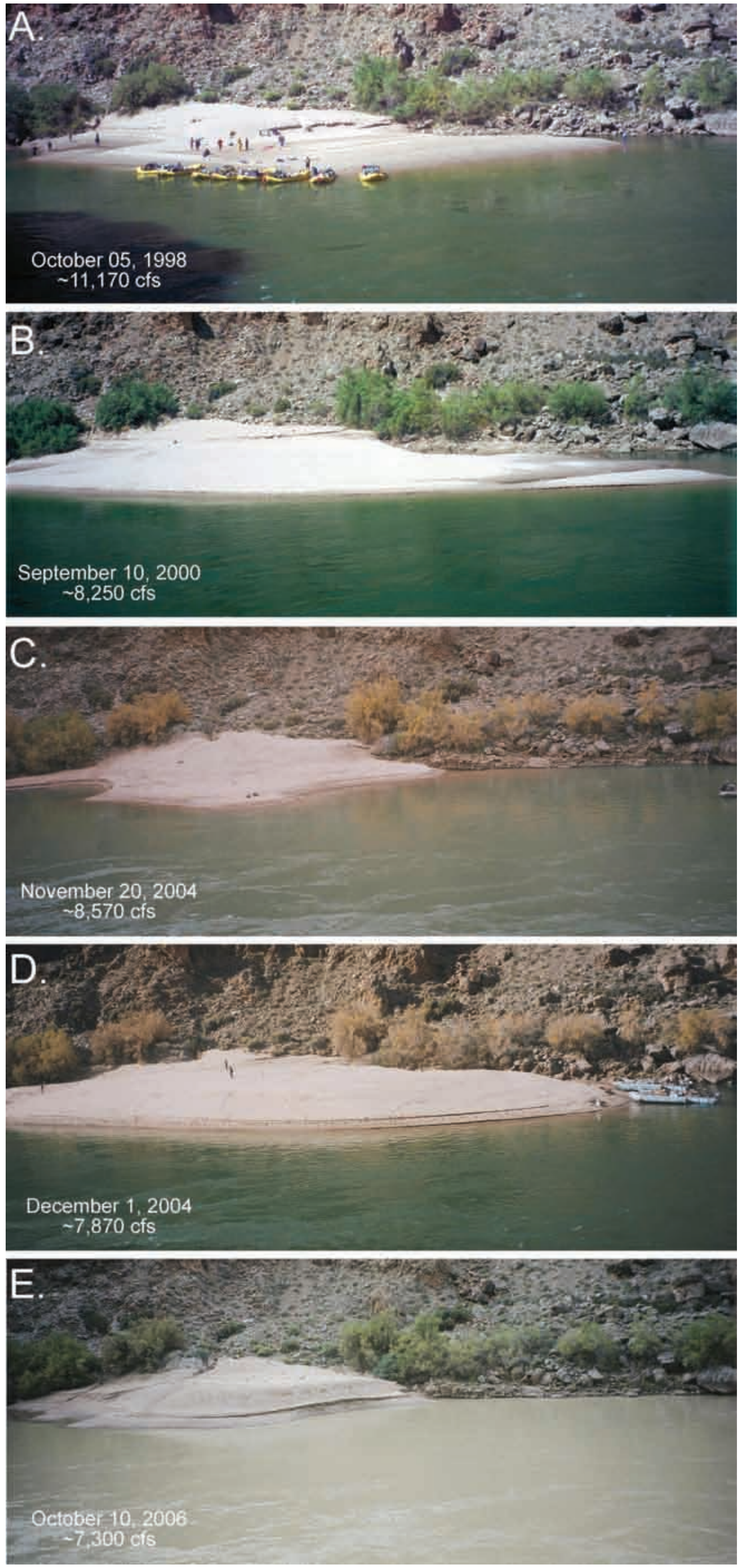

Figure 5. Time series of repeat photographs of the sandbar and campsite located at RM 30.7 illustrating campsite area changes at a nonvegetated sandbar. Flow in main channel is from right to left. The photograph in A shows the bar in 1998 after 2 years of erosion following the 1996 high-flow release 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

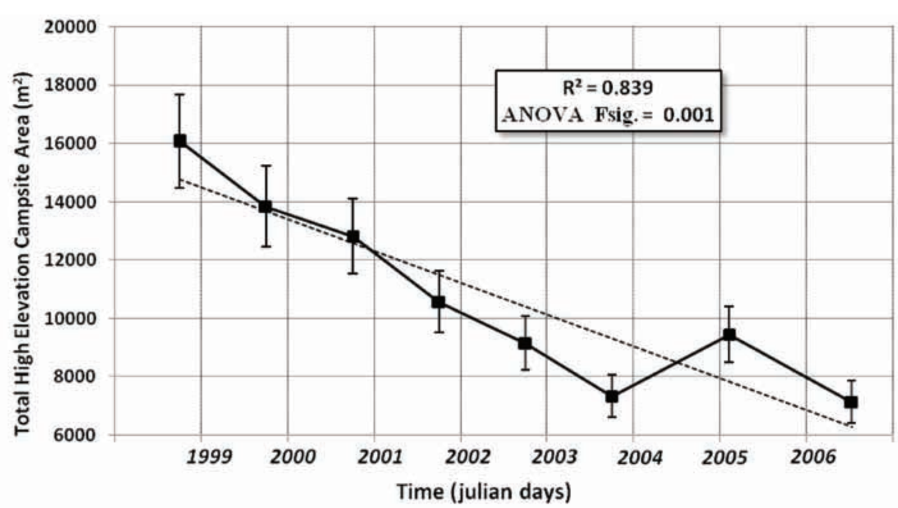

Figure 6. Total high-elevation campsite area for each survey between 1998 and 2006 (with 10 percent uncertainty). The dashed line shows the linear regression fit. Regression coefficient of determination and significance of one-way analysis of variance (ANOVA) are also shown.

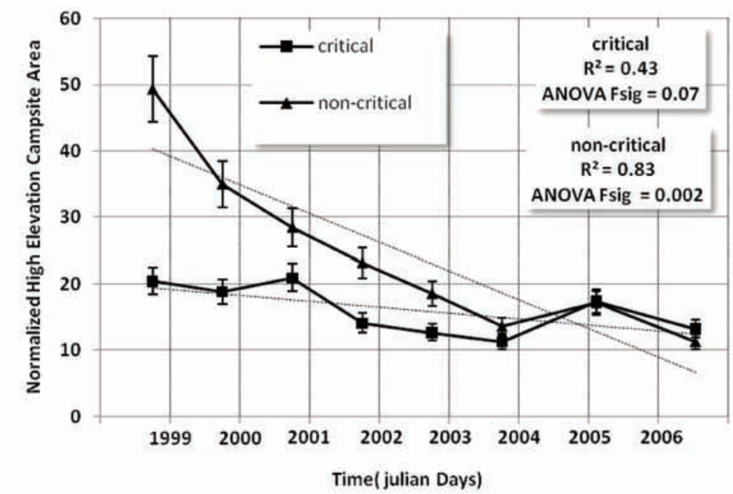

Figure 7. High-elevation campsite area in critical and noncritical reaches between 1998 and 2006 (with 10 percent uncertainty). The dashed lines show the linear regression fit. Regression coefficient of determination and significance of one-way analysis of variance (ANOVA) are also shown.

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.

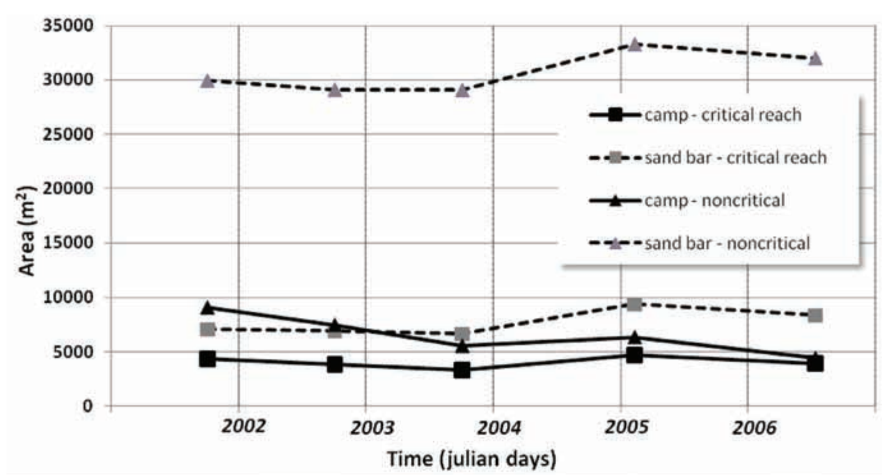

Figure 8. High-elevation campsite area and sandbar area in critical and noncritical reaches between 1998 and 2006 (with 10 percent uncertainty).

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.

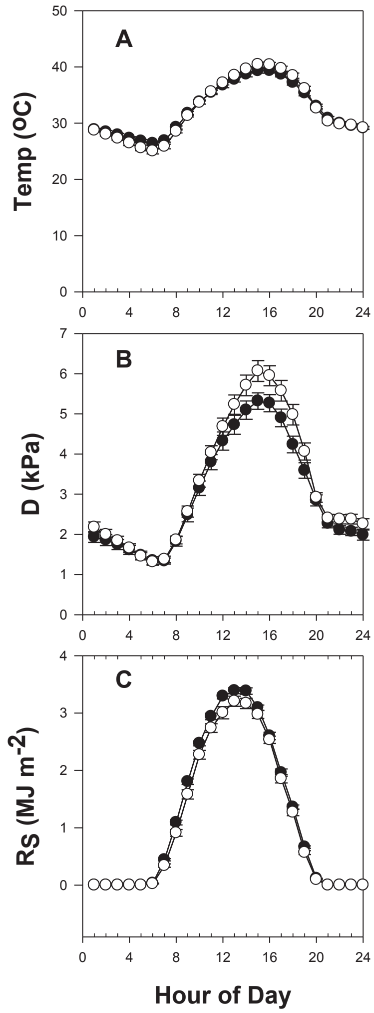


Figure 1. Diurnal patterns of (A) air temperature, (B) vapor pressure deficit, and (C) solar radiation at the Parker AZMET station near Cibola National Wildlife Refuge on the lower Colorado River during sap-flow measurements of transpiration in 2007 (closed circles) and 2008 (open circles).

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

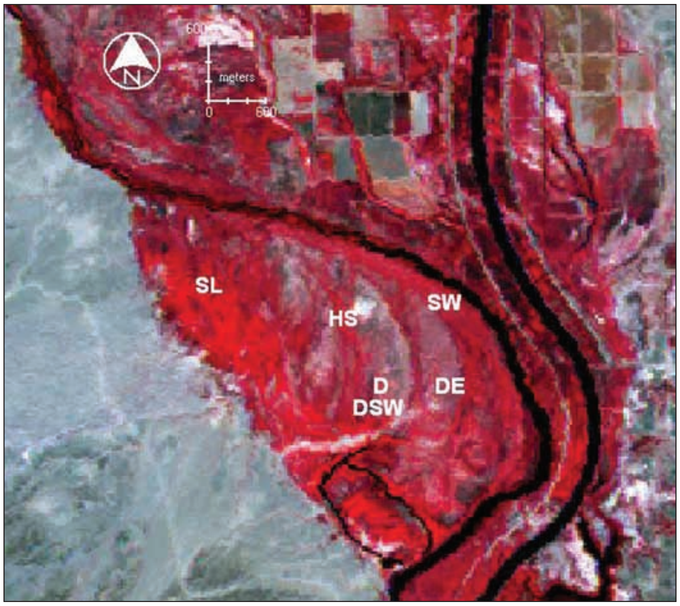


Figure 2. Approximate location of sap-flow study sites at Cibola National Wildlife Refuge on the lower Colorado River. Sites were named Slitherin (SL), Swamp (SW), Diablo Tower (D), Diablo South West (DSW), Diablo East (DE), and Hot Springs (HS). Sites are displayed on a Landsat ETM+ image. Data for 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:

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

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

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

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

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

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

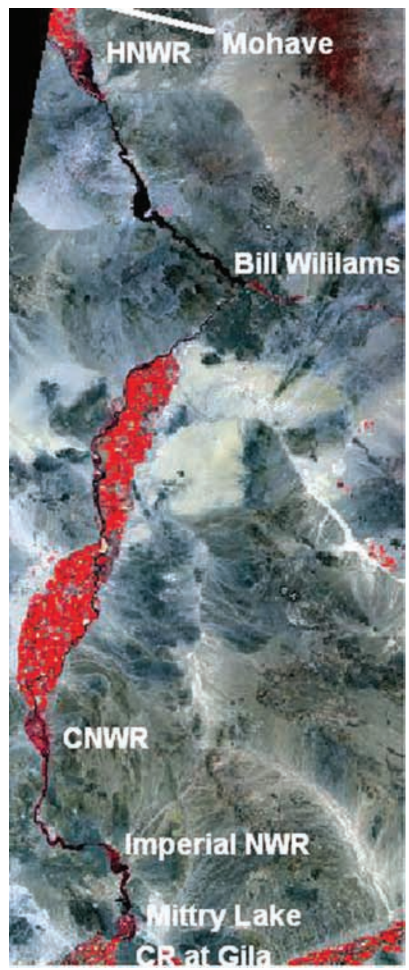


Figure 3. Location of wide-area sampling sites for estimating ET from MODIS EVI* pixels on the lower Colorado River. The Mohave site is not visible on this composite Landsat ETM+ scene, but it is just north of the irrigation district at the upper edge of the image.

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

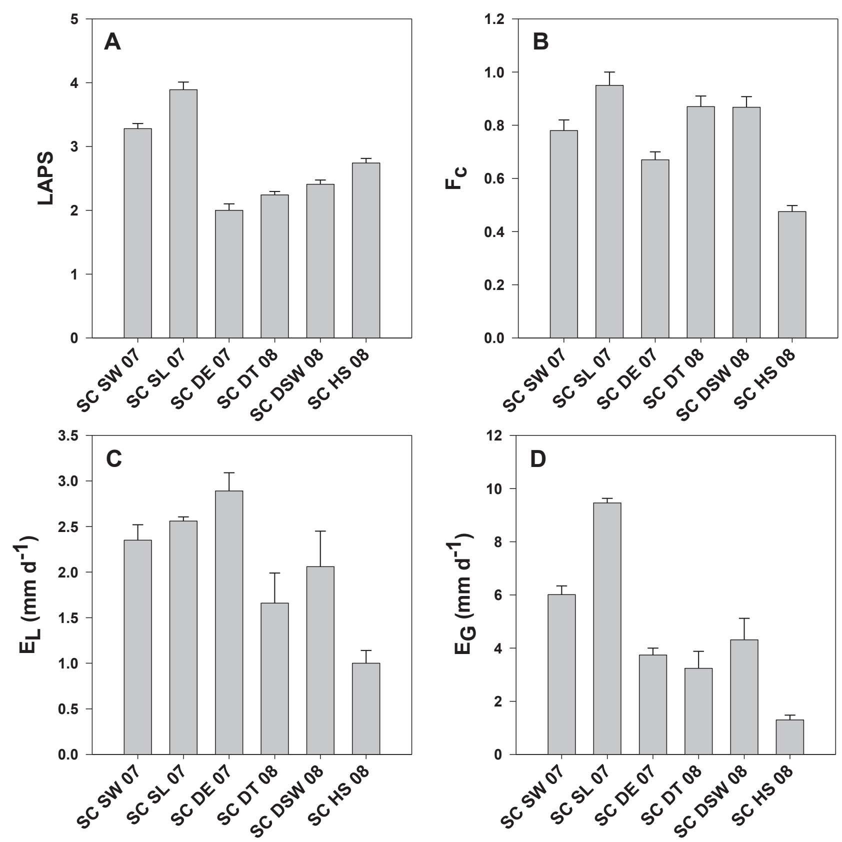


Figure 4. Values of saltcedar (SC) plant-specific (A) leaf area index (LAPS); (B) fractional cover (f_c); and (C) transpiration on a leaf-level (E_L) and (D) ground-level (E_G) basis. The x-axis shows the site (SL = Slitherin; SW = Swamp; DT = Diablo Tower; DSW = Diablo Southwest; DE = Diablo East; HS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

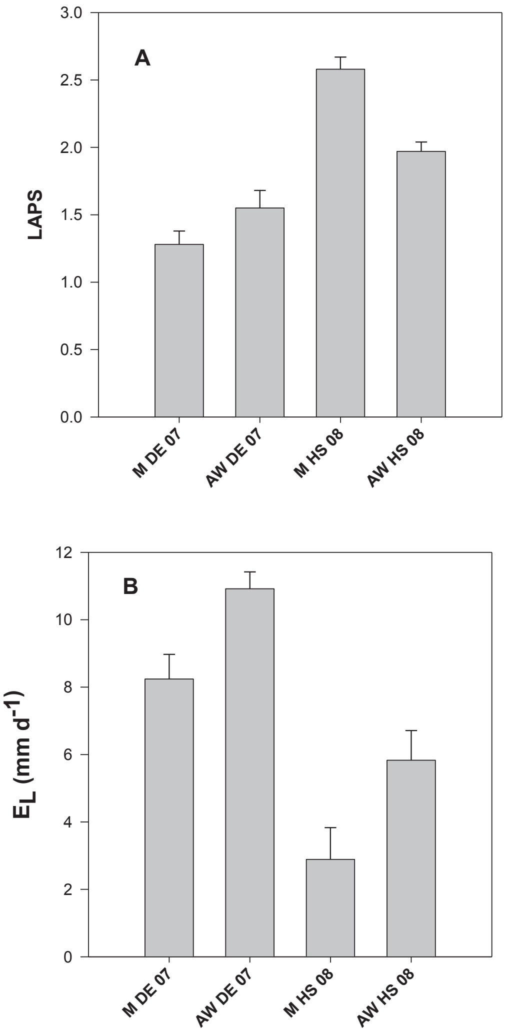


Figure 5. Plant-specific (A) leaf area index (LAPS) and (B) leaf-level transpiration (E_L) of mesquite (M) and arrowweed (AW) at different sites (HS = Hot Springs, DE = Diablo East) at Cibola National Wildlife Refuge in 2007 and 2008. Error bars are standard errors of the means.

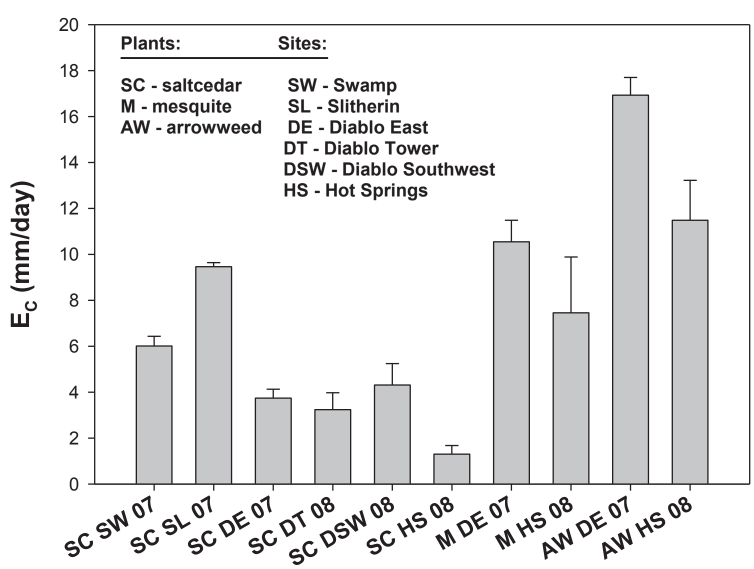


Figure 6. Canopy-level transpiration rates (E_C) of saltcedar (SC), mesquite (M), and arrowweed (AW) at different sites (SL = Slitherin; SW = Swamp; DT = Diablo Tower; DSW = Diablo Southwest; DE = Diablo East; HS = Hot Springs) at Cibola National Wildlife Refuge in 2007 and 2008.

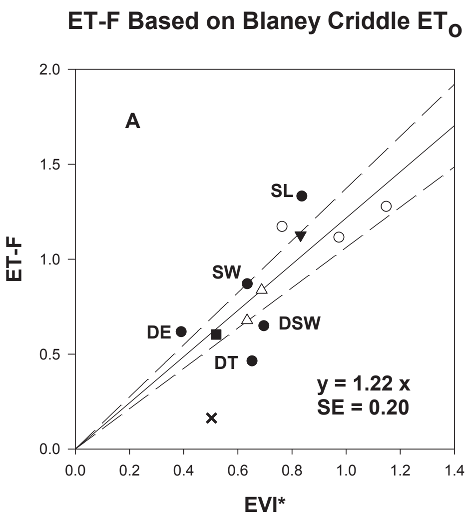
ranged from 5.5 to 16.0 mm d^{-1}. All the species showed high variability among sites, but saltcedar clearly did not have higher E_C than possible replacement species at CNWR. Over wider areas, saltcedar could have higher E_G than mesquite or arrowweed owing to differences in plant spacing. We were 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)



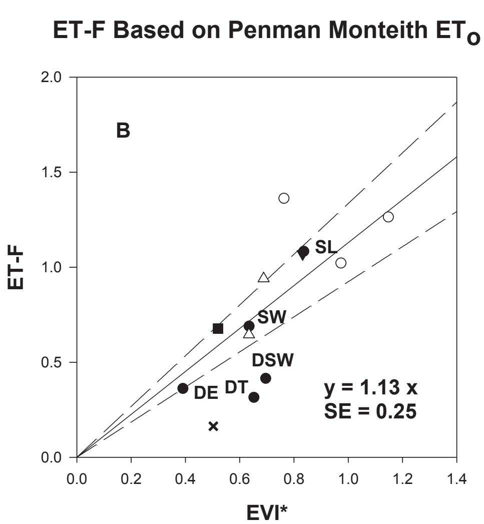


Figure 7. Ratio of actual ET to reference crop ET (ET-F) for plants on the lower Colorado River. Plants and locations are: saltcedar at Slitherin (SL), Swamp (SW), Diablo East (DE), Diablo Southwest (DSW), and Diablo Tower (DT) at Cibola National Wildlife Refuge (closed circles); saltcedar at Hot Springs 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

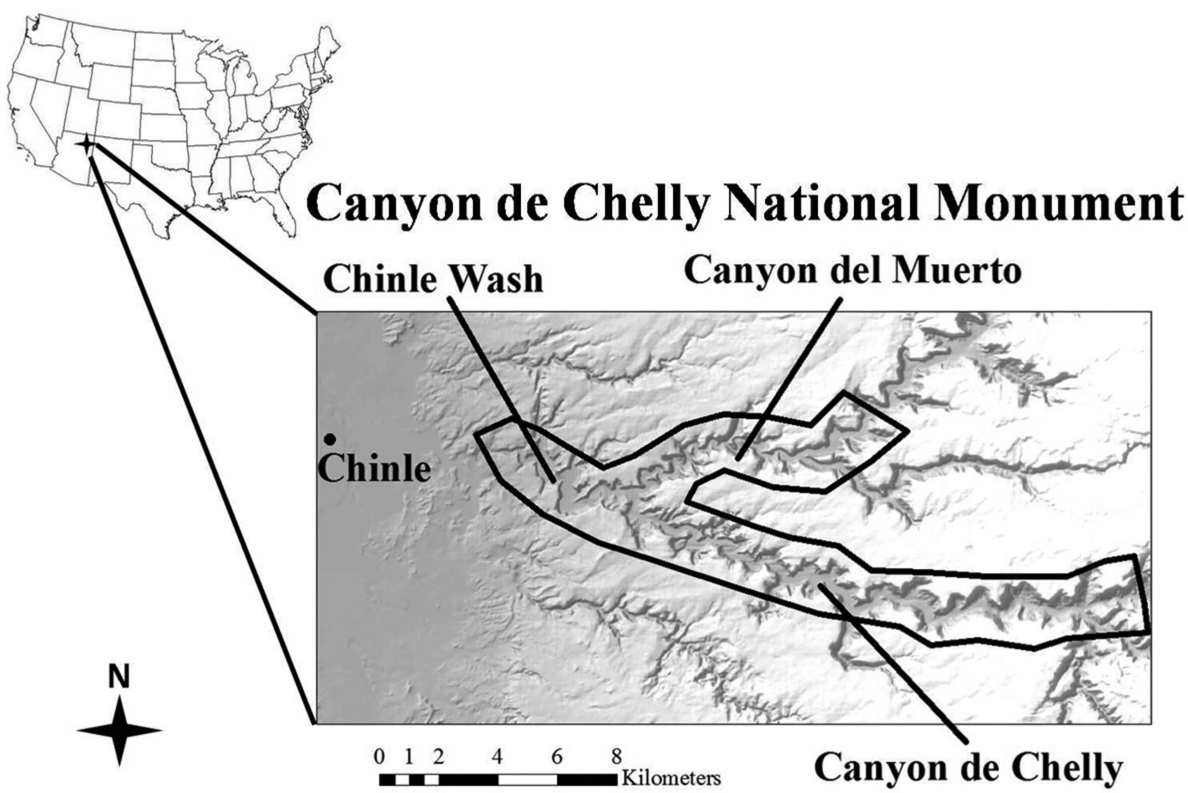

Figure 1. Canyon de Chelly National Monument. Our study area within Canyon de Chelly was limited to riparian areas within the canyon and is outlined in bold on the map.

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.

