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.

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.

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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

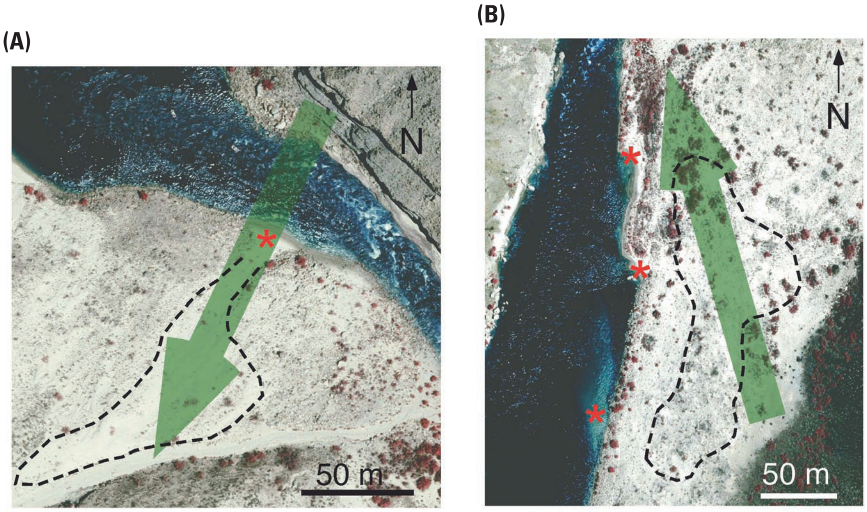


Figure 1. (A) Example of a modern fluvial sourced (MFS) aeolian dune field in Grand Canyon. The dune field (within dashed boundary) is directly downwind from a sandbar formed by flows at or below 45,000 cubic feet per second (asterisk). Here, the dominant wind direction is from the northeast (green arrow), 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

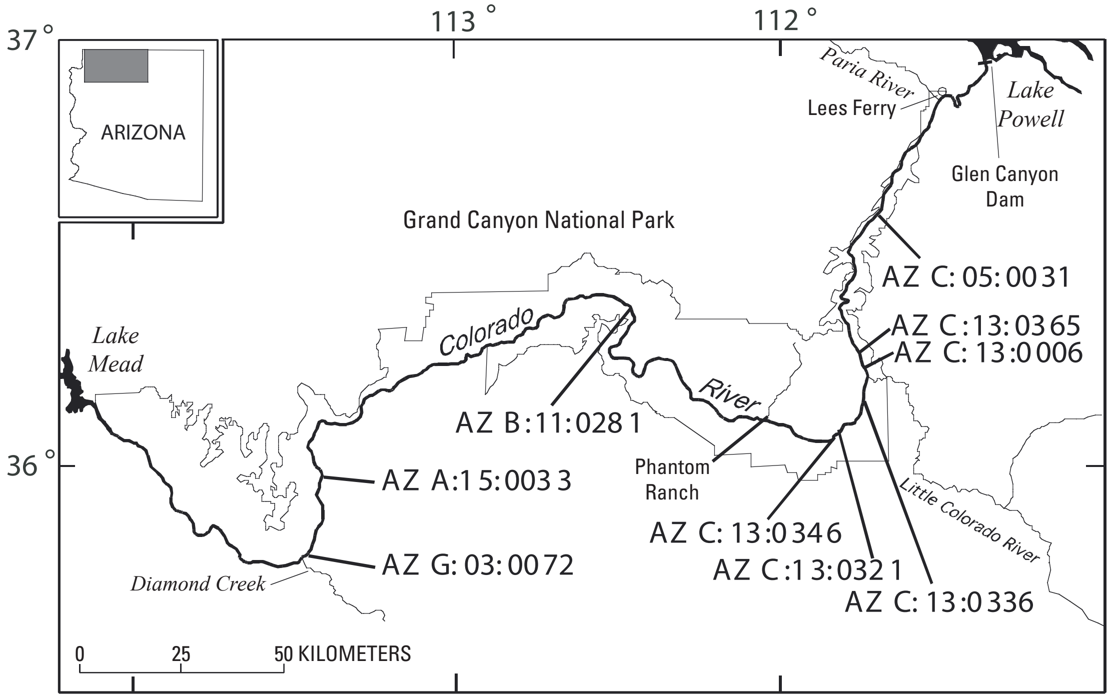
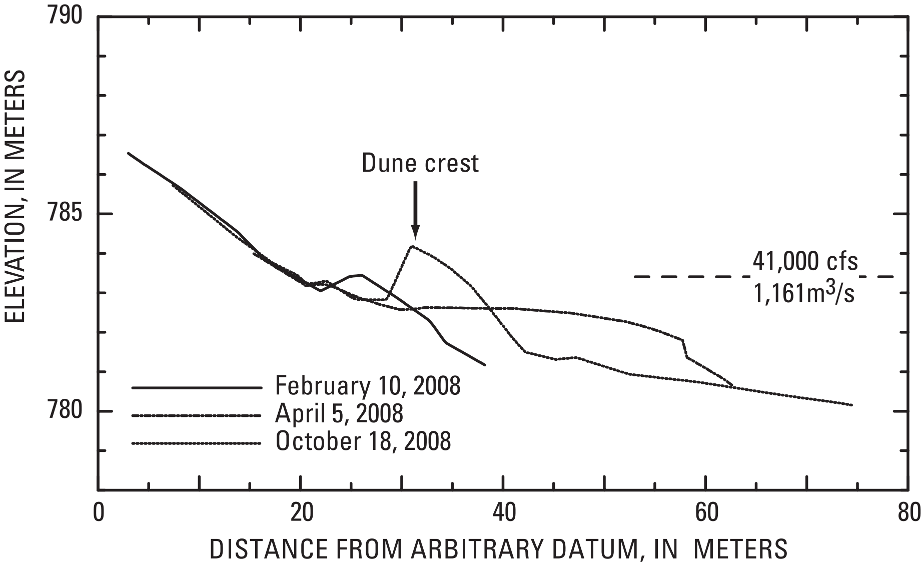