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

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

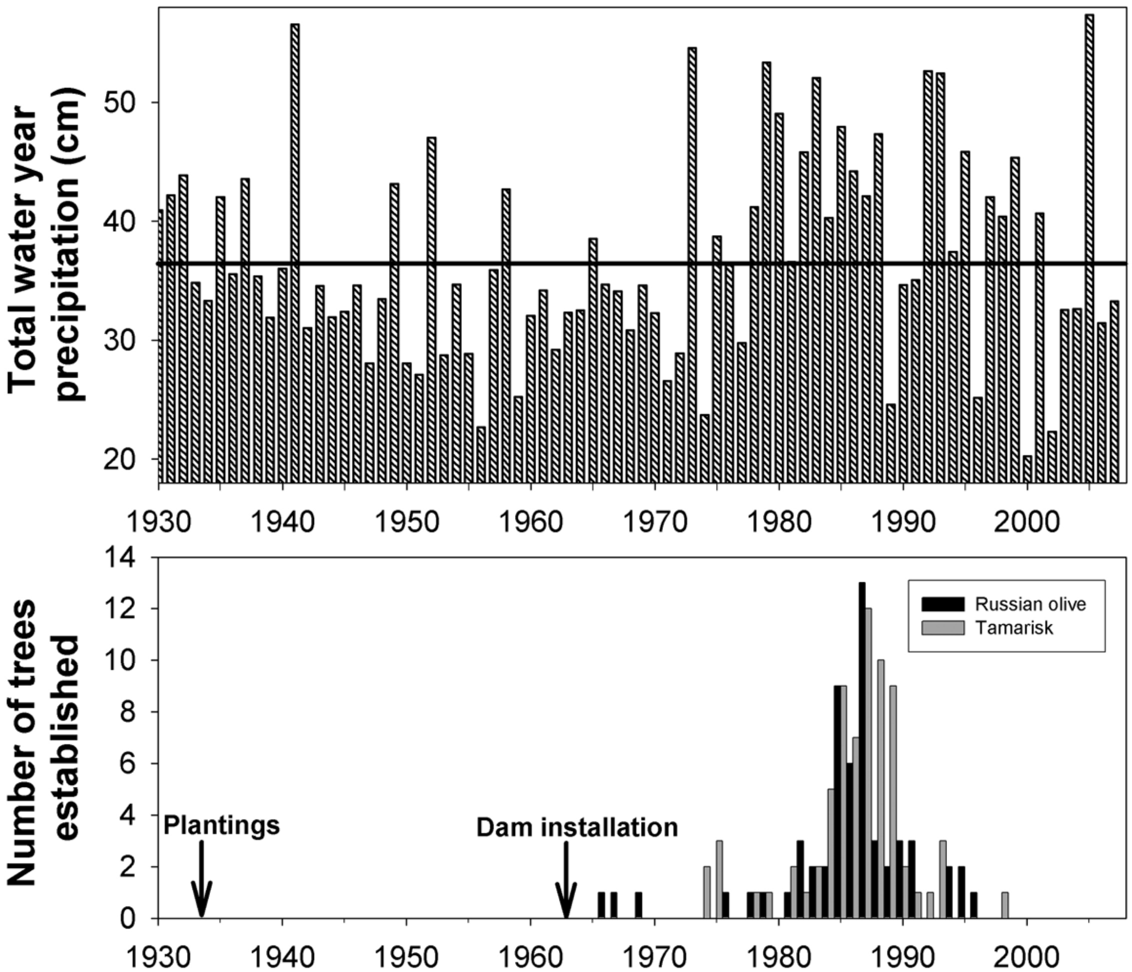

Figure 2. Total water year (October–September) precipitation (cm) for years 1930–2007 (top panel). The bold horizontal line indicates the 100-year average for total water year precipitation (37 cm/year). The bottom panel indicates the number of trees established in our study sites in each year.

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

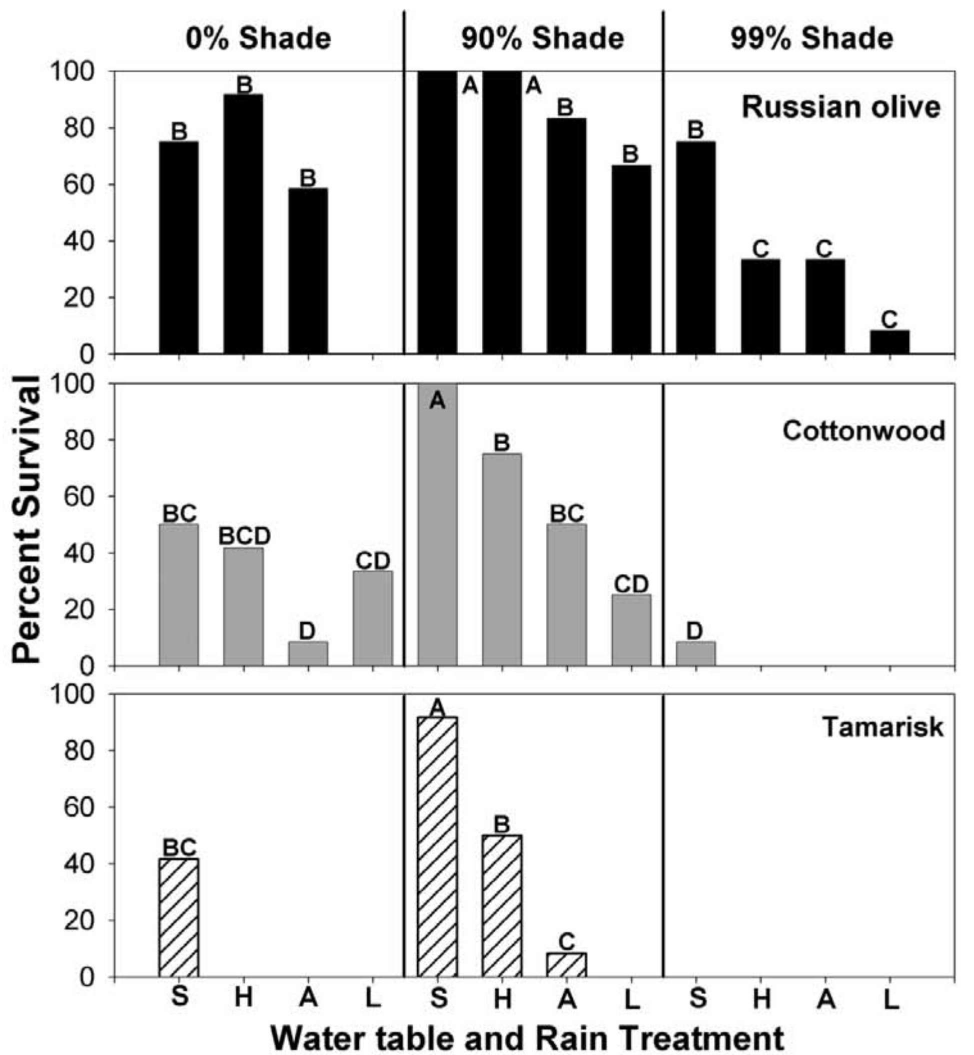

Figure 3. Percent survival of Russian olive (top), cottonwood (middle), and tamarisk (bottom) for each treatment. Shading treatment is indicated by the large boxes: full sun (0% shade), 90% shade, and 99% shade treatment. Watering treatments are indicated along the x-axis: S is shallow water table, H 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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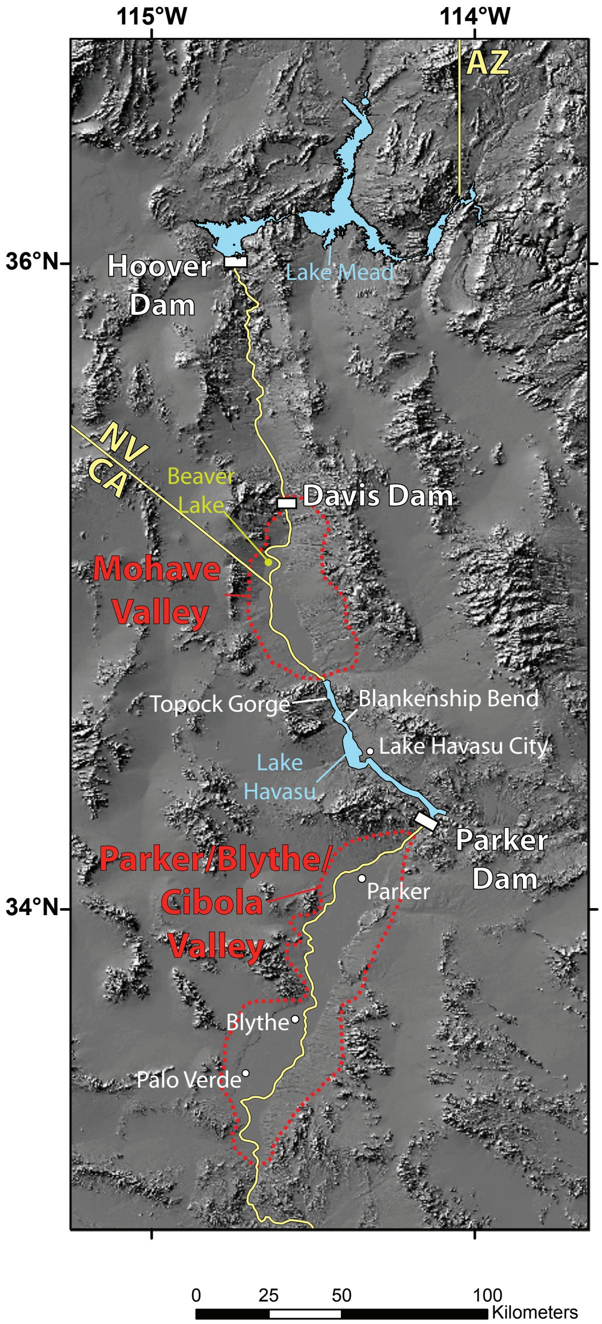

Figure 1. Hillshade image of the lower Colorado River valley based on a digital elevation model.

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).

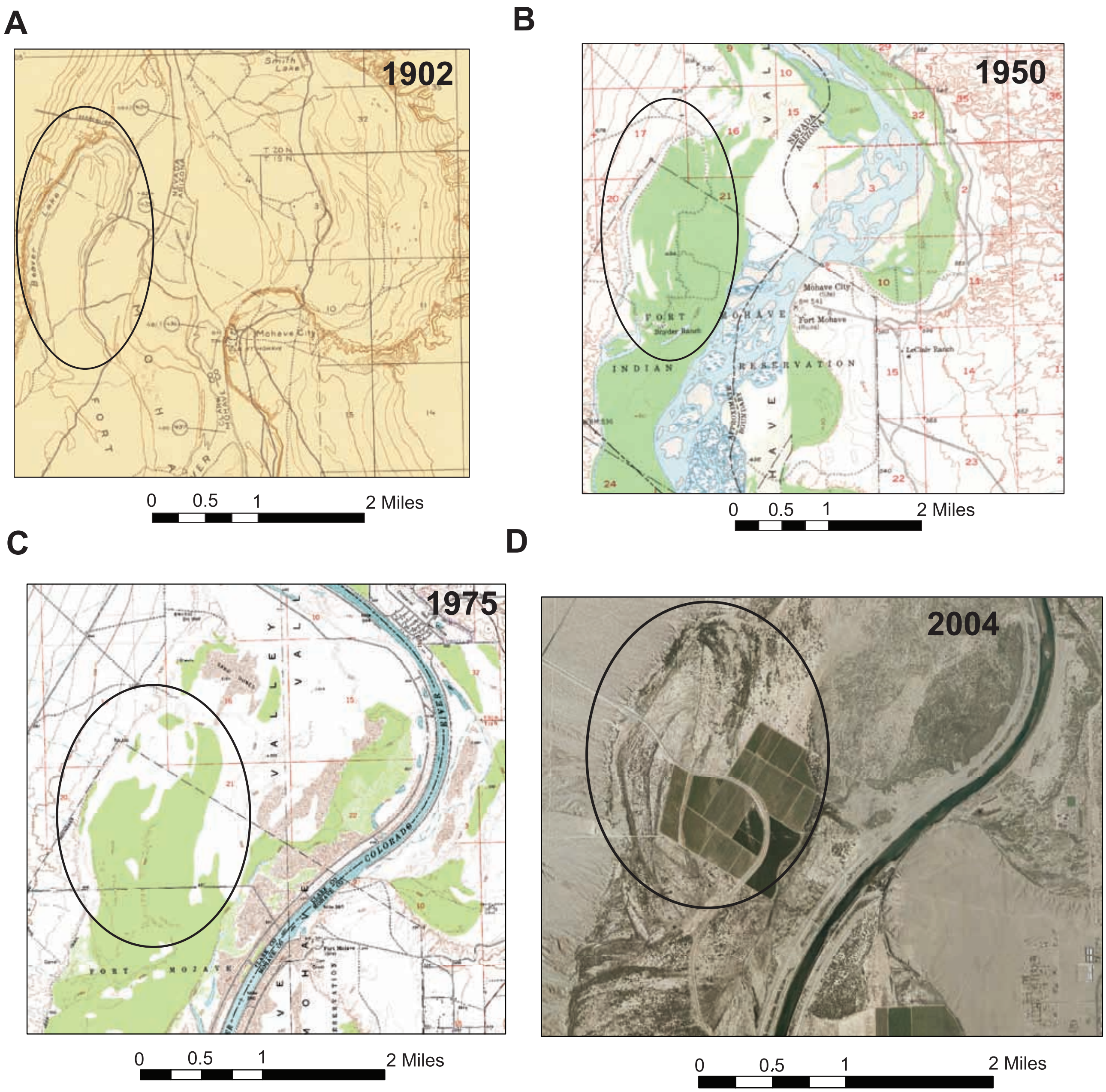

Figure 2. Beaver Lake, in Mohave Valley, as shown in three different topographic maps from (A) 1902, (B) 1950, and (C) 1975, and (D) in a recent (2004) digital orthophotograph.

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

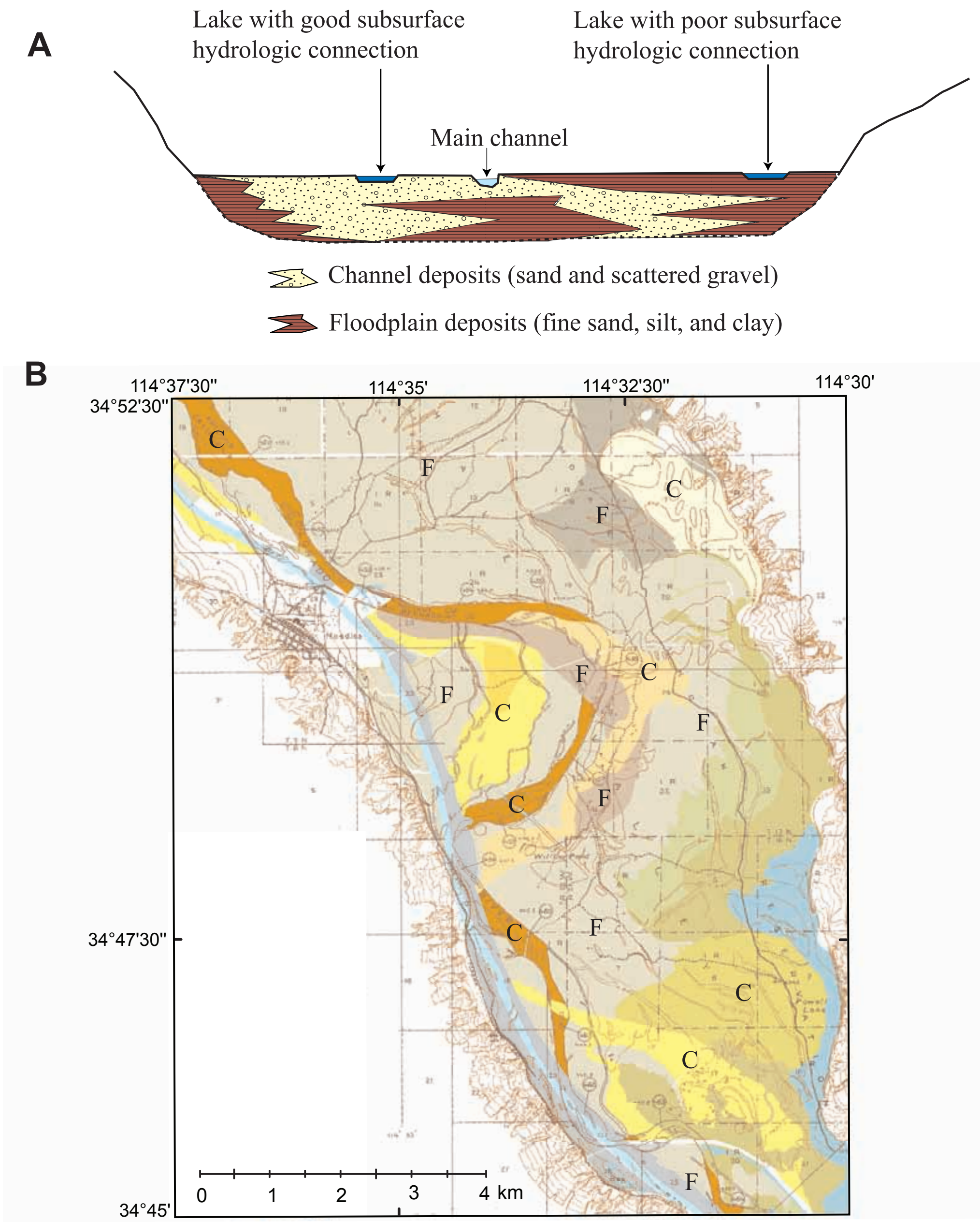

Figure 3. (A) Schematic cross section of coarse and fine sediments in the lower Colorado River valley, showing a location for a lake having adequate subsurface hydrologic connection with main channel by way of a continuous sand body, and a location with a poor subsurface hydrologic connection. (B) Mapped 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.

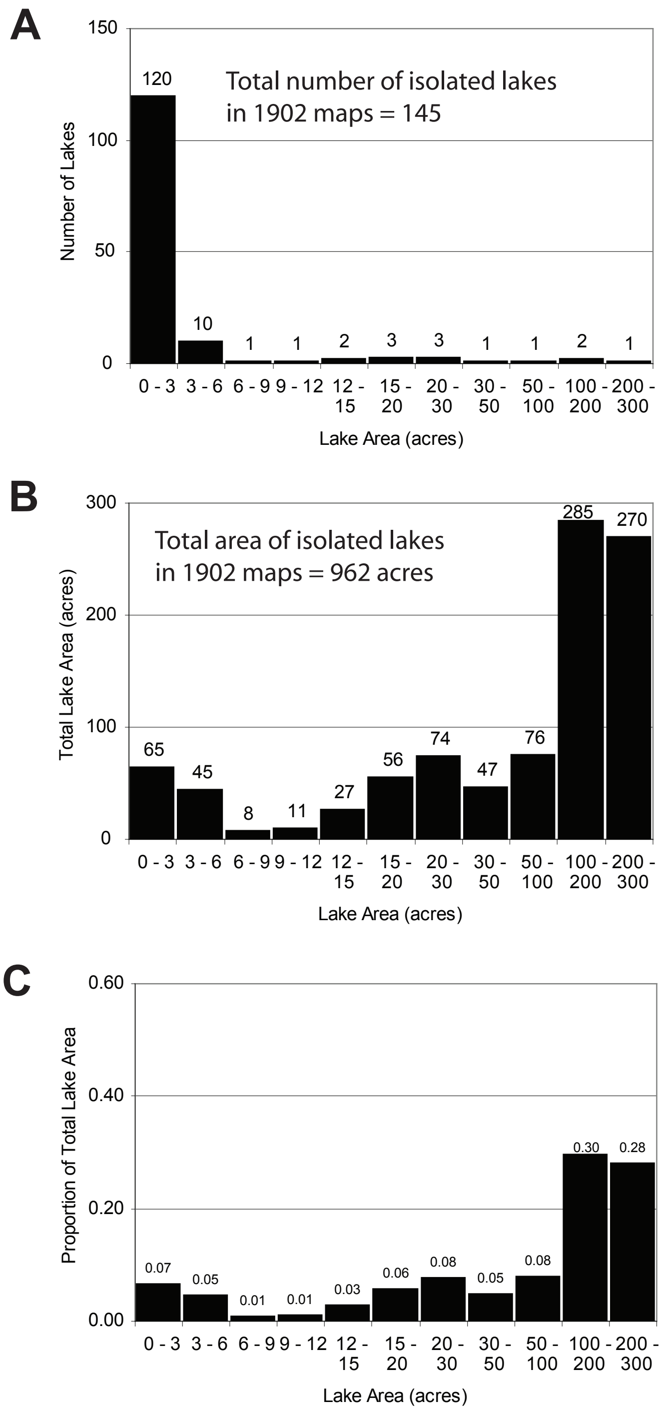

Figure 4. Statistics for isolated lake area in the pre-engineered Colorado River (Mohave, Parker, Blythe, and Cibola Valleys) from digitizing lakes on 1902 plane-table survey maps of the lower Colorado River. (A) Number of lakes in the natural system. (B) Total lake area within lakes of different size.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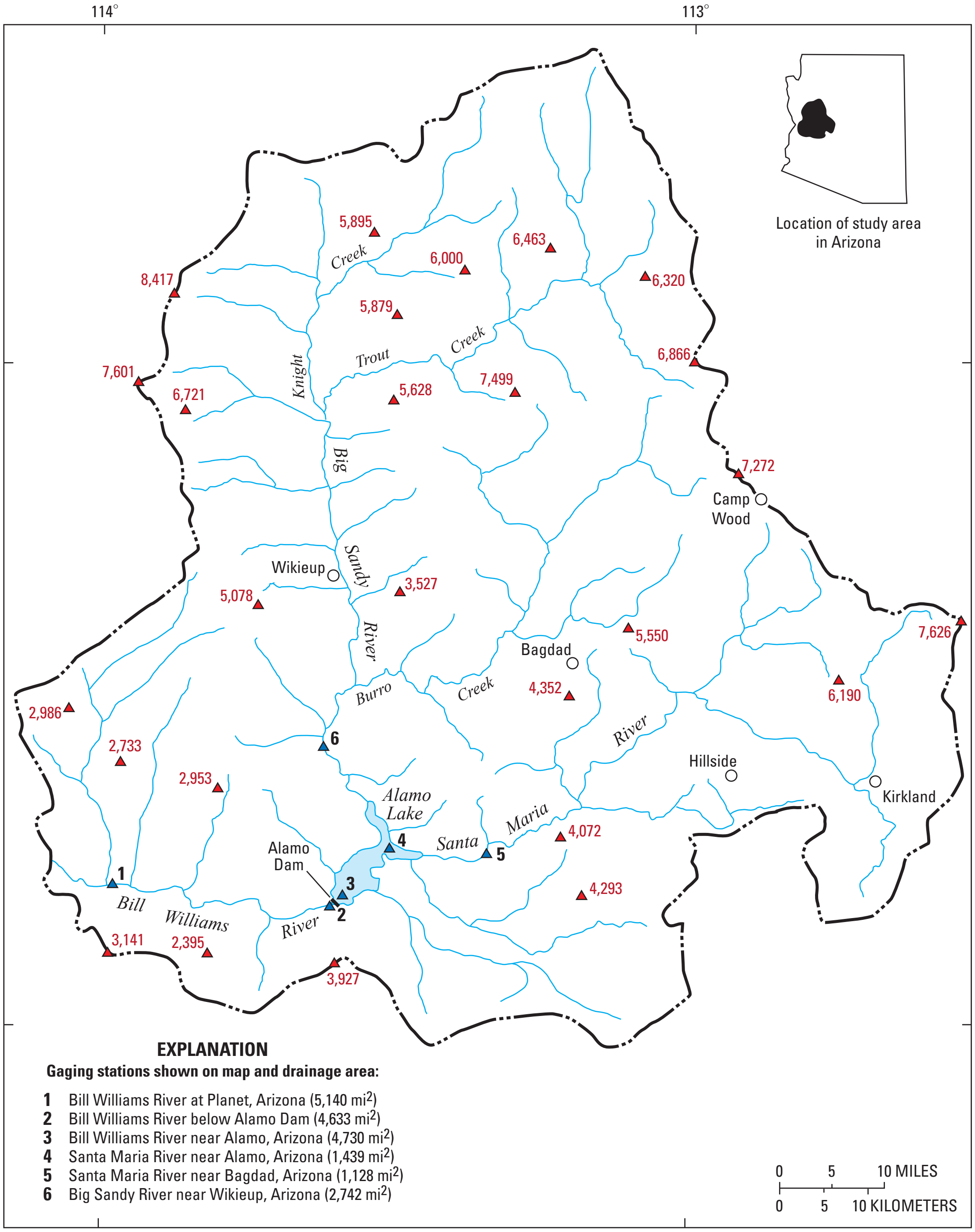

Figure 1. The Bill Williams River Basin (from Shafroth and Beauchamp, 2006).

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.

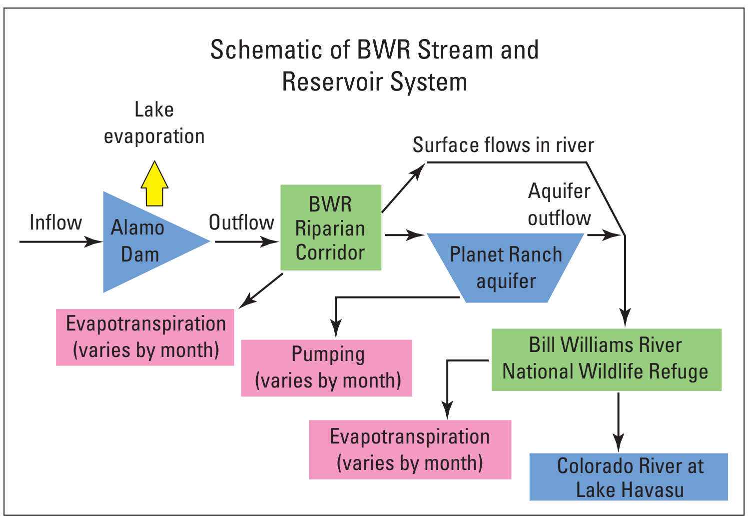

Figure 3. Schematic of Bill Williams River (BWR) stream and reservoir system used in hydrologic modeling.

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

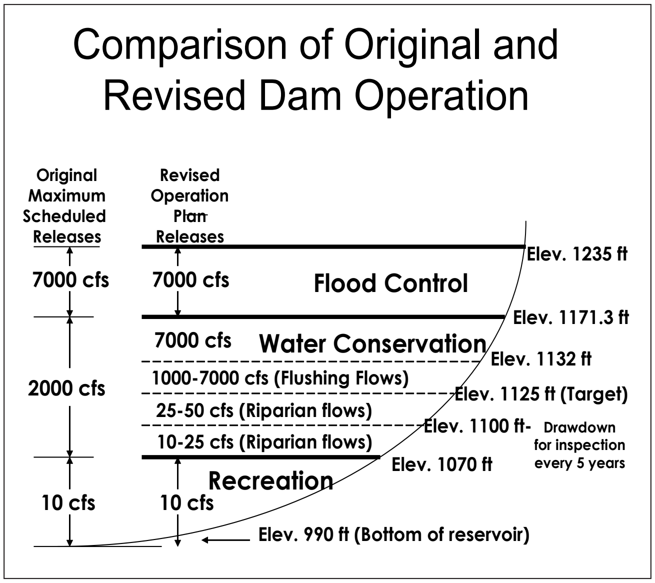

Figure 4. Original and revised dam operation schedules.

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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Posters

Aeolian Reworking of Sandbars from the March 2008 Glen Canyon Dam High-Flow Experiment in Grand Canyon

By Amy E. Draut,[1] Joseph E. Hazel Jr.,[2] Helen C. Fairley,[3] and Christopher R. Brown[2]

Abstract

The March 2008 high-flow experiment (HFE) replenished many sandbars along the Colorado River corridor in Grand Canyon downstream from Glen Canyon Dam. Some of those sandbars are source areas from which windblown sand moves inland to feed aeolian (wind-formed) sand dunes. Aeolian movement of sand following HFEs is important because some sand-dune fields in Grand Canyon contain archaeological sites that depend on a supply of windblown sand to remain covered and preserved. At two of nine sites where weather and aeolian sand transport are monitored, HFE sand deposits formed 1-meter-high dunes that moved inland during summer 2008, indicating successful transfer of sand to areas inland of the HFE high-water mark. At the other seven study sites, sand movement in nearby inland dunes was no greater than before the HFE. In order for HFE sand to move inland from sandbars toward aeolian dunes and archaeological sites, (1) sandbars must form upwind from archaeological sites (which requires sufficient sand supply in the Colorado River downstream from Glen Canyon Dam to sustain fluvial sandbar rebuilding through HFE releases); (2) local wind conditions must be strong enough and have the correct direction to move sand inland before subsequent river flows (after normal Glen Canyon Dam operations resume) erode the HFE sandbars; (3) sand transport must be unobstructed by vegetation or topographic barriers; and (4) sandbars must be dry enough for sand to be mobilized by wind.

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Introduction

The March 2008 high-flow experiment (HFE) of 41,000 cubic feet per second (ft^3/s) released from Glen Canyon Dam was intended to rebuild sandbars in the Colorado River corridor through Grand Canyon. This was the third such experimental flow; the earlier two occurred in March 1996 (45,000 ft^3/s; Webb and others, 1999) and November 2004 (41,000 ft^3/s; Topping and others, 2006). Some of the sandbars rebuilt by the HFEs are source areas from which windblown sand moves inland to replenish aeolian (wind-formed) sand dunes. Aeolian movement of sand following HFEs is important because some sand-dune fields in Grand Canyon contain archaeological sites that depend on a supply of windblown sand to remain covered and preserved (Neal and others, 2000; Draut and others, 2008). The U.S. Geological Survey (USGS) monitored aeolian transport of sand at selected study sites before and after the 2004 and 2008 HFEs. This paper discusses the degree to which sandbar enlargement by the 2008 HFE promoted windblown movement of sand inland toward dune fields and archaeological sites and compares the effects of the 2004 and 2008 HFEs on aeolian sand transport.

The 2008 HFE followed above-average input of sand and finer sediment to the Colorado River by the Paria River, 15 miles downstream from Glen Canyon Dam. Unlike in 2004, dam releases following the March 2008 HFE did not include experimental higher daily flow fluctuations like those that rapidly eroded sandbars after the 2004 HFE. Newly rebuilt sandbars, therefore, had not eroded much by the start of the 2008 spring windy season—aeolian sand transport tends to be greatest in Grand Canyon between April and early June—giving us the first opportunity to measure post-HFE aeolian sand transport with large sandbars still present.

Two Types of Aeolian Sedimentary Deposits in Grand Canyon

Previous research by Draut and Rubin (2008) defined two types of aeolian sedimentary deposits in the Colorado River corridor—modern fluvial (river) sourced (MFS) and relict fluvial sourced (RFS) deposits. The two types are distinguishable by their position relative to modern fluvial sandbars (those that formed at river flows of 45,000 ft^3/s or less) that could have provided windblown sand (fig. 1; Draut and Rubin, 2008). MFS dune fields are situated directly downwind from active (post-dam) fluvial sandbars and formed as the wind moved sand inland from sandbars, creating dune fields (fig. 1A). RFS deposits, in contrast, formed as wind reworked sediment from older (pre-dam), higher-elevation flood deposits, forming aeolian sand dunes from sediment left by floods that were larger than any post-dam floods (fig. 1B). RFS dunes may receive some sand from modern sandbars if the wind direction is appropriate, but their major source of sand is older deposits left by floods greater than 45,000 ft^3/s.

HFE releases of approximately 45,000 ft^3/s that rebuild modern sandbars can, therefore, replenish the sand sources that supply sand to inland MFS dune fields. After the 2004 HFE, at one study site where the new sandbar was not rapidly eroded by high fluctuating flows, aeolian sand-transport rates were significantly higher in the year after the HFE than in the year before (Draut and Rubin, 2008). However, in order to supply substantial amounts of new sand to RFS dune fields, much larger, sand-enriched high flows would have to occur.

The position and extent of MFS and RFS aeolian dunes are related to the magnitude of high flows that recur with sufficient frequency to provide a source of sand. Because all post-dam high flows since 1983 have been approximately 45,000 ft^3/s, the present location of MFS dunes is determined by sandbars deposited by those events. Changes in the high-flow regime could result in a change in the location and extent of MFS dunes. For example, an increase in high-flow magnitude may result in upslope expansion of the area of MFS aeolian dunes. Conversely, a decrease in peak-flow magnitude could result in downslope retreat of MFS dunes and a decrease in the area covered by active aeolian sand.

Aeolian Sand Monitoring Before and After the 2008 HFE

Since early 2007, the USGS has monitored weather conditions and aeolian sand-transport rates at nine aeolian dune fields in the Colorado River corridor where windblown

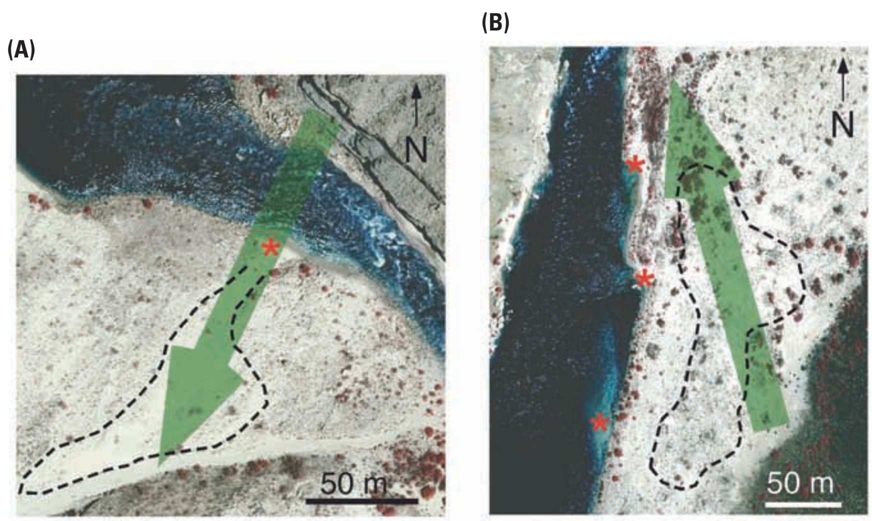


Figure 1. (A) Example of a modern fluvial sourced (MFS) aeolian dune field in Grand Canyon. The dune field (within dashed boundary) is directly downwind from a sandbar formed by flows at or below 45,000 cubic feet per second (asterisk). Here, the dominant wind direction is from the northeast (green arrow), so wind moves sand inland to form the dune field. High flows that rebuild sandbars, such as the March 2008 HFE, could supply new sand that then reaches MFS dune fields by wind transport. (B) Example of a relict fluvial sourced (RFS) aeolian dune field in Grand Canyon. The dune field (within dashed boundary) is not downwind from places where any modern sandbars form (asterisks). Instead, these aeolian dunes formed because the wind reworked sand from older, pre-dam flood deposits on terraces inland of the river (Hereford and others, 1996). The dominant wind direction in this area is from the southwest (green arrow), so sand is unlikely to be blown inland to the dunes from the modern sandbar sites (asterisks), even if those sandbars are enlarged by HFEs.

sand movement is important to the stability and preservation of archaeological sites. To evaluate whether the wind moved sand inland from sandbars that were enlarged by the 2008 HFE, we can compare measured rates of windblown sand transport in those dune fields during the year before and the year after the HFE. Similar records from some of the same sites are available from late 2003 to early 2006, capturing the year before and the year after the November 2004 HFE (Draut and Rubin, 2008). This allows us to compare some effects of the two high flows. In 2008, the size and shape of sandbars at five of the nine study sites were also monitored using topographic surveys (for example, Hazel and others, 2008) and repeat oblique photography before and after the HFE.

Methods

General locations of study sites are shown in figure 2 (exact locations cannot be disclosed, owing to their association with archaeological sites; we report only the site number, not its latitude, longitude, or river mile). At each site, one or more arrays of wedge-shaped, metal passive-sampling sand traps (Fryrear, 1986) catch samples of windblown sand that moves through the dune field. Researchers return to the sites periodically and collect the sand samples. The sample mass that accumulates in the traps over a known interval of time is used to estimate rates of sand flux moving through the dune field. Weather stations at or near each array of sand traps record wind speed and direction every 4 minutes, from which the net direction of probable sand transport can be calculated using vector sums of wind data from times when the wind was strong enough to move sand. The weather stations also record rainfall, temperature, humidity, and barometric pressure, so that we can determine if weather conditions were conducive to windblown movement of sand (wet sand will not blow around in the wind).

Results and Discussion

Of the nine sites where the USGS monitored aeolian sand transport before and after the 2008 HFE, two sites, AZ C:13:0321 and AZ C:13:0365, showed unequivocal evidence that sand deposited on sandbars by the HFE subsequently moved inland by wind action.

At AZ C:13:0321, topographic surveys before and after the 2008 HFE showed that the sandbar area increased by 129 percent and volume increased by 90 percent, owing to new sand deposition by the HFE. During the summer of 2008, sand formed a new aeolian dune 1–2 meters (m) high (fig. 3). The shape and orientation of the dune face implied that it was migrating (and moving sand) inland, toward a well-established dune field consisting of larger, vegetated dunes >10 m tall that are inland above the post-dam high-water elevation. As of October 2008, the new dune was taller (by 1.5 m) than the surface of the sandbar deposited by the HFE, and its crest was approximately 1 m higher than the maximum elevation reached by the HFE water. Because this site was monitored

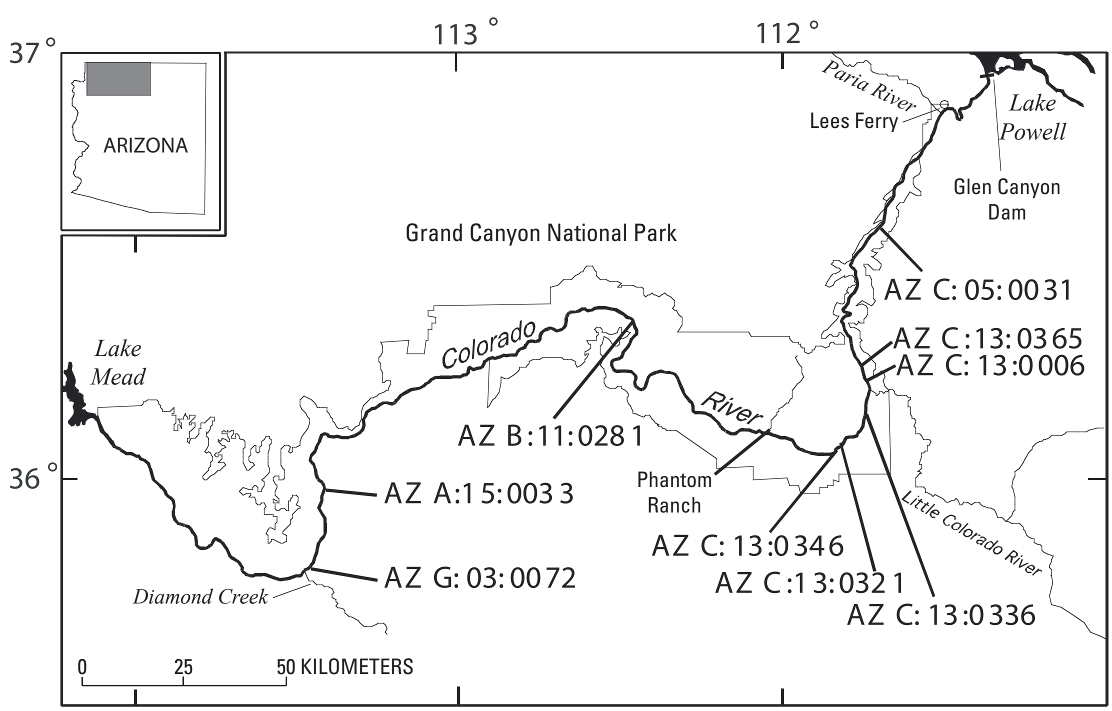
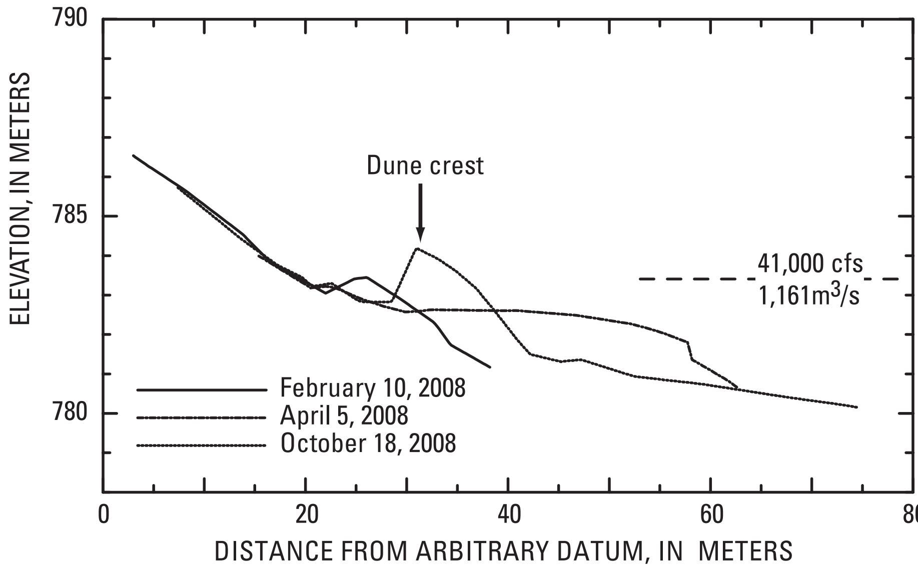


Figure 2. Sites where aeolian sand transport is monitored in the Colorado River corridor, Grand Canyon, Arizona. Site numbers refer to archaeological sites near weather stations and sand traps that measure weather conditions and rates of windblown sand flux in dune fields.

A



B

Figure 3. (A) Surveyed cross-section profiles across the sandbar at site AZ C:13:0321 made in February 2008 (1 month before the high-flow experiment (HFE)), April 2008 (1 month after the HFE), and October 2008 (7 months after the HFE). Growth of the sandbar from HFE sand deposition is apparent, as is the formation of an aeolian dune crest between the April and October surveys. The elevation of the dune crest in October was approximately 1.5 m higher than the surface of the sandbar left by the HFE, and nearly a meter higher than the maximum elevation reached by the HFE waters (horizontal dashed line). The orientation of the dune crest and slipface show dune migration (and sand transport) inland. (B) The aeolian dune crest that formed on the HFE sandbar at site AZ C:13:0321 taken on July 29, 2008.

beginning in February 2008, it is not possible to compare sand-transport rates with the year before the HFE, but daily sand flux measured at the site during summer 2008 was similar to that of the most active dune fields in the canyon, at approximately 3 grams per centimeter width.

At site AZ C:13:0365, topographic surveys showed that the HFE caused a loss of sandbar area (by 17 percent) but increased sandbar volume (by 14 percent). During the summer of 2008, one end of the HFE sandbar formed an aeolian dune, similar to the one observed at site AZ C:13:0321. As of July 2008, the dune crest was approximately 1 m higher than the surrounding sandbar, and the dune shape and orientation indicated dune migration inland from the river toward a large MFS aeolian dune field where sand-transport rates are some of the highest known in Grand Canyon (Draut and Rubin, 2008). Wind conditions measured by two weather stations at AZ C:13:0365 were consistent with inland-directed sand transport, as the dominant wind direction blew from the sandbar site inland toward the large dune. In the spring windy season of 2008 (after the HFE), windblown sand transport was greater near river level at this site than at any time measured between mid-2004 and early 2006 (no data are available for this site between January 2006 and February 2008). Higher up in the dune field, sand-transport rates in spring 2008 were similar to those measured between 2004 and early 2006.

At the seven remaining study sites, there was no clear evidence for HFE-deposited sand moving inland by wind. At two of the sites, AZ C:13:0336 and AZ A:15:0033, this was the expected result because aeolian dunes there are RFS sedimentary deposits, the sand sources of which occur at too high an elevation to have been replenished by the March 2008 HFE. At the remaining five study sites, lack of renewed aeolian sand transport to the dunes is attributable to inappropriate wind conditions or to blocking of MFS sand by vegetation or topography. Three of these five study sites (AZ C:05:0031, AZ B:11:0281, and AZ G:03:0072) contain apparently MFS aeolian dunes, which lie downwind from fluvial sandbars capable of being enlarged by HFEs, but had wind conditions after the 2008 HFE that were not effective at moving sand inland. At AZ C:05:0031, increased aeolian sand transport from the sandbar to the dune field was documented after the November 2004 HFE, but no similar response occurred after the 2008 HFE. The 2008 HFE caused some growth of the sandbar there (increasing area by 1 percent and volume by 8 percent). Although the wind commonly blows inland toward the dune field at AZ C:05:0031, between March and June 2008 the wind instead blew predominately upstream, parallel to the river. Wind conditions, therefore, were not conducive to moving sand inland from the new HFE deposit toward the dunes during the 2008 spring windy season. At AZ B:11:0281 and AZ G:03:0072, although the prevailing wind directions from March to June 2008 were oriented from the river margin inland toward dune fields, neither area experienced a significant increase in wind strength during that time of year, so spring sand transport was no higher in 2008 than in 2007. The degree of sandbar growth from the HFE is unknown at those two sites because they were not surveyed.

The final two MFS study sites showed no increase in aeolian sand transport after the 2008 HFE either because sandbars there did not enlarge much or because, although in the past fluvial sand was able to move inland toward these dunes, the dune field at each site is now separated from the associated river-level sand deposits by vegetation and (or) topographic barriers. At AZ C:13:0006, the HFE removed 13 percent of the sandbar area but increased its volume by 15 percent. The typical wind direction at this site is consistent with movement of sand inland toward an MFS aeolian dune field; however, sand-transport rates in the dune field were no higher in 2008 than in 2007. Lack of increased sand flux in the AZ C:13:0006 dune field may be because not much new sand was available on the source sandbar (having lost area) and (or) because sand must cross a side canyon, about 5 m wide, in order to move from the sandbar site into the aeolian dune field. Although this topographic influence (the side canyon) is not new, and windblown sand must have crossed it in the past to form the dune field, it is likely that a much larger sandbar would be required upwind in order for sand transport across the side canyon to increase measurably.

At site AZ C:13:0346, although wind conditions were appropriate to have moved sand inland and upslope toward large dunes, neither of two sand-trap arrays measured any increase in aeolian sand transport in 2008 relative to 2007. Any new HFE sand deposited on sandbars upwind from this dune field is separated from the dunes by a thick band of vegetation parallel to the river, which would have been less of an obstacle during pre-dam time, as this vegetation has grown substantially since the 1960s (apparent in historical aerial photographs). It is likely that although the aeolian dunes at site AZ C:13:0346 can be considered MFS deposits (downwind from sandbars at the 45,000 ft^3/s level), new sand would not readily move toward the dunes unless the vegetation were removed.

Implications for Management

Investigations of the 2004 and 2008 HFEs have shown that under sufficiently sand-enriched condition, HFEs can create new sandbars and enlarge existing ones, at least on time scales of months. Unlike the 2004 HFE sandbars, which quickly eroded because of high fluctuating flows, the 2008 HFE sandbars were present during spring months, the season when windblown sand transport generally is greatest in Grand Canyon.

At two of nine study sites (AZ C:13:0321 and AZ C:13:0365), spring and summer winds reworked the 2008 HFE sand deposits to form new aeolian dunes. The shape of the dunes in both cases indicated sand movement inland toward larger, well established dune fields. At site AZ C:13:0365, measured spring windy-season sand transport near river level was substantially greater after the 2008 HFE than after the 2004 HFE (when sandbars eroded before the 2005 spring windy season).

At the other seven study sites, HFE deposits did not form sizeable aeolian dunes, and sand-transport rates after the 2008 HFE were similar to or lower than in previous years. At several sites, inappropriate wind conditions in spring 2008 likely limited the inland movement of HFE sand; at other sites, lack of increased sand flux is attributable to blocking by vegetation or local topography. Vegetation removal could facilitate the movement of sand inland from sandbars by wind, although this has not yet been attempted in Grand Canyon.

In general, sandbars created or enlarged by HFEs can potentially contribute new sand to MFS dune fields (those downwind from sandbars formed or replenished by the HFE), but these sandbars are not expected to contribute much additional sand to RFS dune fields (which formed as wind reworked sediment left by larger, pre-dam floods). The number and proportion of Grand Canyon archaeological sites that are downwind from MFS sandbars and, thus, could benefit from HFEs are not known precisely, because wind conditions and sediment substrate vary substantially from site to site, and wind conditions and sedimentary history have been studied in detail at only about a dozen sites (this study and Draut and Rubin, 2008). The precise relation between sandbar size, resulting quantity of sand transferred to a MFS dune field, and how long new sand remains in the dune field is uncertain. Recent light detection and ranging (lidar) surveys in the river corridor are providing valuable information about landscape evolution around archaeological sites that will help to address these outstanding questions (Collins and others, 2008).

The greatest potential for inland sand movement after HFEs is in the spring, when weather commonly includes stronger winds with less rain likely than at other times of year; dam operations that maintain large sandbars in spring months, therefore, provide the best chance for sand to move inland by wind toward MFS dunes and any associated archaeological sites.

The effectiveness of HFEs to supply new sand to MFS aeolian dunes depends on the following:

1. The formation or enlargement of sandbars upwind from the dunes. This requires a sufficient sand supply in the Colorado River downstream from Glen Canyon Dam to sustain fluvial sandbar rebuilding through HFE releases (Wright and others, 2008).

2. The dominant local wind direction and intensity after the HFE near each sandbar.

3. Windblown sand moving from a sandbar to a dune field without being blocked by vegetation or topography.

4. Dryness of sandbars after the HFE. Even high winds cannot transport sand if rain or daily flow fluctuations keep the sandbar surfaces wet.

Acknowledgments

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Applying an Ecosystem Framework to Evaluate Archaeological Site Condition Along the Colorado River in Grand Canyon National Park, Arizona

By Helen C. Fairley[1] and Hoda Sondossi[1, 2]

Abstract

The Colorado River corridor in Grand Canyon National Park encompasses numerous archaeological sites, many of which are actively eroding. This desert riparian ecosystem is currently experiencing significant ecological change, and many of these changes have been attributed to the emplacement and operation of Glen Canyon Dam. Because archaeological sites are physical remains of past human activities embedded within biophysical terrain, they are subject to the same agents of change that affect ecosystems on a landscape scale. To assess the effects of dam operations on downstream archaeological sites, the U.S. Geological Survey Grand Canyon Monitoring and Research Center is developing a monitoring program that "unpacks" the concept of archaeological site condition according to the key ecological factors that shape and maintain ecosystems in general, as defined by the Jenny-Chapin conceptual model of ecosystem sustainability. This process-based approach to monitoring archaeological site condition has several potential advantages over more traditional approaches to monitoring cultural resources that typically rely on the professional judgments of archaeologists to assign qualitative ratings such as good, fair, or poor without distinguishing the diverse factors that contribute to these judgments. Specific advantages of an ecosystem-based approach for monitoring dam-related impacts at archaeological sites include the following: (1) the approach recognizes that dam effects are ecosystemic, not point specific; (2) the approach explicitly recognizes that impacts to archaeological sites are fundamentally an extension of the effects influencing ecosystem change as a whole, and therefore, dam-related impacts may include effects resulting from the loss or diminishment of certain fundamental ecological processes (e.g., reduction in the intensity or frequency of flood-induced disturbance processes) as well as direct impacts from current dam-controlled water releases; (3) the approach acknowledges that archaeological sites are constantly undergoing change, even under the most stable ecological conditions, and therefore, impacts from dam operations must be evaluated in a dynamic ecosystem context; and (4) the approach explicitly recognizes that archaeological site condition, like the ecosystems of which they are a part, reflects the long-term, cumulative effects of interacting ecosystem processes over time, and therefore, relatively recent dam-related effects must be understood and evaluated in this larger temporal context. By designing the monitoring approach for cultural resources within an ecosystem-based conceptual framework, scientists and managers can acquire the types of data needed to distinguish and evaluate the role of dam operations relative to the multiple additional ecological factors and processes that contribute to physical stability and erosion of archaeological sites in the Colorado River corridor.

Introduction

Archaeological sites are physical remains of past human activities that have left a tangible imprint on the landscape. As such, they are embedded within biophysical terrain and are subject to the same agents of change that affect ecosystems on a landscape scale. The Colorado River corridor in Grand Canyon National Park, a landscape and ecosystem encompassing numerous archaeological sites (Fairley and others, 1994; Fairley, 2003), is currently experiencing significant ecological change (Carothers and Brown, 1991; Webb, 1996; Webb and others, 2002), much of which is attributed to the emplacement and operation of Glen Canyon Dam (U.S. Department of the Interior, 1995).

The effects of Glen Canyon Dam operations on downstream natural and cultural resources in Glen Canyon National Recreation Area and Grand Canyon National Park have been a focus of scientific inquiry by the U.S. Geological Survey's (USGS) Grand Canyon Monitoring and Research Center (GCMRC) since inception of the Glen Canyon Dam Adaptive Management Program (GCDAMP) in 1997. Systematic

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monitoring of resource condition is necessary not only to determine whether management policies and actions are having intended effects on a given resource, but also to determine what management actions are most likely to be effective under varying environmental conditions. Furthermore, Federal laws, such as the National Environmental Policy Act (Public Law 91–190), the National Historic Preservation Act (Public Law 89–665), and the Grand Canyon Protection Act (title XVIII, §§1801–1809, Public Law 102–575), mandate that Federal agencies evaluate the effects of their management decisions and actions on the affected environment and on cultural resources specifically. Because archaeological sites situated on or embedded within eroding river terraces and sandy deposits lining the Colorado River corridor are some of the resources potentially affected by operations of Glen Canyon Dam, the GCMRC has been charged with developing scientifically defensible monitoring protocols to track the status and trends of archaeological resource condition in the Colorado River ecosystem (CRE). The GCDAMP strategic plan (Glen Canyon Dam Adaptive Management Program, unpub. document, 2003) advocates using an ecosystem-based approach to evaluate dam effects. To fulfill the intent of existing laws, and in keeping with the GCDAMP strategic guidelines, USGS scientists are collaborating with Utah State University geomorphologists, National Park Service (NPS) archaeologists, and other technical experts in a multiyear research initiative to develop an ecosystem-based approach to monitoring archaeological site condition in the CRE.

We are meeting this challenge by applying a model of ecosystem sustainability first proposed by Jenny (1941, 1980) and subsequently refined by Chapin and others (1996) to structure the monitoring approach. This conceptual model is currently being applied in other monitoring contexts outside the CRE (e.g., Miller 2005; Chapin and others, 2006), although it has not previously been applied to monitoring archaeological sites specifically. While archaeological sites differ from landscape-scale ecosystems in several important respects, especially in terms of their resilience (Holling, 1973; Pimm, 1984; Berkes and Folke, 1998), their condition is affected and largely determined by the same dynamic processes that shape the ecosystems in which they occur; therefore, an ecosystem framework is appropriate for assessing how dam operations, in conjunction with other interacting ecosystem processes, influence and impact the physical integrity of archaeological sites in the CRE.