Figure 2. Sites where aeolian sand transport is monitored in the Colorado River corridor, Grand Canyon, Arizona. Site numbers refer to archaeological sites near weather stations and sand traps that measure weather conditions and rates of windblown sand flux in dune fields.

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.

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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

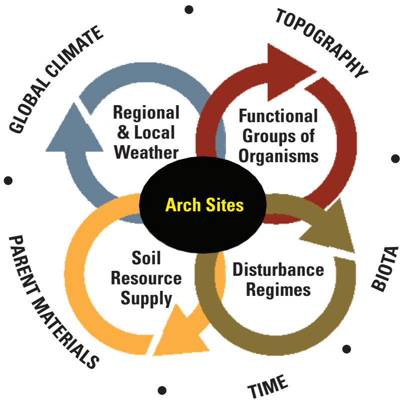

Figure 1. The Jenny-Chapin model conceives of ecosystems as being constrained by state factors (external circle) and sustained by a suite of interacting ecosystem processes known as interactive controls (internal circles). The Grand Canyon Monitoring and Research Center cultural monitoring research and 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 (photograph by Amy Draut, U.S. Geological Survey).

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

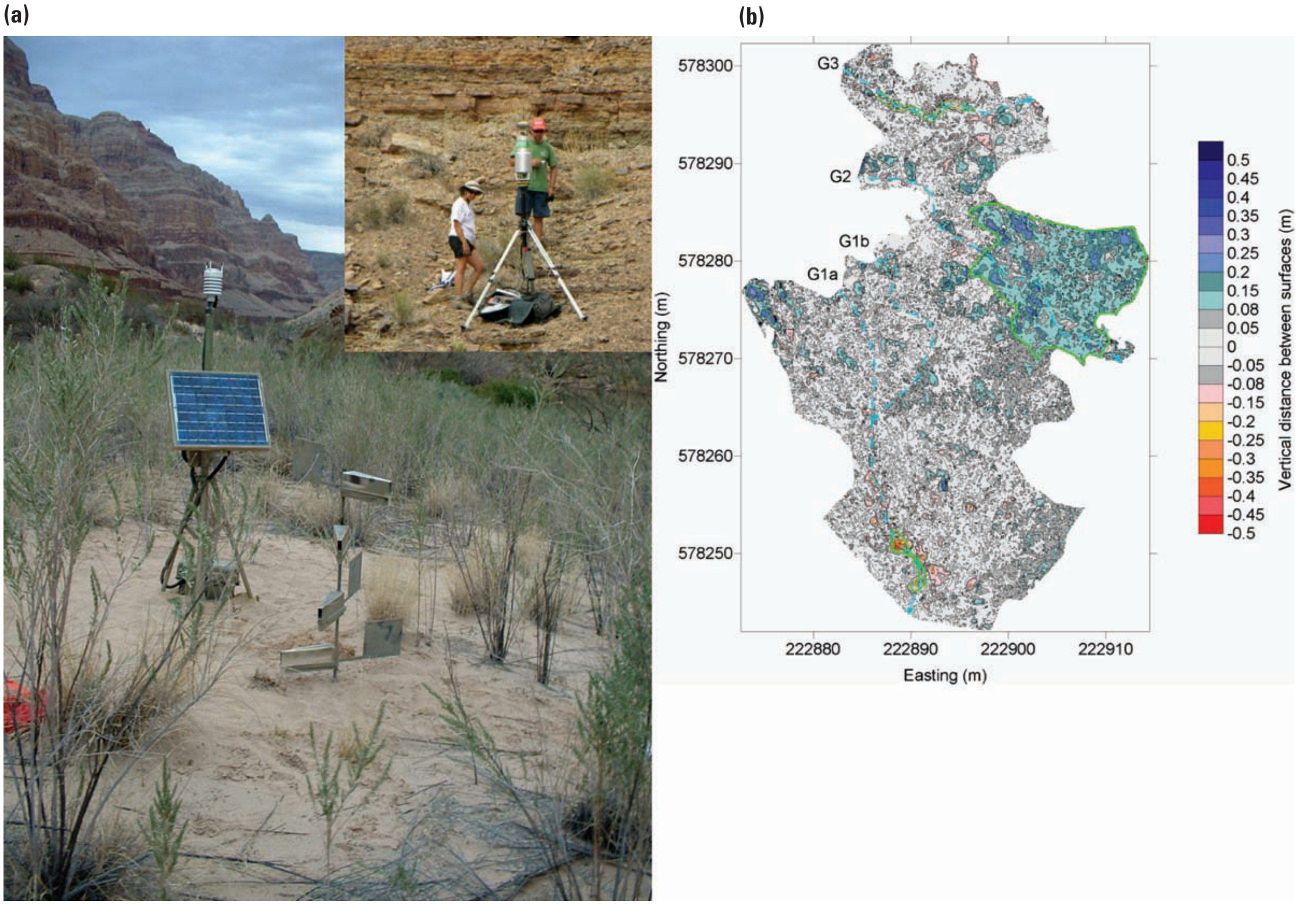