Background and Rationale

NPS archaeologists have monitored archaeological sites in the CRE since the late 1970s (Fairley, 2003). These past monitoring efforts and related studies have documented active erosion occurring at many sites (e.g., Leap and others, 2000; Thompson and Potochnik, 2000; Fairley, 2005; Pederson and others, 2006). In a recent evaluation of past archaeological site monitoring efforts in Grand Canyon, a panel of archaeological experts observed that archaeological site condition is a multi-dimensional construct that needs to be “unpacked” into its primary constituents for the purposes of assessing how operations of Glen Canyon Dam may be affecting the condition of archaeological resources and contributing to their erosion in the Colorado River corridor (Kintigh and others, unpub. report, 2007). Unpacking the concept of site condition not only requires articulating the various types of “impacts” that contribute to an assessment of archaeological site condition, but also it requires defining explicit management goals for the resource (e.g., preservation in place, public interpretation, learning about the past), defining the variables that contribute to perceptions about archaeological site condition in a particular management context, and identifying the processes that are likely to change those conditions. In keeping with this recommendation, the GCMRC is developing a new approach for monitoring archaeological sites that explicitly acknowledges the multi-dimensional nature of site condition and the multiple ecosystem processes responsible for changing the condition of these resources over time. We are developing this program through defining and quantifying (directly measuring) the effects of various ecosystem agents and processes that are theorized to affect ecosystem sustainability (Chapin and others, 1996) and thereby have the potential to affect site condition. As outlined in the Jenny-Chapin model (Chapin and others, 1996), the four key processes critical to sustaining ecosystems are local weather regimes, sediment supply dynamics, functional biological systems, and disturbance regimes.

Because archaeological sites are continually being transformed by interacting ecosystem processes that promote weathering of minerals, redistribution of sediment, and organic decay, even under the most stable environmental conditions, archaeological sites generally tend to degrade (i.e., retain less physical integrity) with the passage of time. In other words, unlike most ecosystems that have the capacity to rebound from ecosystem changes as long as certain boundary thresholds are not exceeded (Holling and Meffe, 1996; Berkes and Folke, 1998), archaeological sites lack inherent resilience, and therefore, the processes and impacts that affect their physical integrity are cumulative over time. This poses a philosophical and managerial dilemma for cultural resource managers and archaeologists who are charged with assessing the condition of these nonrenewable resources and preserving them for the benefit of future generations. What does it mean for an archaeologist or land manager to determine that an archaeological site is in “good” or “poor” condition after a site has been subjected to 1,000+ years of episodic flooding, deposition, and erosion? What set of values or criteria are used to make these judgments? If a site has been buried for centuries and is now becoming exposed through erosion, what rate of erosion is acceptable and what rate of change constitutes an unacceptable impairment of resource values?

Some resource management agencies deal with this philosophical conundrum by substituting the concept of current *site stability* for *site condition*. For example, the NPS Archaeological Site Management System (National Park

Service, unpub. document, 2006) defines a site to be in good condition if it shows “no evidence of noticeable deterioration [and] the site is considered currently stable,” whereas a site is rated to be in fair condition if it shows “evidence of deterioration [and] without appropriate corrective treatment, the site will degrade to a poor condition.” Previous methods for determining whether archaeological sites are stable or actively deteriorating and how fast they may be changing and the reasons why typically have been based on qualitative judgments (general observations of change; e.g., Leap and others, 2000) rather than robust quantitative data (measurements of change) and, hence, are not replicable or independently verifiable, two fundamental premises of the scientific method. The current study proposes to use innovative monitoring tools and techniques to increase the quality and quantity of monitoring data and enhance overall understanding of effects from dam operations and other ecological factors on archaeological site condition. Specifically, through the use of various survey tools (e.g., Collins and others, 2008) and weather monitoring instruments (Draut and others, 2009) combined with site-specific geomorphic data (O’Brien and Pederson, unpub. report, 2009) and systemwide data on sediment supply (David J. Topping, U.S. Geological Survey, oral commun., 2008) and vegetation (e.g., Ralston and others, 2008) derived from other ongoing monitoring efforts in Grand Canyon, we are quantifying physical changes occurring at archaeological sites in relation to key measurements of critical ecosystem processes.

The Jenny-Chapin Model as a Conceptual Framework to Guide Monitoring

The Jenny-Chapin model (Chapin and others, 1996) conceives of ecosystems as being constrained by *state factors* and sustained by a suite of interacting ecosystem processes known as *interactive controls* (fig. 1). State factors are relatively static conditions that apply to a given geographic location, such as parent material (bedrock geology), topography, regional climate, and the various organisms that are physically capable of existing at that location. Time is also an important constraining factor. Within these basic limits, four key ecosystem processes interact with each other to create and maintain a given ecosystem. These *interactive controls* on the system are local weather regimes, sediment supply dynamics, functional groups of organisms, and disturbance processes. According to the Jenny-Chapin model, interactive controls maintain ecosystem sustainability through negative feedback loops that counter and, to some degree, offset the effects of individual interactive controls. A basic premise of the Jenny-Chapin model is that when one or more interactive controls change substantially, the ecosystem will become unstable; if the change persists, the ecosystem will become unsustainable and eventually will be transformed into a fundamentally different ecosystem.

In the CRE, interactive ecosystem controls have changed significantly as a direct result of dam operations, altering the feedback loops that formerly sustained the pre-dam ecosystem. In particular, the soil resource supply (sediment supply, grain size, soil chemistry) and disturbance regime (flood frequency, daily and seasonal range of flows, annual volume of flows) have been altered by the presence and operation of Glen Canyon Dam (Turner and Karpiscak, 1980; Schmidt and Graf, 1990; Rubin and others, 2002; Topping, Rubin, and Vierra, 2000; Topping, Rubin, and others, 2000; Topping and others, 2003). These systemic changes appear to be affecting the stability and physical integrity of many archaeological sites in the CRE (Hereford and others, 1993; U.S. Department of the Interior, 1995). For example, although surface erosion was a significant and ongoing process during pre-dam times, the effects of surface erosion were mitigated to some degree by annual spring floods that reworked lower elevation sandbars and periodically deposited sediment at higher elevations. Wind also reworked and re-deposited flood sand across the surfaces of higher terraces and inland dune fields (Hereford and others, 1993; Draut and others, 2005). Thus, in pre-dam times, the downcutting and surface soil loss inherent to erosional processes in a semiarid environment were offset to some

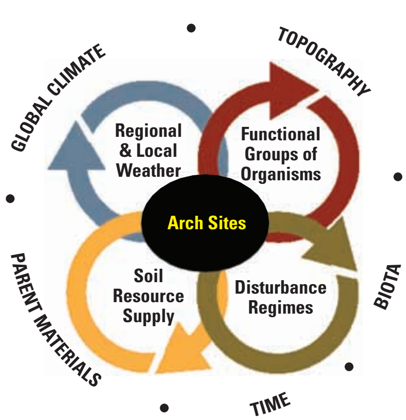

Figure 1. The Jenny-Chapin model conceives of ecosystems as being constrained by state factors (external circle) and sustained by a suite of interacting ecosystem processes known as interactive controls (internal circles). The Grand Canyon Monitoring and Research Center cultural monitoring research and development project initially focused on documenting the various state factors that define the archaeological sites’ physical context; the program is now focused on developing appropriate tools for monitoring the interactive controls that affect the site’s ability to resist change, and hence determine their long-term stability in the face of ecological changes occurring throughout the Colorado River ecosystem (after Chapin and others, 1996).

degree by other interactive controls that promoted backfilling and infilling of gullies and replenished surface sediment on a landscape scale (e.g., McKee 1938; Hereford and others, 1996; Thompson and Potochnik, 2000; Draut and Rubin, 2008), thereby contributing to the sites' capacity to resist erosive agents of change.

Dam operations have also impacted terrestrial vegetation and habitats with potential consequences for archaeological site stability (fig. 2). The near absence of high flows capable of pruning and scouring shoreline vegetation has altered the riparian habitat along the river, particularly in the new high-water zone (Carothers and Brown, 1991). Consequently, shoreline vegetation has increased and shifted in composition since emplacement of the dam (Turner and Karpiscak, 1980; Stevens and others, 1995). Changes in near-shore vegetation have not only affected types and abundance of plants and animals inhabiting the CRE, but also they have affected rates of sediment transport and retention in the ecosystem (Draut and others, this volume). The extent to which dam operations have impacted the old high-water zone remains unclear because of a lack of recent vegetation monitoring above the 60,000 cubic feet per second stage elevation, although past research in the CRE predicted significant changes to old high-water zone vegetation as a result of dam operations (Anderson and Ruffner, 1987; Ralston, 2005). The consequences of ecological changes occurring in the old high-water zone, where many archaeological sites are situated, in terms of current and future site condition, are currently unknown, but the ecosystem-based monitoring approach currently under development by the GCMRC is being designed to help alleviate this crucial data gap.

Changes also have occurred as a result of indirect effects of dam operations, such as increased human disturbance from large numbers of private and commercial recreational boaters, a phenomenon made possible in part by reliable, year-round, dam-controlled flows. Human disturbance from tourism is known to be an important factor affecting archeological site integrity world wide (United Nations Educational, Scientific, and Cultural Organization, 2007). In the CRE, visitor impacts, such as graffiti, artifact removal, and the creation of social trails, have been documented at many of the archaeological sites in the river corridor during previous monitoring by the National Park Service (U.S. Department of the Interior, 2005). How these visitor impacts affect ecological processes within the CRE is less well documented and understood, although land managers generally consider the effects to be adverse (U.S. Department of the Interior, 2005). One way in which visitors have impacted archaeological site stability is by damaging the biological soil crusts that currently stabilize many formerly active aeolian sand surfaces covering archaeological sites. When soil crusts are broken or compacted by human trampling, the shear strength of the soil is reduced (G. O'Brien and J. Pederson, written commun., 2008), and rapid erosion of the underlying sediment during subsequent high-intensity precipitation events may follow, which often leads to new gullies forming along the trails (fig. 3). This is one

Figure 2. Hopi elders examine culturally important riparian plants growing along the Colorado River in Grand Canyon. Vegetation encroachment because of the lack of periodic scouring floods has transformed near-shore habitats and affected the abundance and distribution of native organisms that once sustained the Native American human inhabitants of the Colorado River ecosystem. It has also created new habitats that support many nonnative species. The increase in vegetation has also stabilized many shoreline sandbars, reducing the availability of sand for transport by wind, thereby contributing to the deflation of formerly active dune fields and the consequent erosion of the many Native American ancestral sites. While scientific monitoring can document the ecological processes and the consequent effects to archaeological sites, determining whether these changes translate into "good," "fair," or "poor" resource condition can only be done by the cultures and people who value these "resources" and interpret their meaning for society. (photograph courtesy of Michael Yeatts and the Hopi Tribe).

Figure 3. Biological soil crusts now stabilize many formerly active aeolian sand surfaces covering archaeological sites. When soil crusts are broken or compacted by human trampling, rapid erosion of the underlying sediment may follow, leading often to gullies forming along trails %melqldo^me _v > jv Ao^rq) R+P+ Dblildf`^i Prosbv&.

reason why human disturbance at archaeological sites must be systematically monitored in conjunction with other ecosystem processes: dynamic interactions between ecosystem processes may be as important as individual ecological processes in destabilizing archaeological sites.

In addition to the resource impacts noted above, some ecological changes may be occurring in the Colorado River corridor that have little or nothing to do with dam operations, including effects from global climate change and indirect effects related to worldwide human population increases (e.g., effects to air quality from dust and pollution). Regardless of ultimate cause, all of these factors have direct and potentially profound implications for the future sustainability of the Colorado River ecosystem and the stability of archaeological sites contained within. Furthermore, these factors have important implications for the sustainability of other culturally valued resources in the CRE, such as the native plants and animals of cultural importance to Native Americans who previously inhabited the river corridor and for whom the landscape as a whole continues to have cultural significance (Fairley 2003; Dongoske and others, this volume). By monitoring effects of dam operations in an ecosystem context and specifically in relation to the dam-affected individual ecosystem controls operating in the system, it is possible to begin the process of assessing how dam operations affect cultural resources in a cumulative sense and on an individual site-by-site basis, as well as the overall landscape context in which they exist.

Applying the Conceptual Model to Monitoring of Archaeological Site Condition

The GCMRC currently is designing monitoring protocols to quantify the amount and rates of physical change occurring at archaeological sites in relation to the interactive controls currently operating in the Colorado River ecosystem; the protocols also are designed to track the interdependent effects of these interacting processes. As a first step in this research and development process, a suite of fundamental physical attributes linked to basic “state factors” of the Jenny-Chapin model were defined for each archaeological site, including bedrock geology, primary and subsidiary landforms, surface-cover characteristics, and a ranked assessment of current site stability (O’Brien and Pederson, unpub. report, 2009); important archaeological characteristics and inherent values of each site were also documented (L. Leap, unpub. data, 2007). This information provides a baseline context for evaluating potential changes that may occur in the future and provides an important tool for understanding the diverse geomorphological contexts of archaeological sites in the Colorado River corridor. Next, potential tools and techniques for measuring environmental parameters and detecting and quantifying the amount of surface change were field tested and evaluated in terms of cost-time efficiency, measurement accuracy, and potential resource impacts (Collins and others, 2008; Draut and others, 2009). Two different types of survey technology were deployed and tested simultaneously (but independently), along with multiparameter weather stations, in order to evaluate the potential of each monitoring tool and resulting dataset to inform other monitoring results (fig. 4). For example, terrestrial light detection and ranging (lidar) can precisely measure surface erosion, sediment deposition, and other surface changes occurring at individual archaeological sites (Collins and others, 2008; Collins and others, 2009), while weather stations situated in proximity to the sites provide high-resolution data on wind direction and intensity, rainfall, temperature, humidity, and barometric pressure (Draut and others, 2009). Sand traps near the weather stations collect windblown sediment to track sediment movement from near-shore sources to inland archaeological sites under varying weather and sediment-supply conditions (Draut and others, this volume). By replicating and analyzing lidar survey data in conjunction with local weather and sediment transport data collected during the same time intervals, effects of local weather events or changes in sediment supply (e.g., as a result of sandbar enhancement from experimental high flows or because of change in the density of near-shore vegetation) can be correlated with measured topographic change (Collins and others, 2009). In this manner, episodes of downcutting or infilling of gullies or significant accumulations of sediment at archaeological sites can be linked to specific environmental parameters and to significant changes in local conditions, including those tied to dam operations.

The development of final protocols for monitoring archaeological site condition is a work in progress. In the future, we anticipate that analysis of remotely sensed multispectral aerial imagery collected once every 4 years, in combination with periodic field surveys, will allow scientists to measure changes in vegetation at both site-specific and landscape scales. We are also exploring remote-sensing methods to measure trends in biological soil crust cover at archaeological sites, in order to evaluate how changes in surface cover characteristics bear upon archaeological site stability. Combining these data with high-resolution topographic change measurements (e.g., Collins and others, 2009) and sediment monitoring techniques (e.g., David J. Topping, oral commun., 2008; Hazel and others, 2008; Draut and others, 2009; Draut and others, this volume) will allow us to monitor effects of specific hydrological events, such as natural tributary floods and high-flow experiments, on archaeological sites throughout the system.

In addition to monitoring physical changes at archaeological sites in relation to local weather, sediment-supply dynamics, and other interactive controls, future monitoring data also can be analyzed in relation to the suite of “state factors” that define the sites’ geomorphic context (O’Brien and Pederson, unpub. report, 2009). This will provide a much more robust understanding of how relatively constant environmental factors, such as bedrock geology and topography, in

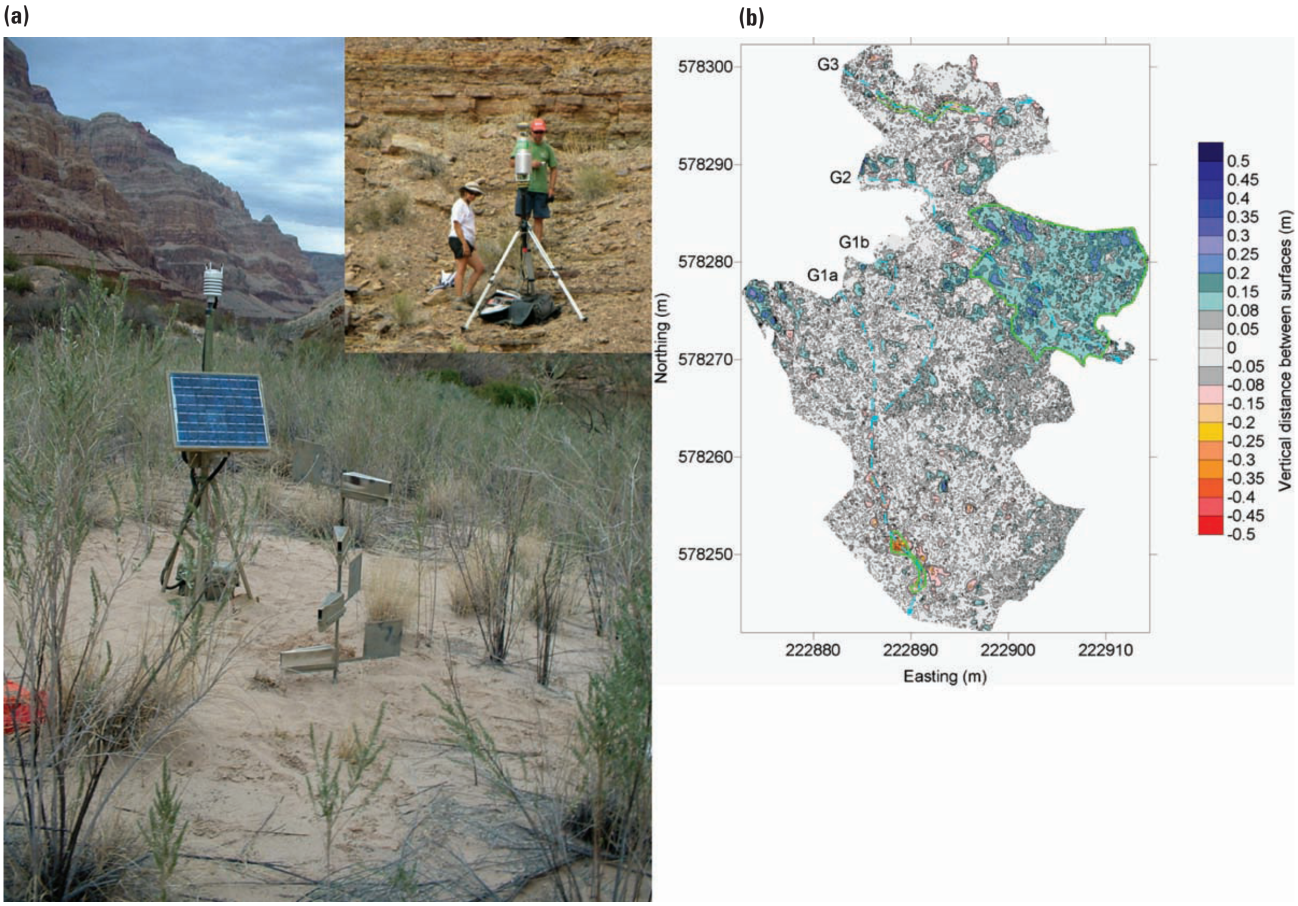

Figure 4. (a) Weather stations and sand traps positioned throughout the river corridor gather detailed data on wind velocity and direction, precipitation, temperature, humidity, and barometric pressure, and the amount of sand transported under varying weather conditions, while modern survey tools, such as terrestrial light detection and ranging, allow scientists to accurately quantify any physical changes occurring at archaeological sites in relation to these ongoing ecological processes. These data in combination can be used to assess relations between local and regional weather conditions, changing sediment-supply conditions, and erosion or stability of archaeological sites. (b) This map illustrates topographic changes monitored at one archaeological site along the Colorado River between May 2006 and May 2007. Red areas document erosion while blue areas show where sediment was deposited (from Collins and others, 2009).

combination with comparatively dynamic ecological factors, such as sediment supply and vegetation, contribute to archaeological site stability and change through time. Ultimately, these data will be useful for developing and refining more complex ecosystem-based models (e.g., Wainwright 1994; Walters and others, 2000) to allow scientists and managers to more accurately predict which sites are most vulnerable to future degradation, which ones may benefit most from implementing erosion-control measures or other preservation actions, and how future changes in dam operations may affect long-term site stability.

Implications for Management

The ultimate goal of this research project is to develop objective, quantitative monitoring protocols for assessing status and trends in archaeological site condition (stability) on a systemwide basis and to be able to directly measure whether and how rapidly resource condition is changing in relation to current dam operations, local weather patterns, and other interactive ecosystem controls. Through using an ecosystem-based approach, we are “unpacking” the concept of site condition so that we can relate measured changes to specific ecosystem processes that contribute to the stability or degradation of archaeological sites. By designing the monitoring program around a conceptual model of interacting ecosystem processes, monitoring data can be collected and reported in a manner that allows scientists and managers to independently evaluate the role of dam operations relative to other environmental factors that contribute to changes in site condition over time. The data generated by this project and by the future long-term monitoring program will be useful for informing managers on how potential modifications to dam operations, in combination with other environmental factors and ongoing mitigation efforts, may affect archaeological site condition. The data may also have utility for constructing future risk assessment models that can predict the relative stability of archaeological sites in a dynamic landscape setting. These results can then be used by managers to guide their selection of the most appropriate management options for improving site stability and achieving preservation objectives (Pederson and others, 2006). While monitoring data can accurately document the amount and rate of physical changes occurring at archaeological sites and can relate those changes to the dam-influenced ecosystem processes operating in the Colorado River corridor today, determining whether the resulting condition of archaeological sites in the CRE should be judged as "good," "fair," or "poor" will ultimately depend on the specific value system and explicit goals of the management agencies that are responsible for preserving and interpreting these nonrenewable cultural resources.

Acknowledgments

This project reflects the hard work and thoughtful input of numerous individuals. Thanks are due first and foremost to Mark Miller, who initially recognized the utility of the Jenny-Chapin model for structuring monitoring programs in the American Southwest. Brian Collins, Jen Dierker, Amy Draut, Lisa Leap, Gary O'Brien, and Joel Pederson contributed their wealth of personal knowledge and professional expertise in helping us devise and evaluate approaches for monitoring change at archaeological sites. We also thank the Bureau of Reclamation, Upper Colorado Region, for funding this research, Carol Fritzinger for organizing the field logistics, and the boatmen and women from Humphrey Summit Support who safely transported us down the Colorado River. Finally, we thank the two anonymous reviewers for their thoughtful and helpful comments on an earlier draft.

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The Development of Two Portable and Remote Scanning Systems for PIT Tagged Fish in Lentic Environments

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Abstract

Two portable passive integrated transponder (PIT) scanning units were developed and tested for monitoring razorback sucker (*Xyrauchen texanus*) in Lake Mohave, Arizona and Nevada, and Imperial Ponds on the Imperial National Wildlife Refuge (INWR), Arizona. One unit used mostly off-the-shelf equipment purchased from Biomark®, and the other unit was mostly home built with a user-constructed antenna, an Allflex® tag reader, and a custom-built logger board. Biomark® units in Lake Mohave contacted 167 unique fish in 1,400 hours of scanning and about 30 man-hours of effort. Allflex® units in Imperial Ponds contacted 38 unique fish in 22 hours of scanning and about 1 man-hour of effort. Biomark® units require less time to develop and fewer technical skills to operate than Allflex® units, but Allflex® units cost $800 each while Biomark® units cost $11,500 for a two-scanner system.

Introduction

Passive integrated transponder (PIT) tags have been used in fisheries research for nearly 30 years. Their small size, long life, and individual identification have made them a powerful tool in fisheries management. In the past, tagged fish had to be captured and handled for individual identification. However, recent technological advances have increased reception range allowing for remote sensing of PIT tags, i.e., identifying a tagged fish without capturing it. Portable PIT scanners or "PITpacks" (Hill and others, 2006) have been used to monitor behavior, movement, and habitat use of fishes in shallow waters of small streams (Roussel and others, 2000; Zydlewski and others, 2001; Riley and others, 2003; Roussel and others, 2004; Cucherousset and others, 2005). Fish movement has also been monitored in larger streams by using units that are usually permanently or semipermanently mounted to the substrate or manmade structure (Lucas and others, 1999; Bond and others, 2007; Enders and others, 2007), although attachment to a structure is not required (Connolly and others, 2008). Off-the-shelf PIT scanner components from fisheries companies as well as home-built components have proven effective. Less studied is the application of remote PIT sensing technology in lakes and ponds.

In the lentic waters of lakes and ponds, mark-recapture analyses often are used to estimate life-history parameters and population size. Data are acquired through marking and recapturing fish, requiring repeated handling of fish, which often is stressful to the study animals (Paukert and others, 2005). In addition, capture methods usually result in bycatch and incidental mortality and require crews of two to three people working multiple days to acquire adequate data for analysis. Portable PIT scanner units may be used to augment or completely replace data from these techniques in mark-recapture analyses. The effectiveness of a PIT scanner unit in a large lake or pond environment is unknown and is likely species specific. As part of ongoing monitoring projects, two portable, remote PIT scanner units were developed to target shallow, less than 3 meter (m), lentic waters of ponds and lake margins. Both projects focused on razorback sucker (*Xyrauchen texanus*), an endangered, benthic, endemic species of the Colorado River. The equipment brands used in this study were familiar to the researchers involved and should not be construed as an endorsement.

Methods

The first unit was based on off-the-shelf equipment purchased from Biomark® (fig. 1). Each Biomark® unit was set up to run two FS 2001F-ISO readers with individual batteries (Werker U1DC deep cycle lead acid 31 ampere-hour (Ah) or A12-33J AGM sealed gel cell 33 Ah or equivalent) and two Biomark® 660 x 305-millimeter (mm) flat plate antennas.

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Flat plate antennas were selected because of their negative buoyancy, which serves to anchor the instrument housing in place. These scanners and antennas are designed to detect 134.2 kilohertz (kHz) full-duplex PIT tags.

Scanner units, tuning boxes, and batteries were housed in a Sherpa 50-quart series cooler by Yeti™, which features “O” ring type lid seal, rubberized latch closure, and high-strength lifting handles. The lid was fitted with a 204-mm clear polycarbonate inspection hatch for instrument observation. Two 102 x 25-mm polyvinyl chloride (PVC) pipe reducers were fitted in the lid to allow cable connections, which were sealed with split and cored no. 5 rubber stoppers. Optional stability pontoons of capped and sealed 762 x 102-mm acrylonitrile-butadiene-styrene (ABS) pipe were affixed to the sides of the housing with 25-mm nylon webbing and over center or “quick lock” type buckles through 25-mm stainless steel footman’s loops, which were through-bolted to the housing with 51-mm, 10 x 24 stainless machine screws and stainless nylock nuts with stainless fender and neoprene washers sealing the screw holes.

Antennas were tethered with 5 m of 6-mm polypropylene rope to act as strain relief for the antenna cables, and 1-m loops of polypropylene were affixed to the swing-out attachment flanges of the antennas, providing boat-hook contacts for deployment and pickup. Interference between antennas was avoided by maintaining a minimum separation of 3 m. The system was tested in high-wind conditions that generated 1-m waves without water intrusion, which could lead to instrument failure. Some drifting of antenna placement was experienced in high-wind conditions. Length of deployment time with continuous operation was up to 48 hours with fully charged batteries. The range of deployment depth was 0 to 4 m.

Each antenna was tuned during deployment by adjusting a Biomark® tuning box connected inline between the reader and antenna cable within the cooler. Tuning boxes have a fine-tuning adjustable dial, a rough-tuning switch (+ or –), and jumper switches within the box for greater tuning range. Jumper settings were generally adjusted in the laboratory. Field tuning involved adjusting the fine-tuning dial until a maximum output current was achieved. Output current was read directly from the PIT scanner display. Read range was then estimated by passing a PIT tag encased in epoxy and mounted to the end of a 2-m section of 25-mm PVC pipe over each flat plate antenna at various depths, which were estimated visually to the nearest 50 mm. At the end of deployment, a test PIT tag was passed over each antenna to ensure the unit was still operational.

The second unit was mostly home built with a user-constructed antenna consisting of six turns of 12 American Wire Gauge (AWG) stranded copper wire encased in 38-mm PVC pipe (2.3 x 0.7-m rectangular pipe frame) and attached to an Allflex® scanner (fig. 2). Allflex® scanners are “naked” printed circuit boards with loose wires for antenna and power connection and two light-emitting diode lights to indicate scan rate and tag encounters. A rubberized water-resistant two-conductor 14 AWG cable connected the antenna to the scanner. The cable-PVC interface at the antenna was made watertight by passing the cable through a PVC cap and filling the inside of the cap with two-part epoxy before cementing the cap in place.

Each unit was powered by a Power-Sonic® 12-volt, 26-Ah battery and connected by way of a serial cable to a data logger. Data loggers were custom built and provided by Cross Country Consulting, Inc. (Phoenix, AZ). The scanner, data logger, and battery were stored in a sealed model 1520 Pelican™ case. Allflex® scanners sent tag data to the loggers by way of serial interface. Data loggers recorded tag numbers and a date-time stamp for each tag encountered.

A Coleman® model CL-600 solar charger was mounted to the top of the Pelican case and wired to the battery to extend deployment time. Cables running through the case were passed through 13-mm cable grips to maintain a water-resistant seal. The case was placed inside a black inner tube to increase stability on the water. Data were downloaded from the data loggers to a laptop or personal digital assistant by way of a serial cable.

The antennas were positively buoyant, so weights made of 76-mm ABS pipe filled with concrete were attached to the antennas during deployment. Antennas could be oriented flat, standing on long end or short, and placed anywhere in the water column. Total deployment time depended on light conditions and varied from 4 days (no light) to 2 weeks. Allflex® scanner units can detect both half and full-duplex 134.2-kHz PIT tags.

Jumper switches on Allflex® scanners are used to tune antennas. Antennas were tuned in air in the laboratory, with only minor adjustments required before deployment in the field. Allflex® scanners have no display, so a standard multimeter was attached inline with the positive battery terminal to measure scanner current for tuning in the laboratory. Jumpers were added in sequence until peak current was achieved. Field tuning was based

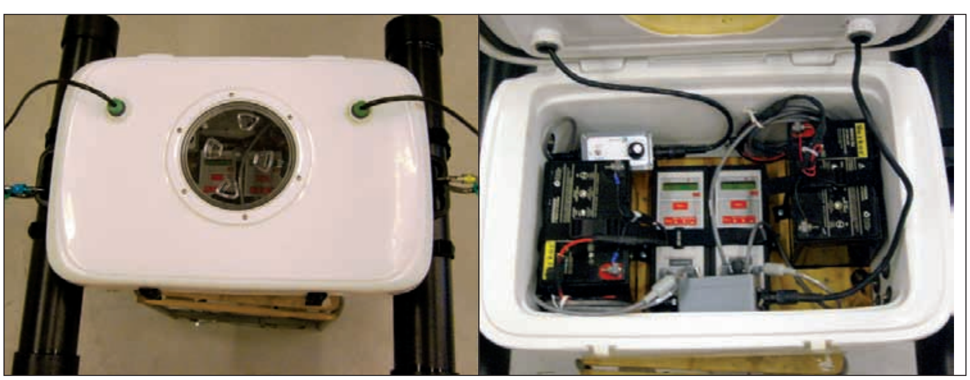

Figure 1. A remote passive integrated transponder (PIT) scanning unit built inside a 50-quart cooler containing two Biomark® FS 2001F-ISO readers and two deep-cycle lead acid batteries.

on achieving maximum reception range. Reception range was tested by approaching the antenna with a palmed PIT tag underwater. Reception range was visually estimated to the nearest 50 mm. At the end of deployment, a test PIT tag was passed through the center of the antenna to ensure the unit was still operational.

Biomark® units were deployed along the shore in Lake Mohave, Arizona and Nevada, between February 13 and May 1, 2008 (fig. 3, top). During this time, a total of 60 deployments were made. Razorback sucker have been PIT tagged and stocked into Lake Mohave for nearly 20 years, but only recently have they been tagged with 134.2-kHz full-duplex PIT tags. The total number of surviving razorback sucker with these tags is unknown. Deployments were monitored, and time-stamped video and images of fish interacting with the antennas were taken.

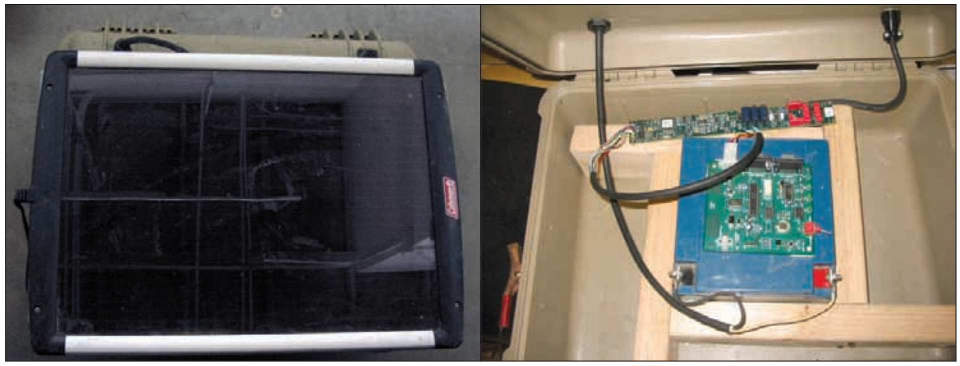

Figure 2. A remote passive integrated transponder (PIT) scanning unit built inside a 1520 Pelican™ case using an Allflex® scanner, a custom logger, a sealed lead acid battery, and a Coleman® model CL-600 solar charger.

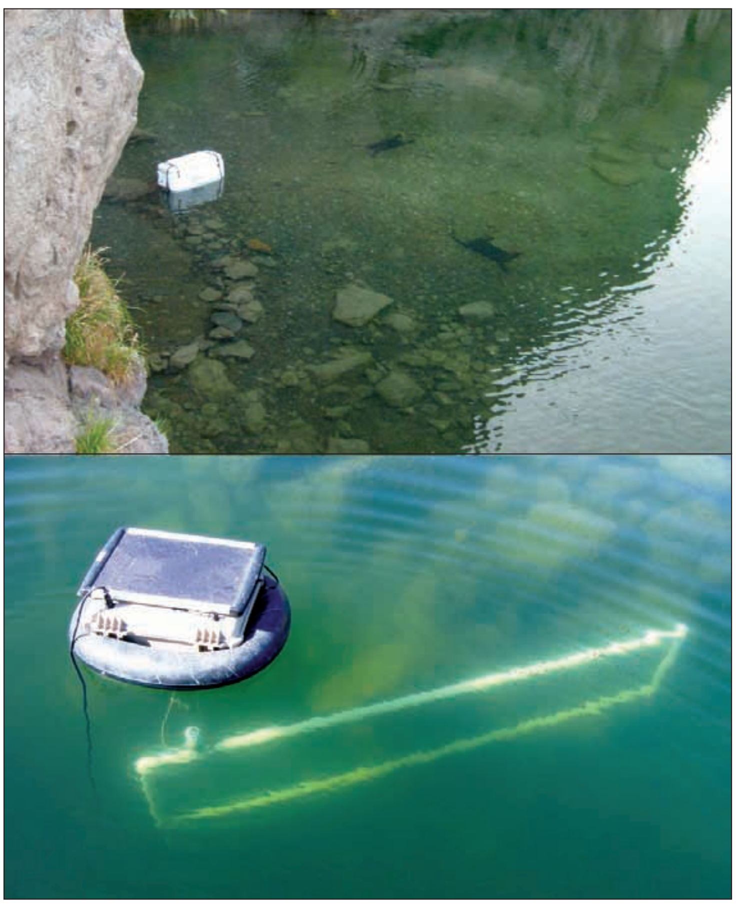

Figure 3. Passive integrated transponder (PIT) scanning units after deployment; a Biomark® in Lake Mohave, AZ-NV (top), and an Allflex® unit in Imperial Ponds, Imperial National Wildlife Refuge, AZ (bottom).

Initial testing of Allflex® units (fig. 3, bottom) was conducted in a 10.2 surface-acre pond in Imperial National Wildlife Refuge (INWR), Arizona. Two units were deployed from August 19 to 21, 2008. The pond was stocked with 272 PIT tagged razorback sucker on November 5, 2007. Visual monitoring of any kind was not feasible in this pond because of a lack of water clarity. Multiple additional deployments have been made since.

Results and Discussion

Biomark® unit deployments in Lake Mohave resulted in 1,731 contacts, of which 167 were unique tags. Total scan time was 1,400 hours, and effort was estimated at 30 person-hours. This relatively small amount of effort contacted nearly as many tagged razorback sucker as annual sampling events in the lake that involve tens of people and hundreds of person-hours. Razorback sucker were observed in shallow-water spawning groups swimming around and over antennas and did not appear affected by the presence of equipment. Allflex® units deployed in the INWR pond recorded 59 contacts of which 38 were unique. Total scan time was 22 hours with an estimated effort of one person-hour. This small effort resulted in contact with nearly 24 percent of the population in the pond based on a mark-recapture population estimate of 160 fish conducted in the same month.

Reception range was similar between the two units at about 250 mm above the antenna surface, but the PVC pipe antennas were larger and, therefore, had a larger scanning “footprint.” In ponds where depth was shallow (less than 3 m) and size was small (less than 15 surface-acres), scanner units were extremely effective. In large bodies of water, the behavior of the species was critical. Razorback sucker occupied shallow waters and did not appear to be affected by the presence of equipment. The design and scanning range of Biomark® flat plate antennas likely restrict their use to demersal species, although other antenna designs, not tested in this study, are available from Biomark®.

Cost was considerably less for the Allflex® units, about $800 compared to $11,500 for the two-antenna Biomark® unit, but labor costs were excluded because costs vary from researcher to researcher. Allflex® units required substantially more technical skill and construction time. The initial investment in remote sensing is substantially higher compared to nets and traps given per unit cost of Biomark® units and labor costs of Allflex® units. However, both systems required minimal manpower once built and debugged. Deployment and retrieval of each unit required less than 10 minutes. Long-term maintenance costs and longevity of each unit were not assessed in this study.

Data acquired from remote sensing are similar to data from sonic or radio telemetry (Enders and others, 2007). However, telemetry tags are relatively expensive, have a limited lifespan, and often require surgery, which limits the number of fish that can be used in a study. Radio and sonic tags are also large enough that their presence alone may affect results. PIT tags have an unlimited lifespan and can be injected with a needle in a matter of seconds, increasing the number of fish that can be used in a study at least by an order of magnitude given a similar effort and budget.

Implications for Management

The advances in PIT scanning technology have led to a broad range of remote-sensing applications that can reduce the need for capturing and handling fish species of interest in nearly every aquatic environment, even in large reservoirs, if the species occupies shallow water. This reduction in capture and handling can also benefit nontarget species that end up in nets as bycatch. This reduction in bycatch can also bolster public support for research in cases where nontarget species have sport or commercial value. Costs can be kept at a minimum if a researcher has the time and technical inclination to build antennas and use Allflex® or similar basic scanner units. Biomark® provides quality equipment when budget is less of a concern than time.

Acknowledgments

Collections were under permit authorization of U.S. Fish and Wildlife Service, and the State of Arizona. Study fish were provided by Willow Beach National Fish Hatchery, and funding for this research was provided by the Bureau of Reclamation, Boulder City, NV.

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Using Changes in Bed-Surface Grain Size as a Proxy for Changes in Bed Sand Storage, Colorado River, Grand Canyon

By Robert Tusso,[1] David M. Rubin,[2] David J. Topping,[1] Hank Chezar,[3] and Michael Breedlove[4]

Abstract

Sand transport in the Colorado River downstream from Glen Canyon Dam, Arizona, is regulated by changes in riverbed grain size and changes in discharge. The dam and its operations have resulted in substantial changes in the amount of sand storage and sand discharge in the Colorado River in Grand Canyon. With the upstream supply of sand cut off by the dam, tributary floods are the only remaining sources of new sand, and they result in a fining of the sand on the bed of the river. Intervening dam releases winnow this bed sand, with net transport downstream. Although bed sand storage data are important for managing sand resources in Grand Canyon National Park, these data are difficult to collect. Measurements of riverbed grain size, in contrast, are easier to collect over the broad scale of Grand Canyon. This report evaluates the relations between changes in the volume of bed sand and changes in bed-surface grain size, with the goal of identifying whether changes in surface grain size could be used as a proxy for changes in bed sand storage. This study compares the changes in these two parameters over four intervals, with varying hydrologic and sedimentologic regimes. During a long period without large tributary sand inputs, the overall trend was toward bed coarsening, although no significant patterns in bed elevation change were observed. During a period of large tributary sand inputs, the overall trend was toward fining and aggradation, with degrading areas showing a higher propensity for coarsening than aggrading areas. Although no consistent pattern was evident for all conditions or all times, insight was gained into the effects of certain dam operations, such as high-flow events. Recognizing these patterns will aid in understanding the mechanics of sediment transport in this system, enabling scientists to better assess the effects of various events, thus providing knowledge valuable for the management of Glen Canyon Dam.

Introduction

To assess the effects of dam operations on the Colorado River in Grand Canyon, the movement of sand on the bed of the river must be monitored (Topping, Rubin, and Vierra, 2000; Topping, Rubin, and others, 2000). Knowing the quantity and location of sand in storage is important for calculating sediment budgets and understanding the mechanics of sand transport during both normal dam operations and experimental high flows. In any given region of the bed, measuring and correlating changes in sand storage to changes in bed-surface grain size can help identify patterns by which sand is transported in response to different dam operations and sediment input conditions. This, in turn, can lead to more efficient and thorough investigatory techniques to further aid decisionmakers in the management of Glen Canyon Dam.

Methods

Five repeat surveys of river bathymetry (compiled from sonar, level rod, and light detection and ranging (lidar) data) were conducted between 2000 and 2004 (fig. 1A) over seven short reaches of the Colorado River between river miles[5] 1 and 88 (fig. 1B). Bathymetric surveys were conducted using methods described by Kaplinski and others (2009), and bed-surface grain size was collected using methods described by Rubin and others (2007). Although 11 study reaches have been identified, only reaches 2–8 have complete survey data for the intervals examined here.

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[5] Distances along the Colorado River in Grand Canyon traditionally are measured in river miles upstream or downstream from Lees Ferry, AZ.

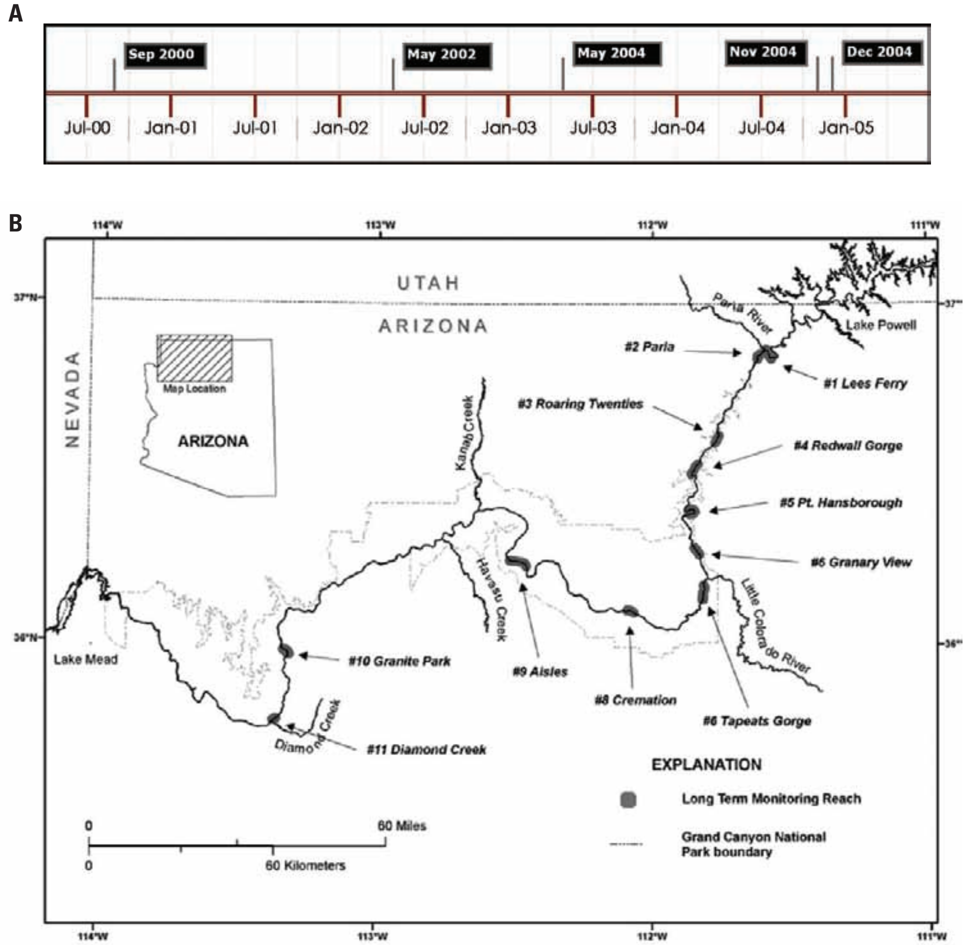


Figure 1. (A) Dates the five bathymetric surveys were conducted and (B) the location of the seven reaches surveyed.

Using these data, maps were created showing the change in bed elevation between each bathymetric survey, revealing the change in the volume of sand in storage on the bed over each interval. Discrete areas exhibiting aggradation or degradation were identified and overlain with bed-surface grain-size measurement points. The data were compared using two different methods to reveal relations between changes in sand storage volume and bed-surface grain size: (1) the "polygon method," which identifies regions of the bed that underwent change and calculates the mean grain-size change within each region, and (2) the "nearest-neighbor method," which compares grain-size point measurements to proximal point measurements from a subsequent survey, in terms of both grain-size change and elevation change.

Polygon Method

Each bathymetric survey yielded a three-dimensional surface model of the riverbed within the surveyed reaches. Comparison of back-to-back surveys reveals specific regions that have aggraded 10 centimeters (cm) or more (increased sand storage volume) and degraded 10 cm or more (decreased sand storage volume); 10 cm was chosen to account for error in the bathymetric surveys. These regions were then overlain with point grain-size data from the two constituent surveys, and the change in mean grain size was determined for each region, allowing the regions to be grouped into one of four categories: (1) aggraded and coarsened, (2) aggraded and fined, (3) degraded and fined, or (4) degraded and coarsened. Although the discrete regions vary in area, each region is subject to a unique sand supply, so in our analysis we have tabulated the number of regions rather than summing the area of all regions with similar parameters and thus letting larger regions skew the data.

Figure 2 shows a sample reach where the volume of sand stored on the bed changed from May 2004 to November 2004, overlain with point grain-size measurements (in millimeters) for each survey, including the before/after change in mean grain size for each region from survey to survey. During the sample period shown in figure 2, there were large sand inputs from the Paria River and there were lower dam releases.

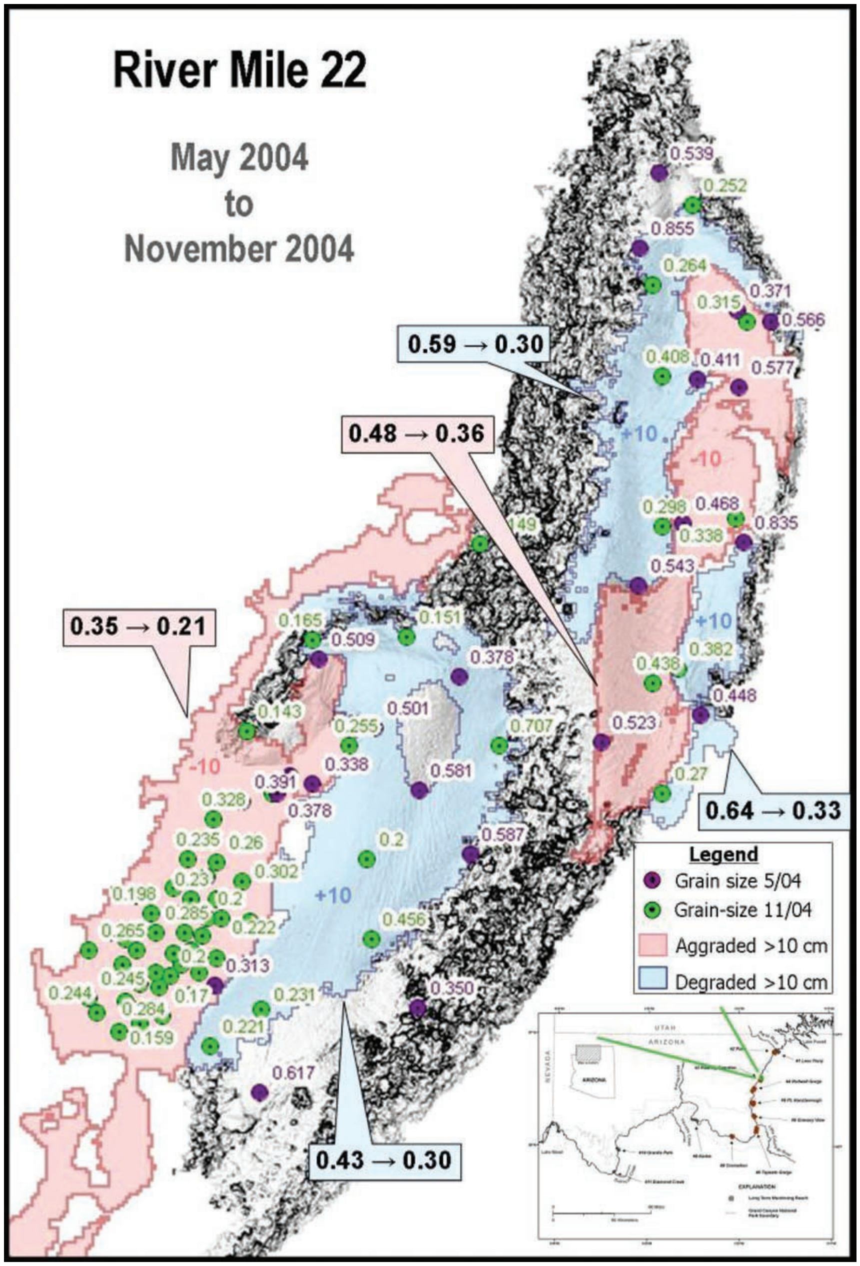


Figure 2. Regions from study reach 3, near river mile 22, that aggraded (blue) and degraded (red) from May 2004 to November 2004, a period of large sediment inputs, overlain with point grain-size measurements (in millimeters) and labeled with the change in mean grain size from survey to survey.

Figure 3 shows the regions where sand volume changed from November 2004 to December 2004, overlain with point grain-size measurements and the before/after change in mean grain size. During this period, there were minimal sand inputs from the Paria River and large dam releases related to an experimental high flow in November 2004 (Topping and others, 2006), herein referred to as the 2004 BHBF (beach/habitat-building flows).

Nearest-Neighbor Method

Bed grain-size observations from two successive surveys were plotted in a geographic information system (GIS). Then using the older survey, for example November 2004, as the baseline (fig. 4), the nearest point from the more recent survey, in this case December 2004, was identified using a maximum radius of 10 meters (m). This radius was chosen to give an adequate number of samples for the analysis based

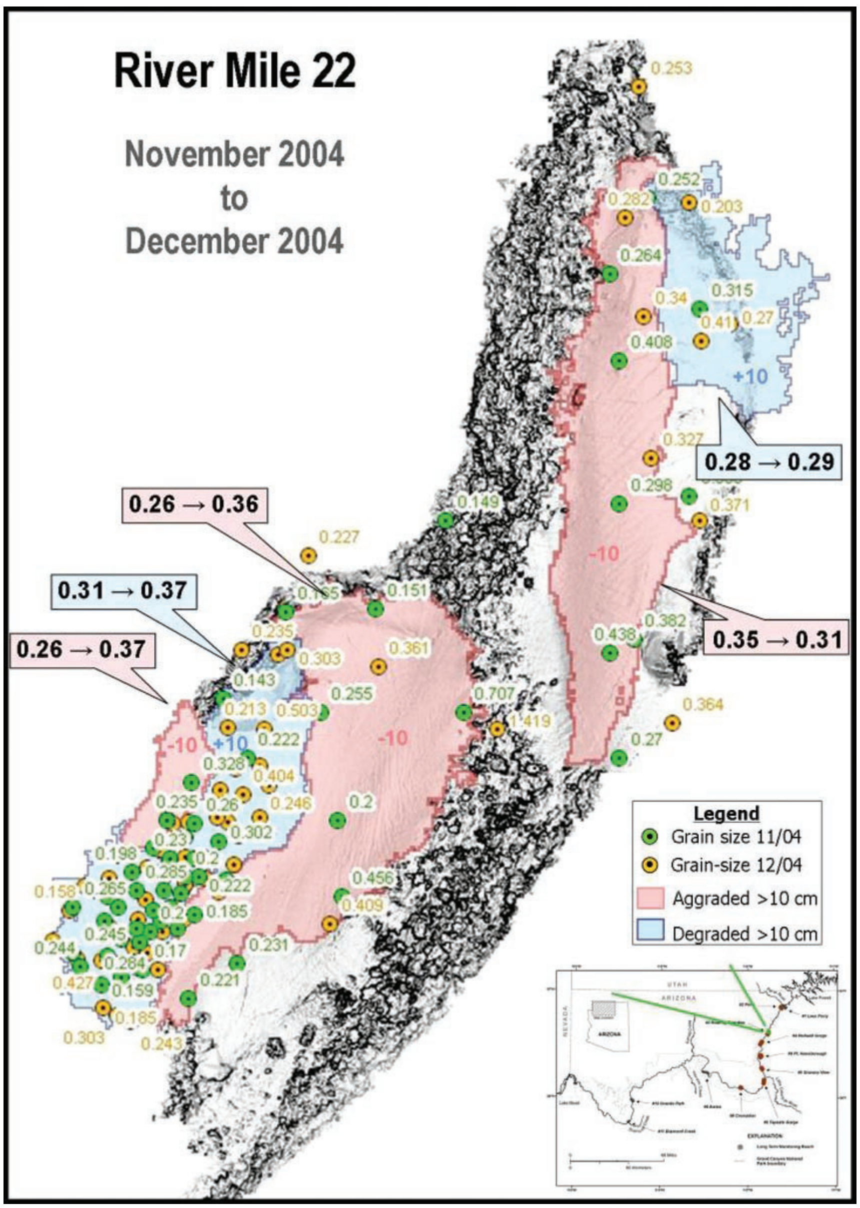


Figure 3. Discrete regions from study reach 3, near river mile 22, that aggraded (blue) and degraded (red) from November 2004 to December 2004, during which the 2004 beach/habitat-building flows (BHBF) experiment was conducted, overlain with point grain-size measurements (in millimeters) and labeled with the change in mean grain size from survey to survey.