Figure 4. (a) Weather stations and sand traps positioned throughout the river corridor gather detailed data on wind velocity and direction, precipitation, temperature, humidity, and barometric pressure, and the amount of sand transported under varying weather conditions, while modern survey tools, such 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.

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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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[1] Marsh & Associates, LLC, 5016 S. Ash Avenue, Suite 108, Tempe, AZ 85282.

[2] Bureau of Reclamation, Lower Colorado Region, PO Box 61470, Boulder City, NV 89006.

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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

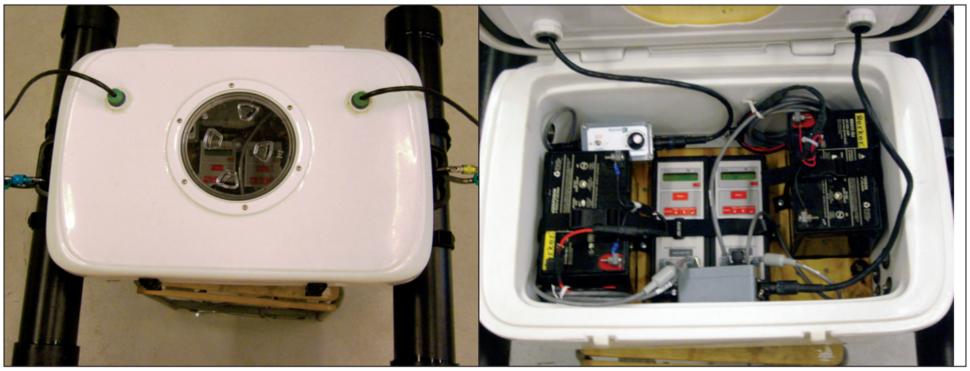

Figure 1. A remote passive integrated transponder (PIT) scanning unit built inside a 50-quart cooler containing two Biomark® FS 2001F-ISO readers and two deep-cycle lead acid batteries.