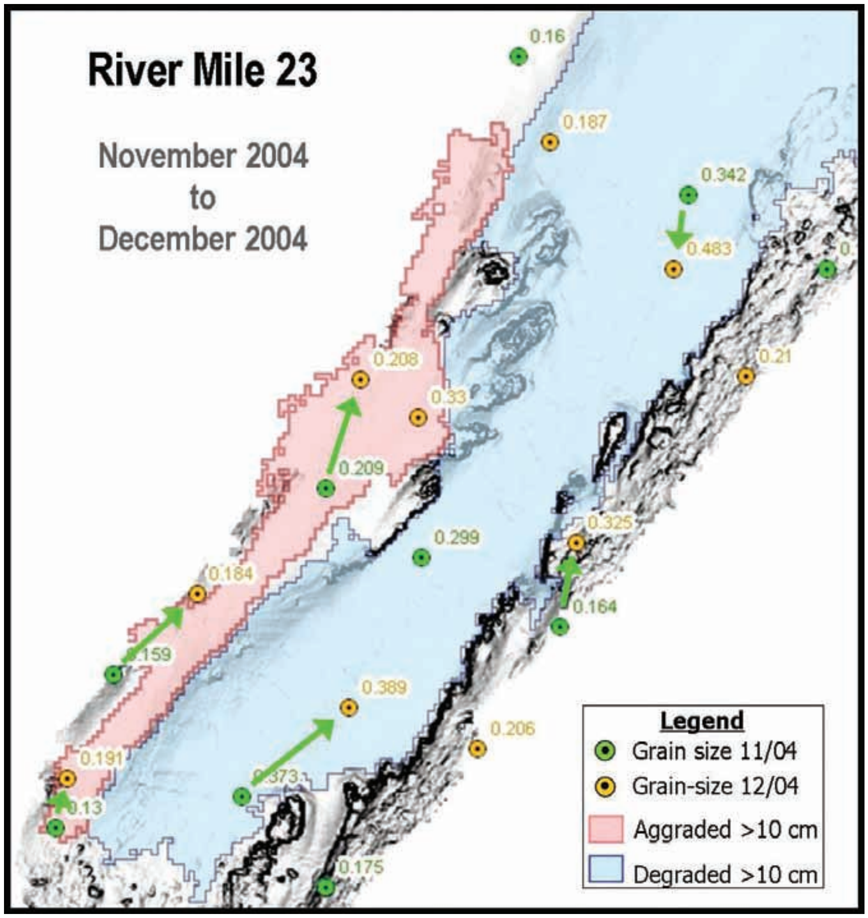


Figure 4. Point grain-size measurements from back-to-back surveys were plotted in a geographic information system (GIS). The nearest subsequent-survey neighbor to each previous-survey point was identified, and the change in grain size was calculated. Also extracted was the change in bed elevation at each previous-survey point.

on measurement density, maximum river depth, and the uncertainty of the point locations caused by the river current displacing the cable-attached camera. The difference in grain size between the two points was recorded, as was the change in bed elevation over the specified time interval at the older set of points. These data were then plotted to show change in grain size versus change in elevation, producing four classifications: (1) aggraded and coarsened, (2) aggraded and fined, (3) degraded and fined, or (4) degraded and coarsened.

Results

The polygon data show that, as a general rule, tributary sand inputs during lower dam releases result in a fining of the bed and an increase in bed elevation, whereas a lack of tributary sand inputs results in a coarsening of the bed with increases or decreases in bed elevation. Table 1 shows the bed response during the intervals between surveys. From September 2000 to May 2004, a period encompassing the first two intervals that saw little tributary activity, coarsening dominated, and the regions that fined were more likely to show aggradation. From May 2004 to November 2004, a period of large tributary sand inputs, fining dominated, especially in regions that aggraded. Although degrading and coarsening was the most common response from November 2004 to December 2004 (2004 BHBF), it was not as dominant as might be expected for an event capable of exporting large amounts of sediment. Reach-by-reach investigation of this event (fig. 5) shows that fining is more associated with degradation downstream.

Although the nearest-neighbor analysis does not illustrate patterns clearly on its own, it does support some of the patterns identifiable from the polygon data. The overwhelming trend from September 2000 to May 2002 (fig. 6A) and May 2002 to May 2004 (fig. 6B) was coarsening of the bed, although no strong aggradation/degradation signal can be found. The large sand inputs from May 2004 to November 2004 (fig. 6C) can be recognized in the large number of points that aggraded and fined. Although the large number of points that aggraded during the 2004 BHBF (fig. 6D) can be largely attributed to collection methods that emphasized eddy sandbars, the trend toward coarsening can only be attributed to the winnowing effects of the higher flow.

Table 1. The number of regions of the bed having each type of response during the intervals between surveys. From September 2000 to May 2004, a period encompassing the first two intervals that saw little tributary activity, coarsening dominated. From May 2004 to November 2004, a period of large tributary sand inputs, fining dominated, especially in regions that aggraded.

	9/2000 – 5/2002	5/2002 – 5/2004	5/2004 – 11/2004	11/2004 – 12/2004	All intervals
Aggraded and fined	1	11	30	13	55
Degraded and fined	1	3	17	17	38
Aggraded and coarsened	11	27	5	23	66
Degraded and coarsened	14	27	7	25	73

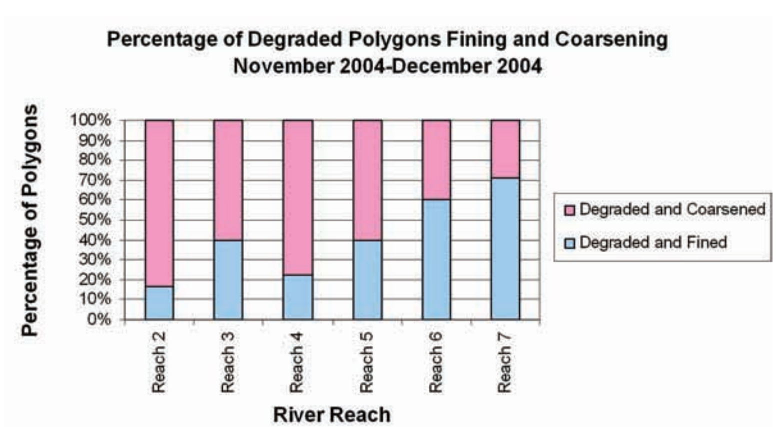
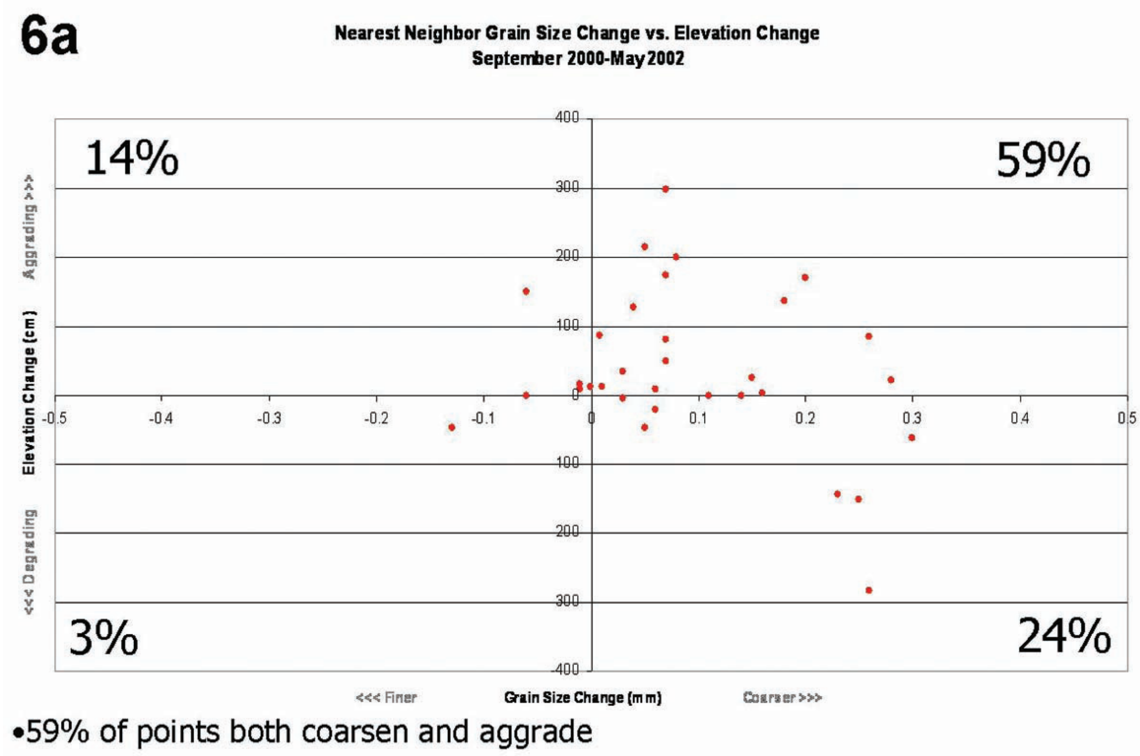


Figure 5. During the 2004 beach/habitat-building flows (BHBF) experiment, areas that degraded were more likely to coarsen in the upstream reaches and more likely to fine in the downstream reaches.



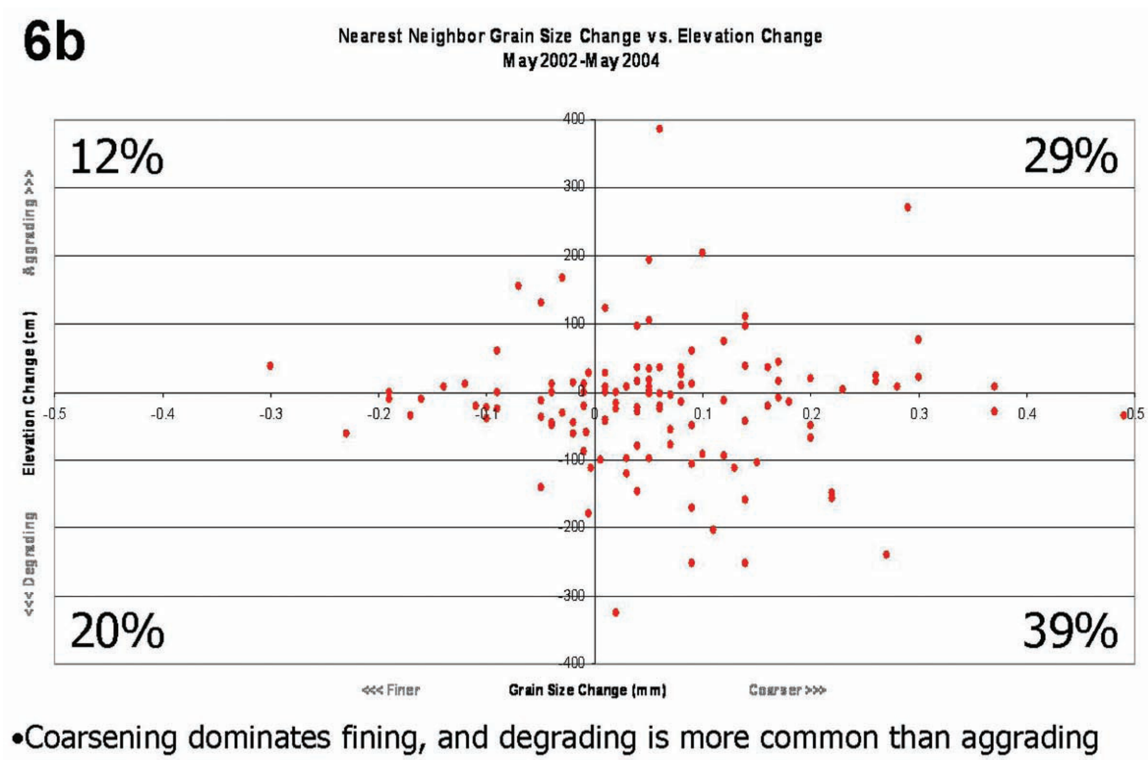
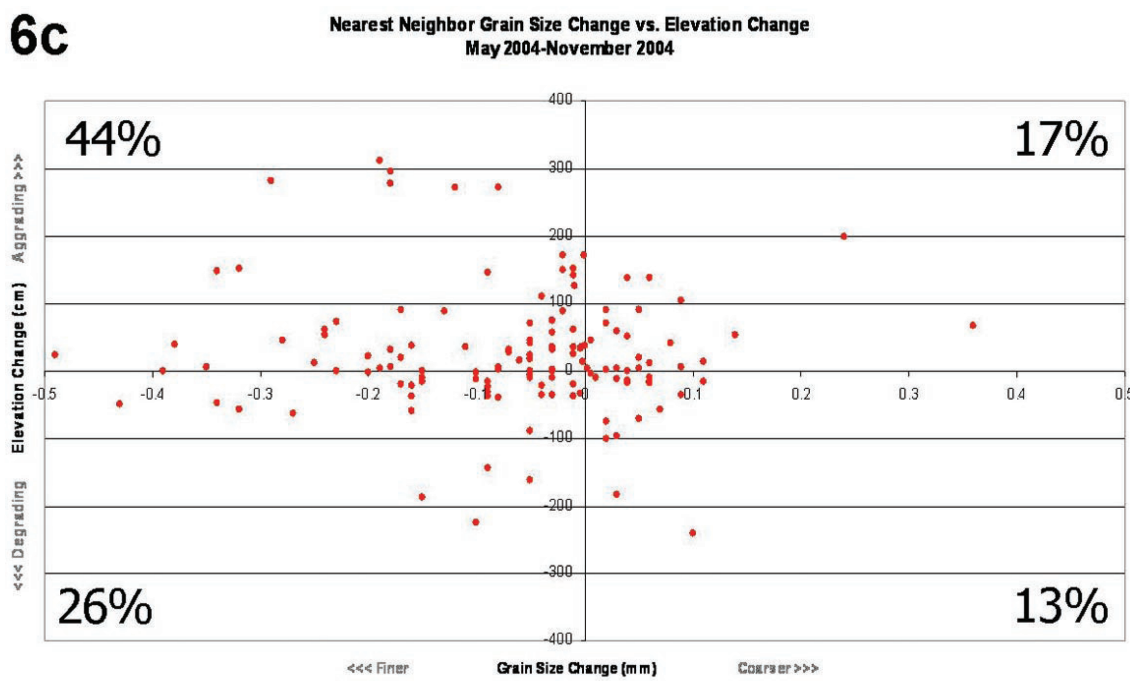
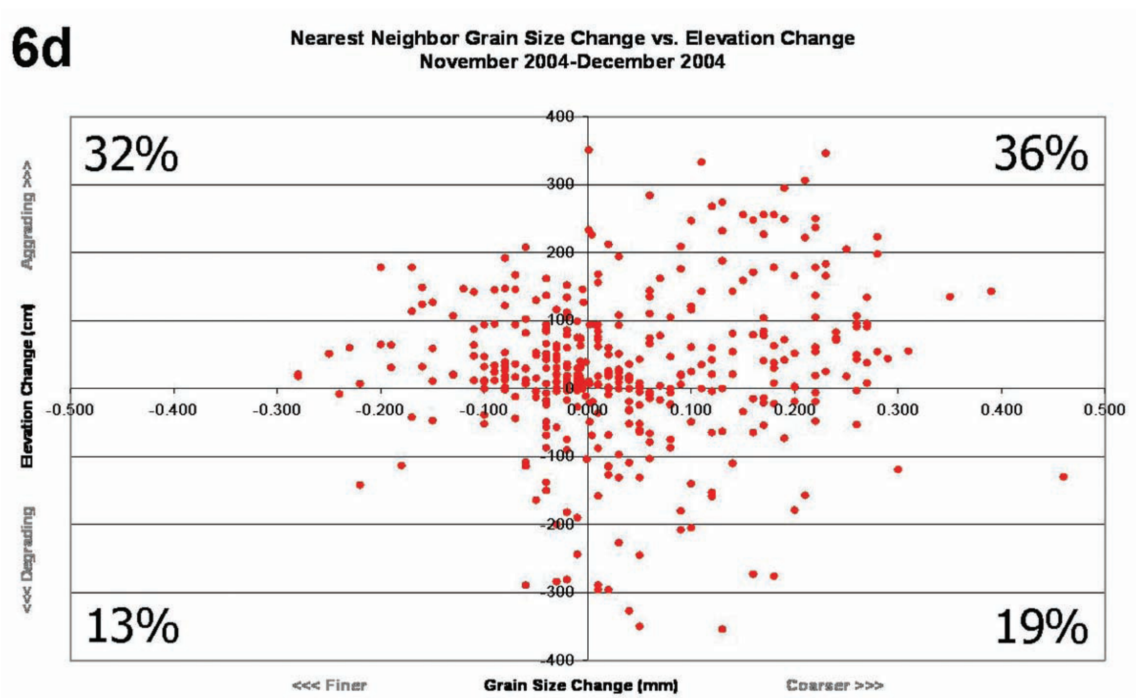


Figure 6. Plots from the nearest neighbor analysis for each survey interval showing relations between changes in riverbed elevation and bed-surface grain size. During the first two intervals, there were minimal tributary sand inputs and coarsening dominated, but during the third interval, there were large tributary sand inputs and fining and aggradation were more prevalent. During the 2004 beach/habitat-building flows (BHBF) experiment (interval 4), the large number of points that aggraded can be largely attributed to collection methods that emphasized eddy sandbars, although the trend toward coarsening can only be attributed to the winnowing effects of the higher flow.



•Fining dominates coarsening, and aggradation dominates degradation



•Coarsening is more common than fining, and aggradation dominates degradation

Figure 6. (continued) Plots from the nearest neighbor analysis for each survey interval showing relations between changes in riverbed elevation and bed-surface grain size. During the first two intervals, there were minimal tributary sand inputs and coarsening dominated, but during the third interval, there were large tributary sand inputs and fining and aggradation were more prevalent. During the 2004 beach/habitat-building flows (BHBF) experiment (interval 4), the large number of points that aggraded can be largely attributed to collection methods that emphasized eddy sandbars, although the trend toward coarsening can only be attributed to the winnowing effects of the higher flow.

Implications for Management

Bed-sediment grain size is important because it influences suspended-sediment concentrations, turbidity, and sediment export down the Colorado River. Changes in grain size in relation to aggradation and degradation of the riverbed were investigated. The results of this study indicate that no single relation exists between these two parameters under all flow and sediment-supply regimes. However, examination of these changes indicates specific responses to particular events. During a period of large tributary sand supply and lower dam releases (May 2004 to November 2004), sites that fined exhibited aggradation at a nearly 2:1 ratio to degradation, and sites that aggraded exhibited fining at a 6:1 ratio to coarsening, suggesting a relation between aggradation and fining. Periods with minimal tributary sand inputs or higher dam releases exhibit coarsening, with no unique relation between changes in grain size and changes in bed elevation. Although bed sand storage response to high-flow events is complicated, mapping the bed texture response contributes to the overall understanding of the effects and dynamics of these events.

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Use of Specific Conductance in Estimating Salinity and as a Natural Tracer of Water Parcels in the Colorado River Between Glen Canyon Dam and Diamond Creek, Northern Arizona

By Nicholas Voichick[1] and David J. Topping[1]

Abstract

In the Colorado River in Grand Canyon, specific-conductance data can be used both to estimate salinity and to track water parcels traveling downstream because of differences in the salinity of tributary and mainstem water. Salts entering the Colorado River, regulated by the 1974 Colorado River Basin Salinity Control Act, cause millions of dollars in damages annually to municipal, industrial, and irrigation water users. Collecting specific-conductance data using continuously monitoring water-quality instruments is a cost-effective method for estimating salinity (dissolved salts) in the Colorado River. These instruments have been used by the U.S. Geological Survey's Grand Canyon Monitoring and Research Center at seven sites to measure specific conductance of the Colorado River between Lake Powell and Lake Mead. The linear relation between specific conductance and total dissolved solids (a measure of salinity) has been established at two of the study sites, with an R-squared equivalent of 0.94 and 0.82 at the two sites. Specific-conductance data can also be used to track parcels of water traveling downstream in the Colorado River between Lake Powell and Lake Mead. Knowing the travel times of water parcels through this reach of the Colorado River is important for a variety of physical and ecological reasons, including assessing the transport of sediment in water and estimating the available food resource for fish and other aquatic organisms. The specific-conductance signal is especially evident and traceable downstream in the study area when two tributaries of the Colorado River exhibit particular flow patterns. Travel times and water velocities were calculated by tracking the specific-conductance signals from these tributary inputs. In one example, the water traveled from the Colorado River near river mile 30 to the Colorado River near river mile 226 (fig. 1) in approximately 83 hours at an average velocity of 1.06 meters per second (2.36 miles per hour).

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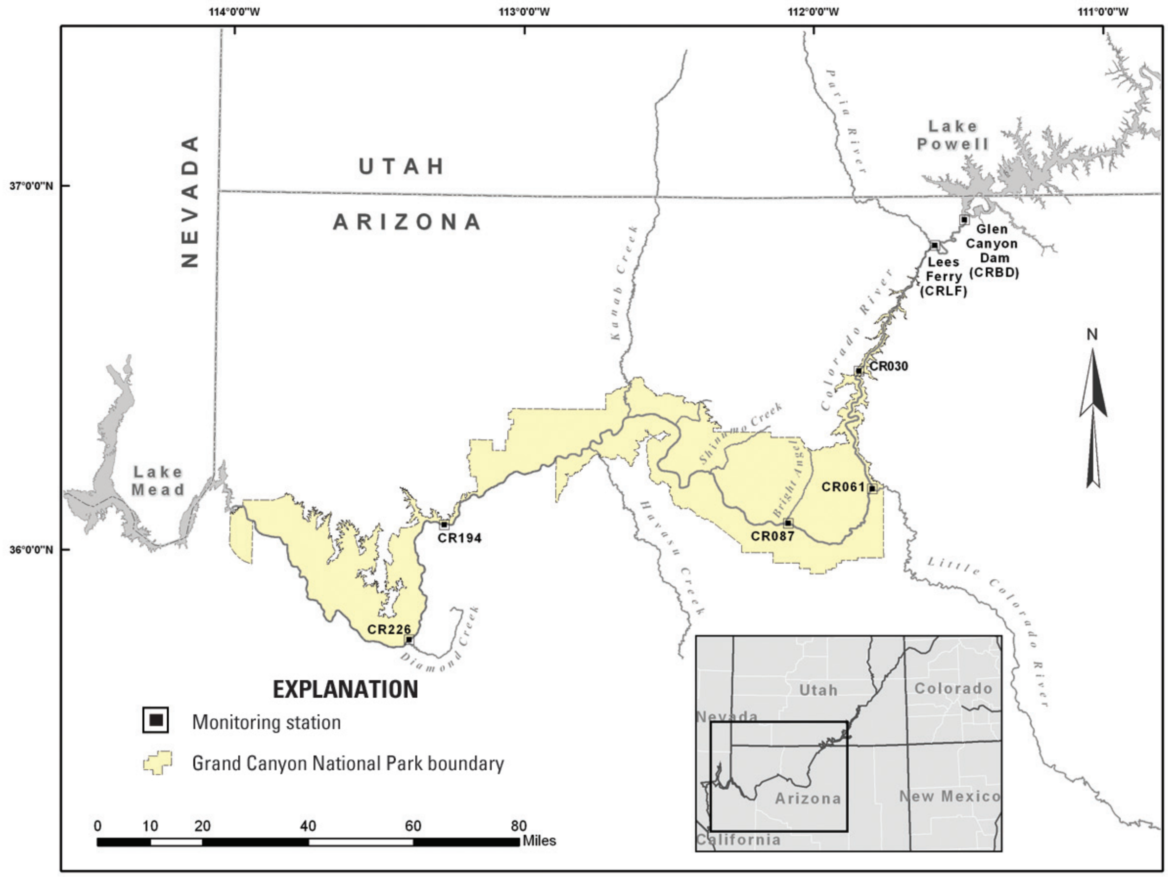
Introduction

Approximately 9 million tons of salt enters the Colorado River annually, about 50 percent from natural sources and 50 percent from human-caused sources (Bureau of Reclamation, 2003). The 1974 Colorado River Basin Salinity Control Act (Public Law 93–320) authorized the construction and operation of a basinwide salinity-control program. Damages caused by the input of salt into the Colorado River, which primarily affects municipal, industrial, and irrigation water users, are estimated to be $300 million annually (Bureau of Reclamation, 2003). Thus, monitoring the salinity of the Colorado River is of economic importance. From the mid-1970s to 2007, the salinity of the Colorado River at monitoring stations downstream from Lees Ferry (fig. 1) decreased, with periodic shorter term increases in salinity (Anning and others, 2007; Voichick, 2008). The short-term and long-term trends in the salinity of the Colorado River were likely caused by natural events, such as changes in precipitation, as well as human-caused events, such as the successful implementation of the salinity-control program (Anning and others, 2007; Anning, 2008). The U.S. Geological Survey's Grand Canyon Monitoring and Research Center has measured specific conductance at seven sites in the Colorado River between Glen Canyon Dam and Diamond Creek (fig. 1) using continuously monitoring water-quality instruments (Voichick, 2008). This data-collection effort is a cost-effective method for estimating salinity in the study area. Total dissolved solids (TDS) concentrations often are used as an indicator of salinity in freshwater systems. The linear relation between specific conductance and TDS was established at two of the study sites, allowing for salinity to be estimated from specific conductance in the study area.



Figures 1. The Colorado River between Lake Powell and Lake Mead, northern Arizona, and specific-conductance monitoring stations.

Water travel time is a useful parameter for analyzing several physical and ecological issues, including assessing the transport of sediment in water and estimating the available food resource for fish and other aquatic organisms. One approach used to measure water travel time of a river is by injecting dye in the water and tracking it downstream (Wilson and others, 1986; Kilpatrick and Wilson, 1989; Graf, 1995). Another method of measuring water travel time is tracking specific-conductance measurements downstream (Marzolf and others, 1999). The specific-conductance approach has the advantage of not injecting an artificial substance into the river, which is especially controversial in a national park. The specific-conductance measurements are collected by pre-programmed instruments; thus, this method does not require a large campaign of fieldwork and is also more cost effective. The Paria River during flood flow and the Little Colorado River during base flow contain saline water with particularly high specific conductance. These types of flows from these two tributaries produce high-specific-conductance spikes in the Colorado River. These specific-conductance spikes were tracked downstream in order to measure water travel time in the study area.

Methods

Conductivity (the reciprocal of resistivity) is a measure of a water-based solution's capacity to conduct an electric current and, thus, can be used to estimate the total dissolved salts in the water. Specific conductance usually is defined as conductivity normalized to 25 degrees Celsius, expressed in microsiemens per centimeter at 25 degrees Celsius (µS/cm). For this study, specific conductance was measured at seven sites in the field area (fig. 1) using instruments that measure and internally log several water-quality parameters. Starting in 1988, the data were collected most often at a 15- or 20-minute logging interval. The multiparameter instruments were located along the banks of the Colorado River (fig. 2) and were cleaned and calibrated on a 1- to 6-month interval following maintenance procedures suggested by Wagner and others (2006).

At two sites in the study area, the Colorado River at Lees Ferry (CRLF) and the Colorado River near river mile[2] 226 (CR226, fig. 1), specific-conductance data from multi-

[2] By convention, river mile is used to measure distances along the Colorado River in Grand Canyon.

Figure 2. The multiparameter instrument at the Colorado River near river mile 61 has been removed for maintenance from the river and is visible in the lower left corner of the photograph.

parameter instruments were compared with TDS concentrations analyzed from samples collected at the sites. The CRLF site is located near the upstream end of the study area, approximately 15 river miles downstream from Glen Canyon Dam (fig. 1). The CR226 site is the furthest downstream site in the study area, located approximately 241 river miles below Glen Canyon Dam (fig. 1). The relation between specific conductance and TDS is dependent on the total and relative amounts of dissolved minerals in the water (American Public Health Association, 1992). Total dissolved solids can be estimated by multiplying specific conductance by a constant, which typically ranges from 0.55 to 0.9 (American Public Health Association, 1992). This constant was calculated at the CRLF and CR226 sites, and the resulting regression through the origin (RTO) at each site was compared with the simple linear regression model (ordinary least-squares, OLS). The RTO and OLS models were compared by evaluating the p-value of the y-intercept and by comparing the standard errors of the RTO and OLS regressions (Eisenhauer, 2003).

Results

Specific Conductance and Salinity

The specific-conductance data that were modeled with TDS ranged from 629 to 978 µS/cm at CRLF and 810 to 1,008 µS/cm at CR226 (fig. 3). The TDS data ranged from 411 to 642 milligrams per liter (mg/L) at CRLF and 527 to 656 mg/L at CR226 (fig. 3). Based on criteria outlined by Eisenhauer (2003), the RTO model was determined to fit the data as well as the OLS model at both sites. The RTO model, which also makes more sense physically (a value of 0 specific conductance should predict a value of 0 TDS), was thus chosen to represent the data. At the two sites, the RTO model yielded nearly identical slopes, 0.653 at CRLF and 0.650 at CR226 (fig. 3). R-squared values reported for RTO models are often inconsistent and ambiguous (Eisenhauer, 2003; Hocking, 1996). A measure analogous to R-squared that is applicable to the RTO model is the square of the sample correlation between observed and predicted values (Hocking, 1996). This statistic was calculated as 0.94 for the RTO at CRLF and 0.82 for the RTO at CR226.

The Little Colorado River is the only tributary in the study area that, at base flow, significantly alters the salinity of the Colorado River. At base flow the Little Colorado River increases the salinity of the Colorado River by approximately 5 to 15 percent. Despite this input of salts from the Little Colorado River, the relation between total dissolved solids and specific conductance does not change significantly downstream from the confluence; the coefficient for the RTO model was determined to be 0.653 at CRLF upstream from the confluence and 0.650 at CR226 downstream from the confluence. In the entire study area, TDS can be estimated from specific conductance by using the following formula:

$$\text{total dissolved solids (mg/L)} = 0.65 * \text{specific conductance } (\mu\text{S/cm})$$

Specific Conductance as a Natural Tracer

In the approximately 280-mile-long reach of the Colorado River between Glen Canyon Dam and Lake Mead, there are a number of large tributaries that contribute water to the Colorado River (fig. 1). During certain flow conditions,

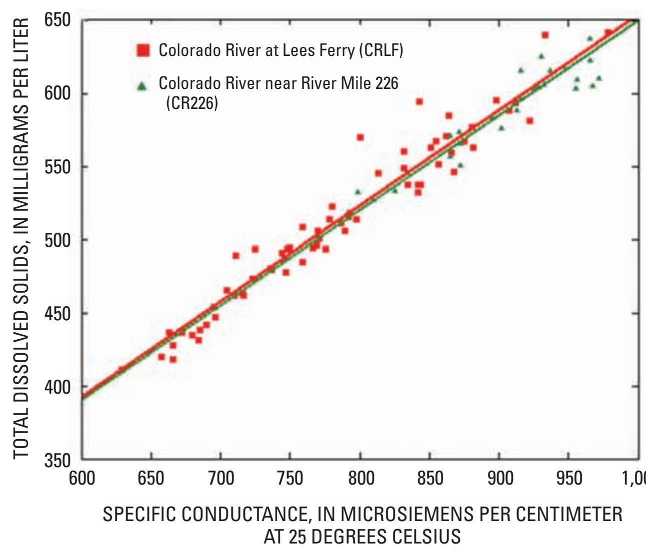


Figure 3. Relation between total dissolved solids and specific conductance of the Colorado River at Lees Ferry (CRLF) from 1991 to 2006 and of the Colorado River near river mile 226 (CR226) from 2002 to 2006. (Refer to figure 1 for the location of the two stations.)

some of these tributaries contain water with much different specific conductance than the Colorado River. In these cases, the specific conductance of the tributary water can be traced downstream after it enters the Colorado River. One such situation occurred in January 2005 when the Paria River was flooding and released a pulse of high-specific-conductance water (approximately 1,900 µS/cm) into the lower-specific-conductance Colorado River water (approximately 900 µS/cm). The result was a high spike in specific conductance in the Colorado River downstream from the Lees Ferry site (where the Paria River enters the Colorado River), which was measured by the multiparameter instruments at four monitoring stations as the spike moved downstream in the Colorado River (fig. 4). The average discharge of the Colorado River in the study area during this time period was approximately 480 cubic meters per second (m^3/s; 17,000 cubic feet per second (ft^3/s)). The travel time of the water, determined by tracking the conductivity spike, was approximately 83 hours from CR030 to CR226 (fig. 1), with an average velocity of 1.06 meters per second (m/s), or 2.36 miles per hour (mph). This water velocity is comparable to results obtained from dye studies in this reach of the river at a similar discharge (Graf 1995, 1997).

A second example of specific conductance from a tributary input that can be traced downstream in the Colorado River occurs when the Little Colorado River (fig. 1) is at base flow (approximately 6.2 m^3/s, or 220 ft^3/s) and the Colorado River has daily fluctuations in discharge (resulting from hydropower generation at Glen Canyon Dam). In June 2005, the specific conductance of the Colorado River was fairly stable (approximately 850 µS/cm) previous to input from the Little Colorado River (fig. 5A). When the higher-specific-conductance water of the Little Colorado River (approximately 4,500 µS/cm) joined the Colorado River,

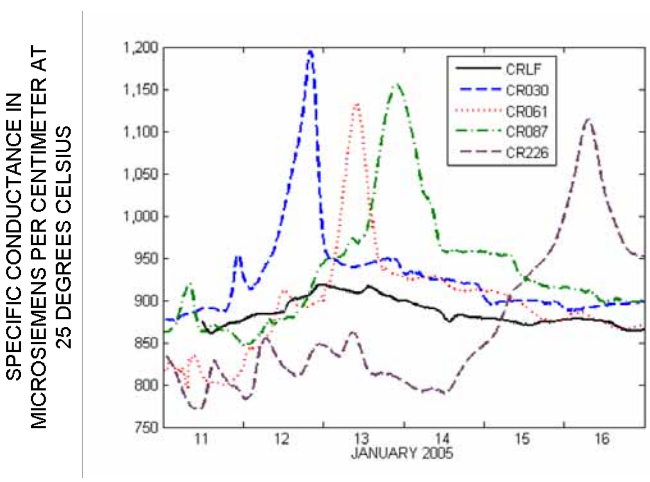


Figure 4. Specific conductance at five of the monitoring stations on the Colorado River from January 11 to 16, 2005. (Refer to figure 1 for the station locations.) The specific-conductance spike at the four stations on the Colorado River downstream from the Paria River is the result of a large Paria River flood.

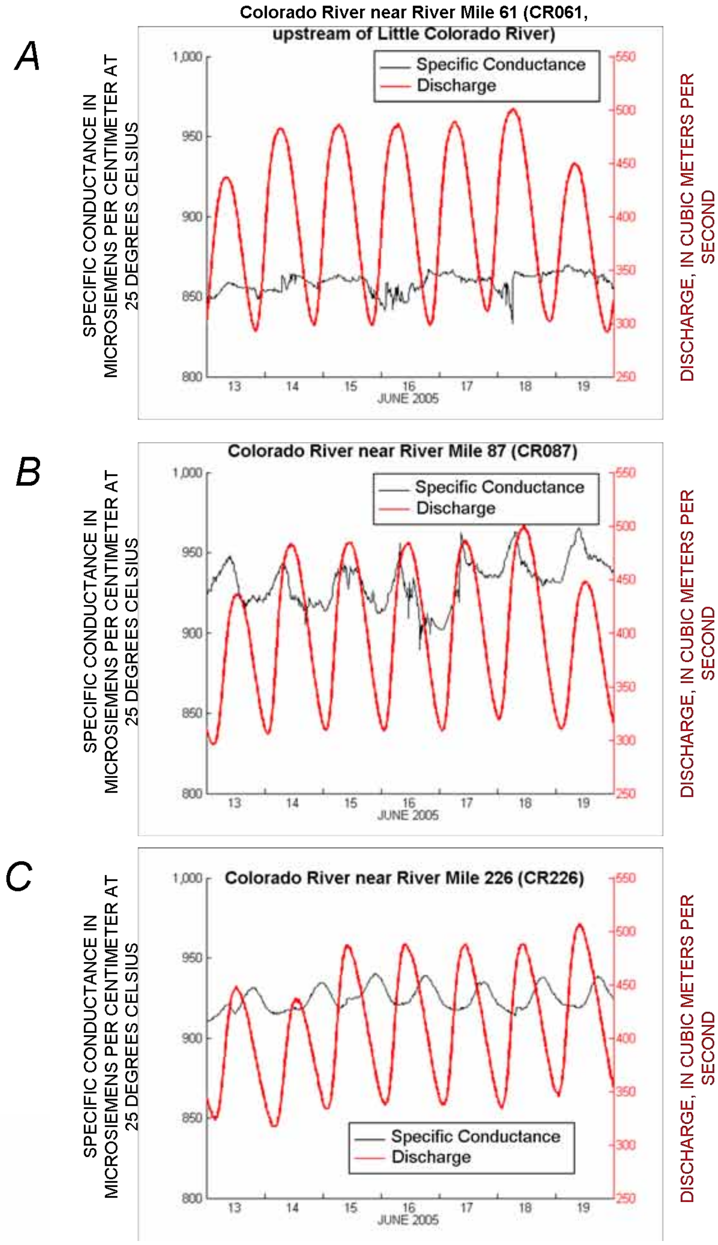


Figure 5. Specific conductance and discharge at three monitoring stations on the Colorado River from June 13 to 19, 2005. (Refer to figure 1 for the station locations.) The Little Colorado River was at base flow, contributing high-specific-conductance water to the Colorado River.

the specific conductance of the Colorado River increased and developed regular peaks. These specific-conductance peaks, which can be tracked downstream (fig. 5B and C), were formed at the confluence of the Colorado River and the Little Colorado River during daily periods of low Colorado River discharge.

Daily fluctuations in the water released from Glen Canyon Dam cause discharge waves to develop in the study area (fig. 5), which travel at a faster speed than the actual water (Lighthill and Whitman, 1955). This difference in speed is evident in figure 5B and C; the specific-conductance peaks, which travel with the actual water, were in different positions relative to the discharge waves at stations CR087 and CR226. The water traveled from CR087 to CR226 in approximately 56.5 hours (1.10 m/s, 2.46 mph) whereas the discharge wave took only approximately 24 hours to travel between the two stations (2.59 m/s, 5.79 mph). The discharge wave velocity was measured by tracking changes in downstream river elevation; the movement of the actual water is more complicated and must be measured using a tracer, which in this case was specific conductance.

Implications for Management

The U.S. Geological Survey's Grand Canyon Monitoring and Research Center has an extensive specific-conductance dataset and continues to monitor specific conductance on a 15- or 20-minute interval from six sites in the study area (fig. 1). These specific-conductance data can be used to estimate the salinity of the Colorado River in the study area by applying a simple linear regression: total dissolved solids (mg/L) = 0.65 * specific conductance (µS/cm). Water travel time of the Colorado River, important for sediment-transport and biological studies, can also be calculated by using the cost-effective and noninvasive method of tracking specific-conductance signals as they travel downstream in the study area.

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Mapping Full-Channel Geometry in Grand Canyon by Using Airborne Bathymetric Lidar: The Lees Ferry Test Case

By Philip A. Davis[1] and Theodore S. Melis[2]

Abstract

In November 2004, we performed one of the first river tests of a new, dual-beam light detection and ranging (lidar) system (Scanning Hydrographic Operational Airborne Lidar Survey or SHOALS) that was designed to simultaneously map topography and bathymetry in coastal areas. This test was performed to determine whether SHOALS is a more noninvasive, alternative method for mapping full-channel geometry of the Colorado River and is useful for sediment and ecosystem modeling. The system was tested at the Lees Ferry reach—a clear-water, "best-case" scenario for SHOALS. Acoustic multibeam surveys were conducted to provide "ground truth" to determine the vertical accuracy and mapping depth of SHOALS. Vertical accuracies of SHOALS bathymetry and topography were the same and very similar to moderate-resolution, airborne topographic lidar systems (33 cm $RMSE_{95}$). Maximum depth obtained by SHOALS was 17.6 meters; the multibeam surveys indicated a maximum reach depth of 24.2 meters. Compared to combined multibeam and land surveys, the SHOALS survey is less invasive, more rapid, and comparable in cost, and SHOALS can map the entire 450-kilometer river corridor in a week, which could not be accomplished in a year by ground surveys. However, SHOALS provides lower point spacing (less surface detail), probably lower vertical accuracies, and less deep-water coverage than multibeam and land surveys.

Introduction

The Grand Canyon Monitoring and Research Center (GCMRC) of the U.S. Geological Survey develops protocols for the release of water from the Glen Canyon Dam in Arizona to determine flow conditions that maintain, and hopefully restore, the sediment resources within Grand Canyon. The terrestrial sediment deposits serve as critical habitats for wildlife and as campsites for the general public. Although terrestrial sediment storage is a focal point, much of the sediment that enters the Colorado River system in Grand Canyon resides within the river's mainstem, which can either be periodically forced onto the river banks with constructive high flows or be continually moved downstream to Lake Mead, which is the general fate of much of the fine-grained sediment (Topping and others, 2000). In order to accurately model the sediment budget and its response to different flow protocols, as well as model the integrated ecosystem response (Korman and others, 2004), it is important to know the complete channel geometry below the flow-stage elevation that is being considered or tested. Currently, the channel geometry is determined at a particular river reach by using a combination of two methods: (1) land surveys that extend into the water a few meters during low-steady flow periods and (2) acoustic-multibeam, watercraft surveys during higher flow regimes so that the two surveys overlap. Although the boat and land surveys are one of the more accurate surveying methods, they are also time-intensive, expensive, and considered invasive.

At the end of the GCMRC remote sensing initiative, conducted from 2000 to 2003, we learned of an airborne bathymetric mapping system that was developed by the Army Corps of Engineers for the Navy (Irish and Lillycrop, 1999; Guenther and others, 2000; Irish and others, 2000; Wozencraft and Lillycrop, 2003) and also manufactured for commercial use. The commercial system is known as the SHOALS (Scanning Hydrographic Operational Airborne Lidar Survey) system, where lidar stands for light detection and ranging. SHOALS is a 1 kilohertz (kHz), dual-laser ranging system that employs a green-wavelength (520 nanometer [nm]) laser to detect the channel substrate elevation and a near-infrared-wavelength (1,064 nm) laser to detect the water-surface and land elevations. Bathymetry is determined from the difference in travel times of the pulses from the two laser systems. Although the system was designed for coastal bathymetric mapping in areas where (or times when) waters are relatively

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clear, we thought it might have application to channel mapping within Grand Canyon, at least within reaches having permissive water conditions. Theoretically, SHOALS could map down to a depth near 50 meters (m), but absorption of light by chlorophyll and yellow substance and strong scattering of light by particles in the water (turbidity) decrease the laser's penetration depth (fig. 1). Before the fall of 2004, no one had used the SHOALS system on a river to determine its real ability for river systems. The potential of SHOALS to provide more rapid, more extensive coverage (full channel geometry) of the river system in a less-invasive manner prompted us to perform a practical test of its capability to better understand the system's cost efficiency, accuracy, and limitations for river environments.

Data Collection and Analysis

We selected two sites for our test of the airborne bathymetric mapping system: a 6.4-kilometer (km) segment of the San Juan River (37 km from its confluence with Lake Powell) and a 4-km segment of the Colorado River just north of Lees Ferry (the southern terminus of Glen Canyon). These two sites represent end members of potential river turbidity with the Lees Ferry reach consistently being the least turbid because its only water source is the dam, which provides very little sediment to Glen Canyon. The study was conducted in late November of 2004 just after a major winter storm that input large amounts of sediment into the basin's tributaries. As a result, the San Juan River was so turbid that its water was a dense, chocolate-brown color. We, therefore, eliminated this test site from consideration and concentrated on the Lees Ferry reach, which is shown in figure 2.

Airborne Bathymetric Lidar Collection

Fugro Pelagos (San Diego, CA) leased the SHOALS 1000T bathymetric lidar system from Optec Corporation and fitted the system in a Bell 206 L-III Ranger helicopter. A helicopter was employed in order to fly at low altitude (300 m) and low speed (65 knots) to obtain a 3-m point spacing within any particular flight line. To obtain a final lidar point spacing near 1 m (the cell resolution used for digital elevation models (DEMs) to conduct modeling and change-detection analyses), we collected seven flight lines that overlapped by 50 percent. At a 300-m altitude, the lidar system collected data over a swath width of 160 m. The total SHOALS collection area is shown in figure 2. Examination of the flight-line point data showed that a 1.1-m point spacing was achieved with three overlapping flight lines, and a 0.9-m point spacing was obtained with four overlapping flight lines. Four flight lines for this 4-km reach were acquired in less than 20 minutes.

The helicopter was equipped with an Applanix POS AV 410 Global Positioning System (GPS) system and an Inertial Measurement Unit (IMU) that tracks the aircraft position and beam pointing. Three dual-frequency GPS base stations were operated at a 1-second recording interval during the overflight. These L1/L2 base stations were within 12 km of the study area; two stations were within 2.4 km. Two stations were used in the kinematic GPS solutions, and the third station was used to verify the solutions. The lidar data were then processed to derive an ellipsoid height and position for each pulse. Positional data were delivered in our standard map coordinate system (State Plane, central Arizona-Zone 202, North American Datum of 1983 (NAD83)). The standard SHOALS system is also equipped with a DuncanTech DT4000 digital camera that acquires natural-color imagery during the lidar collection.

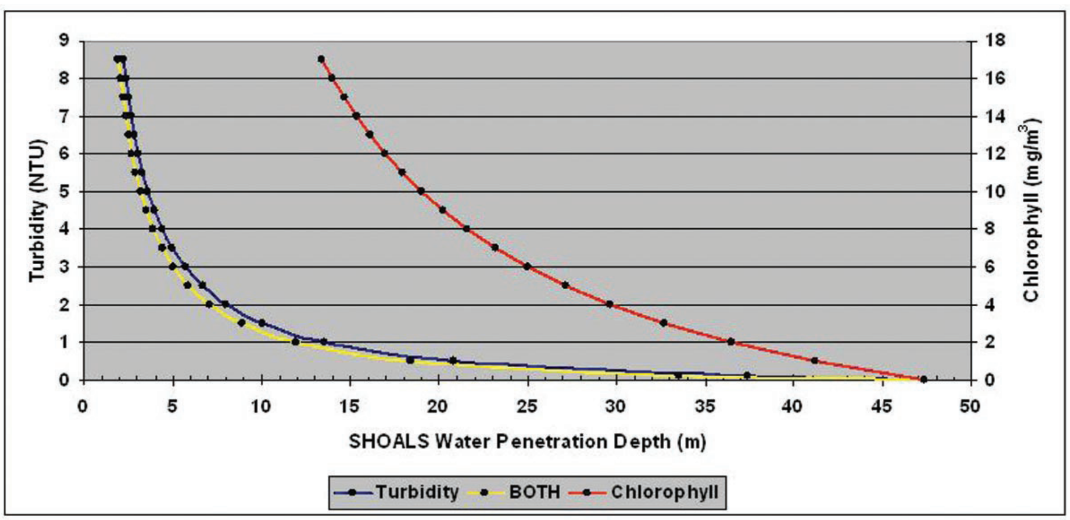

Figure 1. Theoretical water-penetration depth of SHOALS green-wavelength laser as a function of turbidity and chlorophyll concentrations, based on integrated absorption/scattering equations and reported parameter values (Gallegos, 1994, and references therein). NTU is nephelometric turbidity unit; BOTH shows the combined effect of turbidity and chlorophyll.

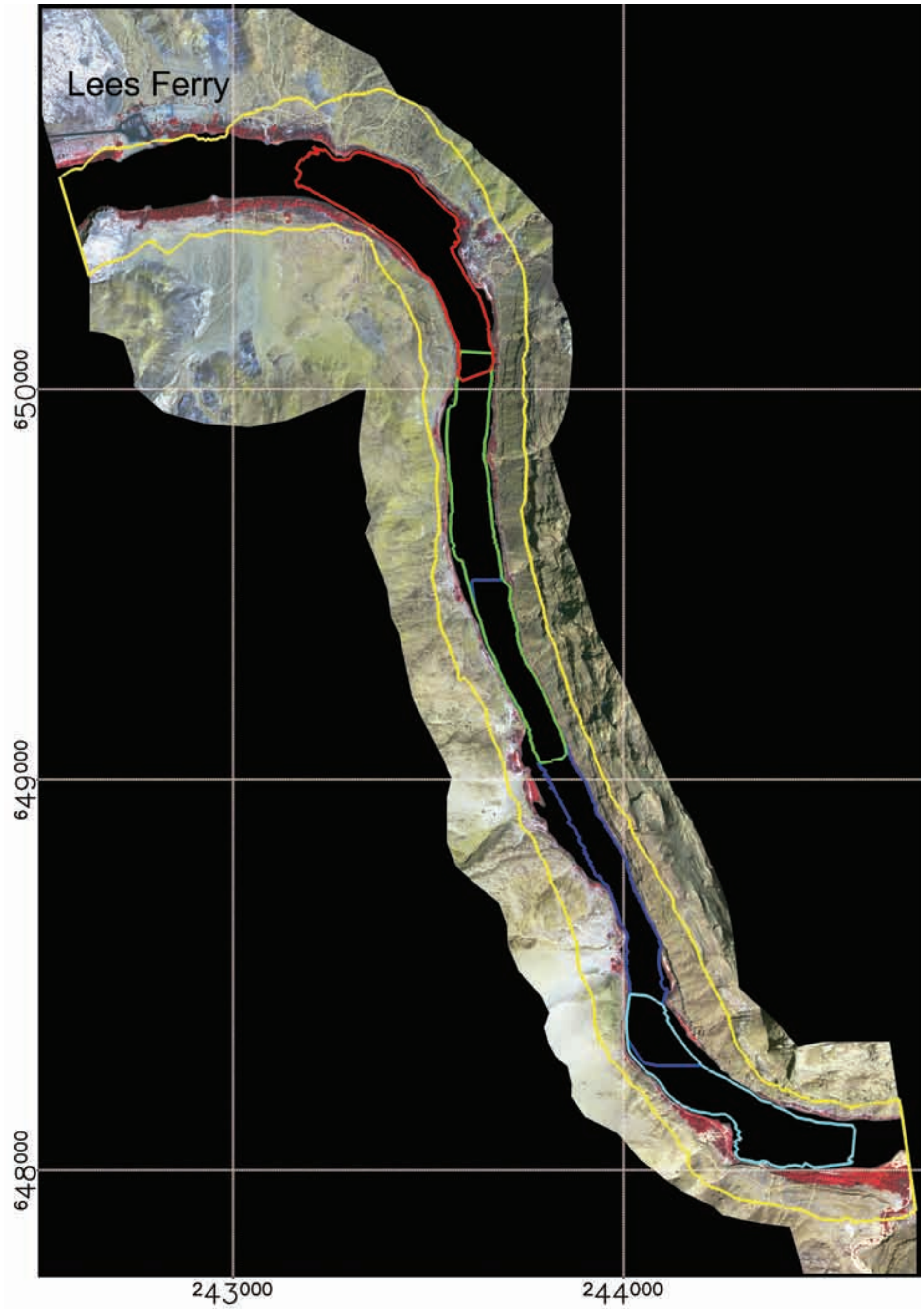

Figure 2. Color-infrared image of the Lees Ferry study area showing the SHOALS data-collection area (yellow polygon), the real-time kinematic (RTK) multibeam collection areas (red, green, and cyan polygons), and the OmniSTAR multibeam collection area (blue polygon). Image is in State Plane (Zone 202) map projection.

We wanted to use the image data of the channel to determine sources of potential error in bathymetric values (e.g., aquatic vegetation, cobble areas), but the digital image data were not properly stored during flight and no useful images were obtained. This problem has now been corrected to provide high-gain, channel imagery.