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.

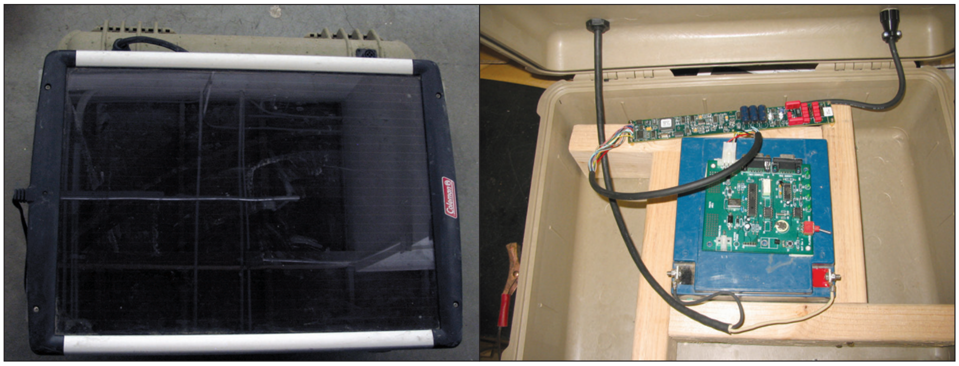

Figure 2. A remote passive integrated transponder (PIT) scanning unit built inside a 1520 Pelican™ case using an Allflex® scanner, a custom logger, a sealed lead acid battery, and a Coleman® model CL-600 solar charger.

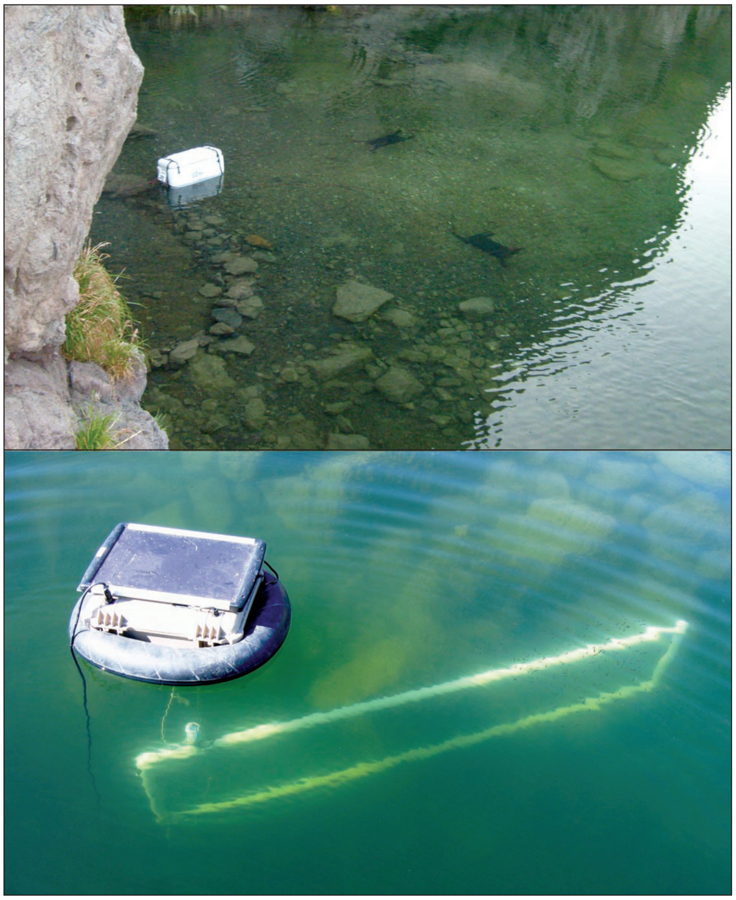

Figure 3. Passive integrated transponder (PIT) scanning units after deployment; a Biomark® in Lake Mohave, AZ-NV (top), and an Allflex® unit in Imperial Ponds, Imperial National Wildlife Refuge, AZ (bottom).