Acoustic Multibeam Bathymetric Data Collection

During the SHOALS overflight in November, the acoustic multibeam system was preoccupied with surveys downstream in response to the early November high-flow experiment; therefore, we were not able to obtain acoustic bathymetry during the overflight. In early May 2005, we performed detailed acoustic multibeam surveys of the Lees Ferry reach. Even though this was 5 months after the overflight, we felt the channel had not changed very much (if at all) because dam releases contain almost no sediment and Glen Canyon is substantially depleted of sand (Grams and others, 2007). The Lees Ferry study area was surveyed in “pools;” pool locations and extents depended on the existence of line-of-sight base stations along the shoreline. Three pools were surveyed using acoustic multibeam coupled to real-time kinematic (RTK) base-station tracking (fig. 2); two L1/L2 base stations were employed for each pool’s survey. Base station occupations used the established primary control for Grand Canyon, one of which was also occupied during the SHOALS data collection and used to process its data. A fourth, intervening pool was surveyed with acoustic multibeam by using an OmniSTAR navigation system because of the absence of line-of-site L1/L2 base stations for a small portion of the channel (fig. 2). OmniSTAR relies solely on GPS satellite positioning and is not as accurate as ground RTK positioning. Therefore, the bathymetry derived from the OmniSTAR survey was not seriously considered in our SHOALS analyses. The multibeam surveys collected data at a 25-centimeter (cm) point spacing, significantly higher than SHOALS. Along the shoreline, where depths are less than 1 m, the acoustic transducer (which extends 1 m beneath the boat) was tilted toward the shore in an attempt to derive bathymetric data in the very shallow areas. This was not always successful because of the rocky substrate and, therefore, we obtained very little reliable data at depths less than 1 m.

It is commonly reported that SHOALS can obtain accurate depth measurements down to 2–3 Secchi depths (Guenther and others, 2000). Thus, we measured the Secchi depths at seven locations within the study area and found the values to be 7.3 ± 0.6 m. This suggests that the maximum mapping depth of SHOALS within the study area is 14.5–21.8 m. Turbidity measurements during 2004 at the Lees Ferry streamflow-gaging station (800 m downstream from Lees Ferry, but upstream from the Paria River confluence) recorded a high value of 1.3 nephelometric turbidity unit (NTU) in April 2004, but all other measurements during 2004, including the last measurement in September, were close to 0.5 NTU (Fisk and others, 2005). Based on theoretical considerations (fig. 1), the maximum SHOALS mapping depth would have been 20.9 m, if turbidity was 0.5 NTU as measured in September 2004, which is similar to the maximum depth suggested by the measured Secchi depth.

Comparative Analyses and Results

We combined the RTK multibeam bathymetric point data into a single point file and produced a DEM with a 1-m cell dimension. Areas outside the extent of the original point file were excluded. The same procedure was used to create a 1-m DEM from the OmniSTAR multibeam data. Before combining the SHOALS lidar point data from the various flight lines, we performed a point-to-point comparison of the ellipsoid heights between all possible pairs of flight lines to determine possible vertical offsets between flight lines, which are quite common in lidar data (Sallenger and others, 2003; Hilldale and Raff, 2008). The point comparisons were performed on bare land and channel substrates with slopes less than 11 degrees (°), using points between a particular pair of flight lines that were within a 25-cm radius. Interflight-line vertical offsets ranged from +11 cm to –7 cm (five of the seven flight lines had offsets within ±3 cm) with no obvious differences between land and water. These relative offsets were applied to their respective flight lines to make the lidar data more internally consistent. The combined lidar dataset was then similarly compared to land and water control points to determine possible absolute vertical offsets. This comparison showed the combined lidar dataset to be 30 cm lower than the ground control; the lidar dataset was, therefore, adjusted upward by that amount.

Data gaps occurred within the multibeam and lidar datasets because of inherent limitations of each survey system. The data gaps occurred in both the multibeam and SHOALS data in the shallow areas along the shoreline, but SHOALS presented fewer shallow data gaps than the multibeam data (fig. 3). The multibeam shallow-water data gaps are caused by the inability of the survey boat to enter shallow-water areas because the acoustic transducer extends 1 m beneath the boat. The SHOALS shallow-water data gaps are because of overlapping errors in the green (substrate) and near-infrared (water surface) laser returns at depths less than 30–50 cm. The SHOALS data also have gaps within the deepest portion of the channel (fig. 4), where the green-laser pulse was attenuated to the point that there was no distinct reflection from the substrate. This occurred at a depth of 17.6 m, based on our collected multibeam data at the deepest SHOALS laser returns. The maximum depth recorded by the multibeam survey for that deep pool (fig. 4) was 24.2 m. Assuming the turbidity was 0.5 NTU in November 2004 (as last measured at the Lees Ferry stream gage in 2004 during September), SHOALS should have theoretically been able to acquire valid data at a depth near 21 m, but if the water’s chlorophyll content was just 1 milligram per cubic meter (mg/m^3) or the turbidity was slightly higher in November (i.e., 0.75 NTU), then the theoretical depth limit for a green-wavelength laser reflection (depicted in figure 1) would be close to that achieved by our SHOALS survey.

We measured the vertical accuracy of the SHOALS bathymetry by comparing its 1-m DEM data to that derived from the RTK multibeam data. This assessment was conducted at 1-m depth intervals in order to determine consistency

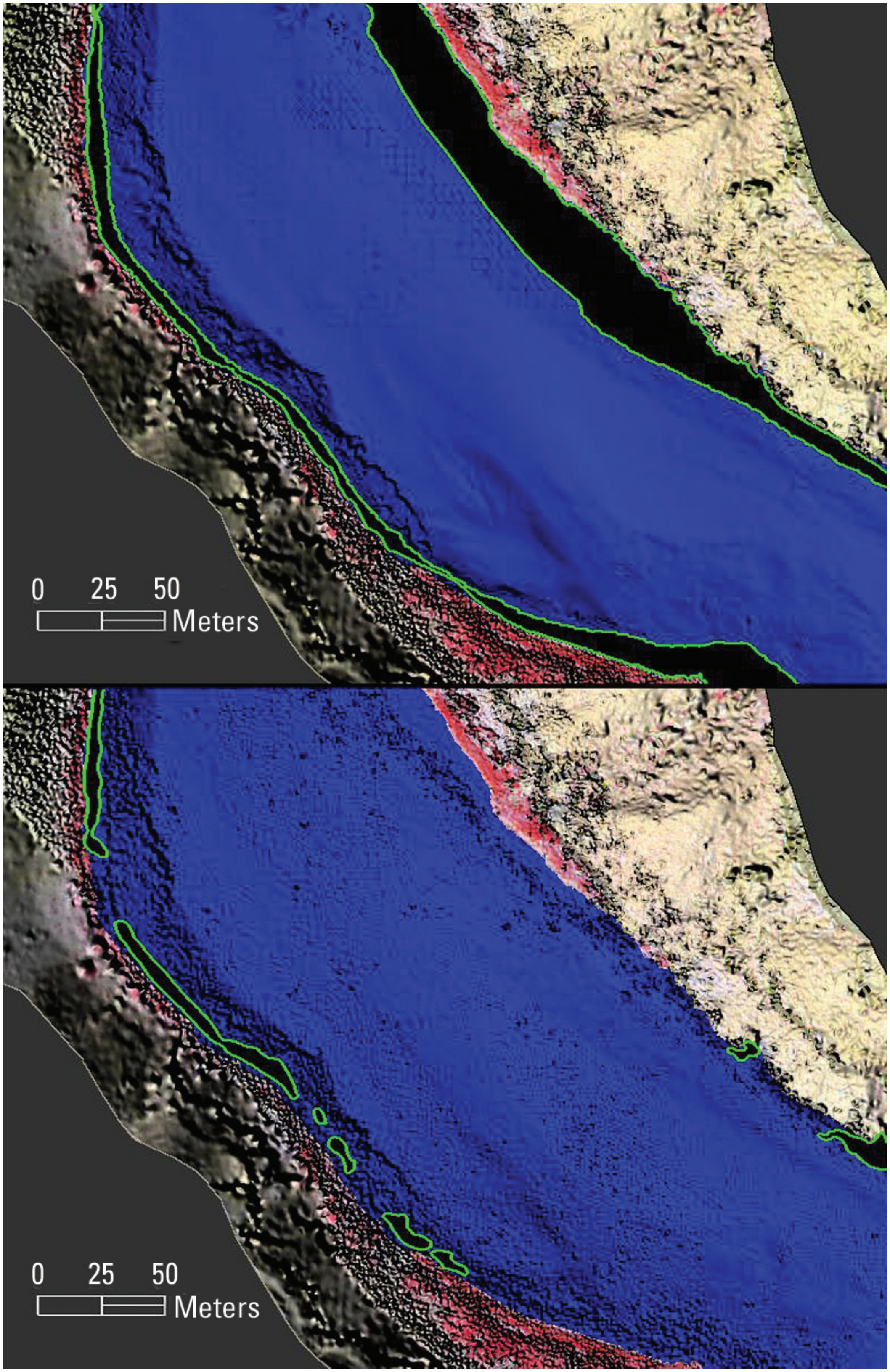

Figure 3. Shaded-relief DEM image of a portion of the Lees Ferry study area showing survey limitations of multibeam (top) and SHOALS (bottom) within shallow-water (<1 m depth) areas. Water is represented as blue, superposed on a shaded-relief, color-infrared image of the study area. Green polygons outline data gaps.

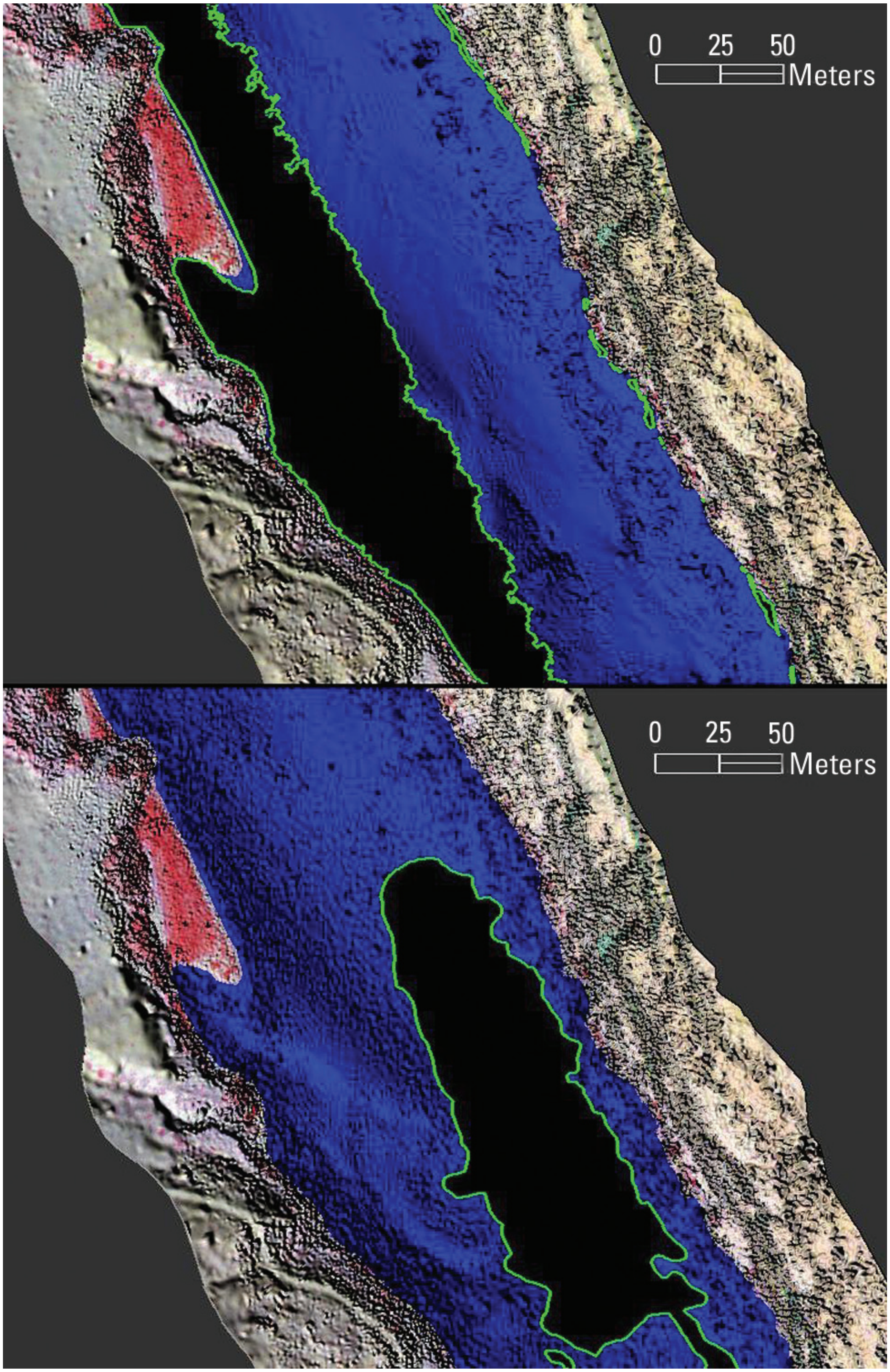

Figure 4. Shaded-relief DEM image of a portion of the Lees Ferry study area showing shallow- and deep-water limitation of SHOALS (bottom) relative to multibeam surveys (top). Water is represented as blue, superposed on a shaded-relief, color-infrared image of the study area. Green polygons outline data gaps.

and limitations of these data at various depths. We initially examined multibeam DEM cells that had slopes less than 11° (≤20 percent grade) and report vertical accuracy using RMSE at the 95-percent confidence level, according to lidar evaluation guidelines for fundamental vertical accuracy that were established by the American Society for Photogrammetry and Remote Sensing (American Society for Photogrammetry and Remote Sensing, 2004). We assessed the vertical accuracy of the SHOALS terrestrial topographic data (obtained using the near-infrared laser returns) by comparing its measured ellipsoid heights with those of 18 ground control-panel locations within the study area. The results of the topographic and bathymetric assessments are listed in table 1.

The vertical accuracy of the terrestrial lidar topographic data is similar to accuracies we have obtained from higher altitude, terrestrial lidar surveys in Grand Canyon (reviewed in Davis, 2004). The very low accuracy at depths less than 1 m (table 1) could be because of multibeam error; there were very few multibeam DEM cells at that depth for comparison, and shallow-water ground surveys were not performed. The vertical accuracies throughout much of the water column are better than the 50 cm ($RMSE_{95}$) that is generally stated by Fugro Pelagos and the Army Corps of Engineers for SHOALS coastal and estuarine surveys. Our higher measured accuracies may be because of the very close proximity (≤12.5 km) of the GPS base stations and slow aircraft collection (65 knots) during our survey relative to the average baseline distances and aircraft speeds used for coastal/estuarine surveys. Although our terrestrial accuracy assessment used stable, well-established photogrammetric control, our bathymetric accuracy assessment is based on two fundamental assumptions. First, the channel substrate had not changed during the 5-month interval between the SHOALS and multibeam surveys. Second, the multibeam data are "truth," but the accuracy of the multibeam surveys within Grand Canyon has not yet been determined, and therefore, our measured accuracies within the channel should be considered relative accuracies.

Only one published study has been done to evaluate SHOALS performance relative to ground-truth data on a river system, and that study was based on 2004–2005 surveys of the Yakima (southern Washington) and Trinity (northern California) Rivers (Hilldale and Raff, 2008). They reported mean absolute elevation errors (MAE) for different river reaches, instead of RMSE values. Their MAE values for different river reaches were in the range of 10–20 cm, similar to the MAE values we obtained and present in table 1 for comparison purposes. Although Hilldale and Raff (2008) did not report turbidity, their SHOALS bathymetric surveys had no problem mapping down to the 6-m depths of the Trinity and Yakima Rivers.

Previous studies of lidar data acquired over land have noted a linear increase in MAE with increasing surface slope because of positional error, such that MAE on 20–30° slope was twice that on relatively flat surfaces (Hodgson and others, 2003; Hodgson and Bresnahan, 2004; Peng and Shih, 2006). A similar relation was also observed in the SHOALS bathymetric study by Hilldale and Raff (2008). Our examination of vertical accuracy ($RMSE_{95}$) of SHOALS bathymetry relative to channel slope showed a strong ($R^2 = 0.98$) linear relation [$RMSE_{95}$ (cm) $= 3.7 \cdot slope_{degrees} - 8.8$ cm]. We had too little topographic ground-truth data to replicate this analysis for the SHOALS topography.

Our analysis of the precision of corrected SHOALS flight-line point data showed a decrease in precision with increasing land and channel slope (fig. 5). This relation on land was strongly ($R^2 = 0.99$) linear [$RMSE_{95}$ (cm) $= 4.0 \cdot slope_{degrees} - 3.3$ cm]. Although the bathymetric precision measurements plot near the topographic regression line, the decrease in bathymetric precision with increasing channel

Table 1. Fundamental vertical accuracy of SHOALS lidar on land and as a function of water depth.

Water depth (m)	$RMSE_{95}$* (cm)	MAE* (cm)	Number of cells compared
< 0 (Land)	33	14	18
0.6–1	98	38	29
1–2	45	17	2,000
2–3	35	14	15,701
3–4	39	15	15,518
4–5	37	13	16,862
5–6	35	13	13,231
6–7	31	12	14,667
7–8	35	14	17,577
8–9	33	12	15,674
9–10	33	13	20,525
10–11	33	11	10,686
11–12	29	13	6,634
12–13	35	13	2,666
13–14	39	15	2,119
14–15	41	17	1,631
15–16	55	23	904
16–17.6	55	23	18

* $RMSE_{95}$ is root-mean-square error at the 95-percent confidence level; MAE is mean absolute error.

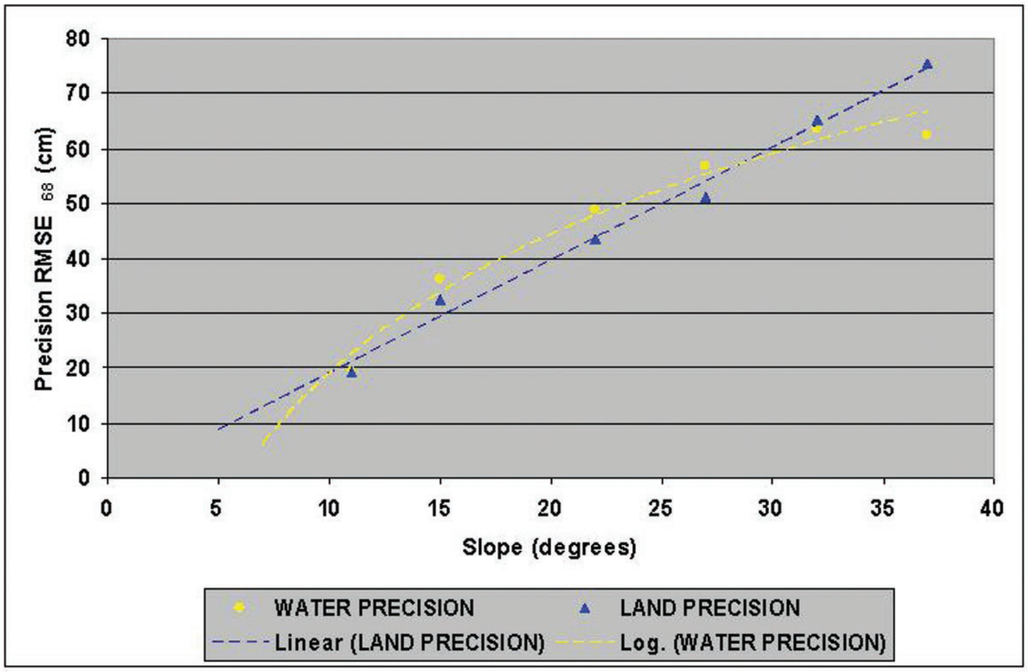

Figure 5. Variation in precision of SHOALS topography (land) and bathymetry (water) relative to surface slope. Dashed blue line represents linear regression of topographic points; dashed yellow line represents logarithmic regression of bathymetric points.

slope is closer ($R^2 = 0.97$) to a logarithmic relation [$RMSE_{95}$ (cm) = 71.5 • ln ($slope_{degrees}$) – 127.3 cm] (fig. 5), similar to optical attenuation in fluid media. For slopes ≤11°, the vertical accuracy and precision on land and within the channel were very similar; 70 percent of the channel has such low slopes. Our analysis of SHOALS bathymetric precision ($RMSE_{95}$) relative to water depth showed two distinct stratifications of error: one at depths less than 9 m (47–59 cm) and the other at depths greater than 9 m (39–45 cm).

Implications for Management

Ground-based and airborne monitoring methods have their own sets of advantages and disadvantages. Program managers for wilderness areas need to consider such factors as areal extent, time, cost, invasiveness, and accuracy of different approaches for a particular monitoring task. This paper examined an alternative airborne approach (SHOALS) to ground-based surveys for monitoring full-channel geometry within Grand Canyon, so let us objectively compare the two approaches, based on a 50-km river reach. *Time*—Ground-based surveys would require about 21 days to map the topography and bathymetry; SHOALS survey would require 4 hours. *Cost*—Ground-based surveys would cost a minimum of $50,000, plus months of data processing; SHOALS survey would cost $149,000 with little post-processing. *Areal Extent*—If the full-channel geometry were required for the entire river corridor, ground-based surveys would require 185 days to accomplish this task (with collection costs approximately $450,000, plus a year of data processing); SHOALS could complete such a survey in 6 days for $400,000 (i.e., there is an economy in scale). *Invasiveness*—Ground-based surveys are invasive; SHOALS would produce minor rotor noise at a 300-m altitude and no ground intrusion. *Accuracy*—Ground-based surveys are very accurate on land; SHOALS surveys cannot compete with the vertical accuracy of land surveys and are not adequate for detailed monitoring of terrestrial sediment storage and transport at the 25-cm level. It is difficult to comment on the bathymetric accuracies because we do not know the true accuracy of multibeam. *Surface Detail*—Ground-based and SHOALS topographic surveys are comparable in their areal point density, but ground-based bathymetric surveys are far more detailed than SHOALS surveys, as demonstrated by a 0.5-m DEM comparison where sand waveforms are very distinct in multibeam data but are not apparent in SHOALS data (fig. 6). However, multibeam data are used mostly at the 1-m cell resolution, at which point SHOALS 1-m data look similar to multibeam data. *Bathymetric Data Gaps*—Ground-based surveys will have data gaps in rapids and along portions of the shoreline; SHOALS surveys will acquire more of the shoreline, will probably have data gaps in rapids because of entrained air and in the deep (>18 m) portions of the channel, and may also have large data gaps at shallower depths because of turbidity introduced into the mainstem by tributaries. The later limitation might be mitigated by careful timing of the SHOALS data collection.

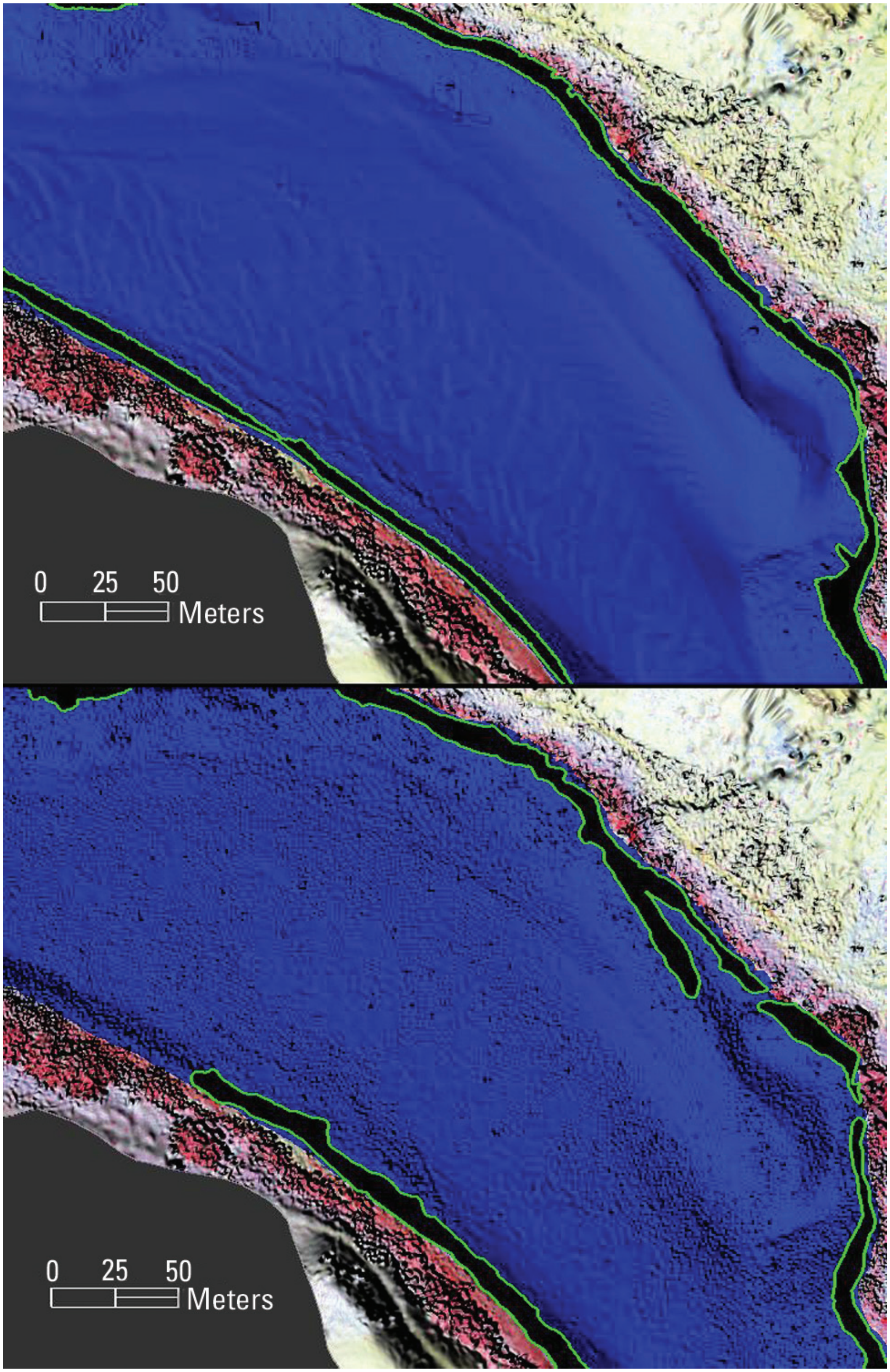

Figure 6. Shaded-relief DEM image (at 0.5-m cell resolution) of a portion of the Lees Ferry study area showing the greater substrate detail provided by 0.25-m multibeam data (top) relative to that provided by 1-m SHOALS data (bottom). Water is represented as blue, superposed on a shaded-relief, color-infrared image of the study area. Green polygons outline data gaps.

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