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

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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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[1] U.S. Geological Survey, Southwest Biological Science Center, 2255 N. Gemini Drive, Flagstaff, AZ 86001.

[2] U.S. Geological Survey, Pacific Science Center, 400 Natural Bridges Drive, Santa Cruz, CA 95060.

[3] U.S. Geological Survey, 345 Middlefield Road, Mail Stop 999, Menlo Park, CA 94025.

[4] Utah State University, Aquatic, Watershed, and Earth Resources, Logan, UT 84322–5210.

[5] Distances along the Colorado River in Grand Canyon traditionally are measured in river miles upstream or downstream from Lees Ferry, AZ.

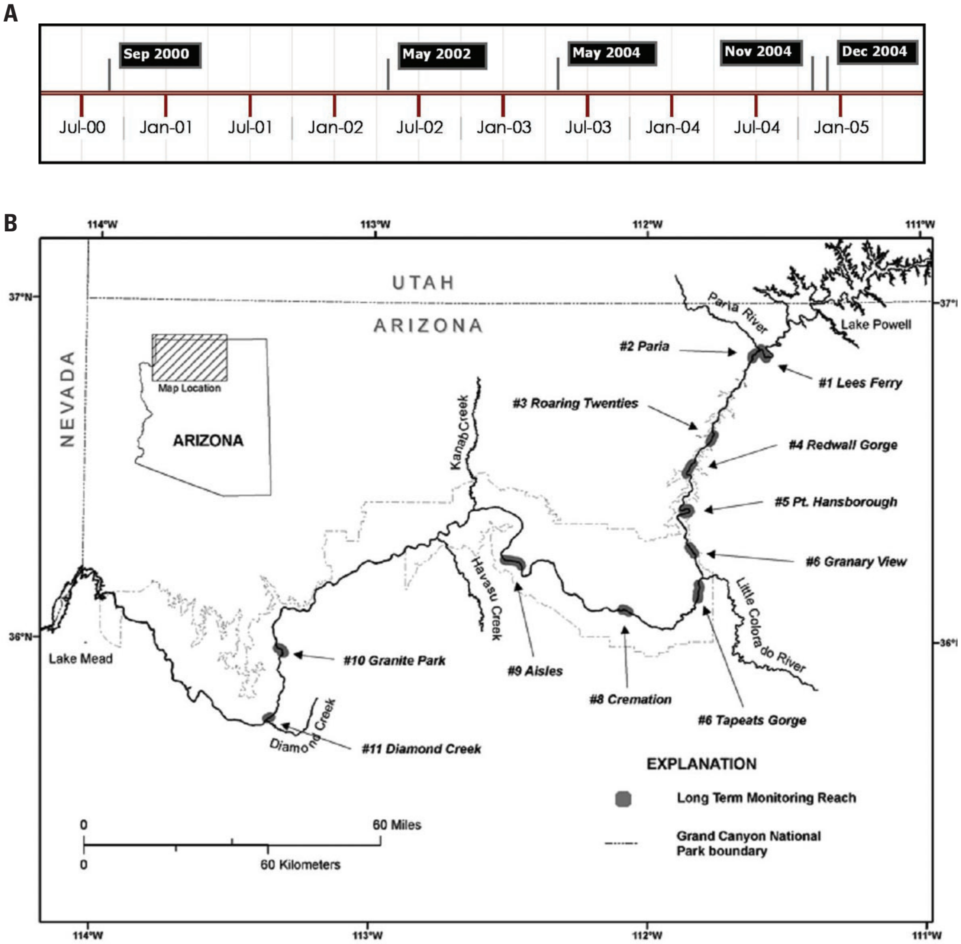


Figure 1. (A) Dates the five bathymetric surveys were conducted and (B) the location of the seven reaches surveyed.

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.

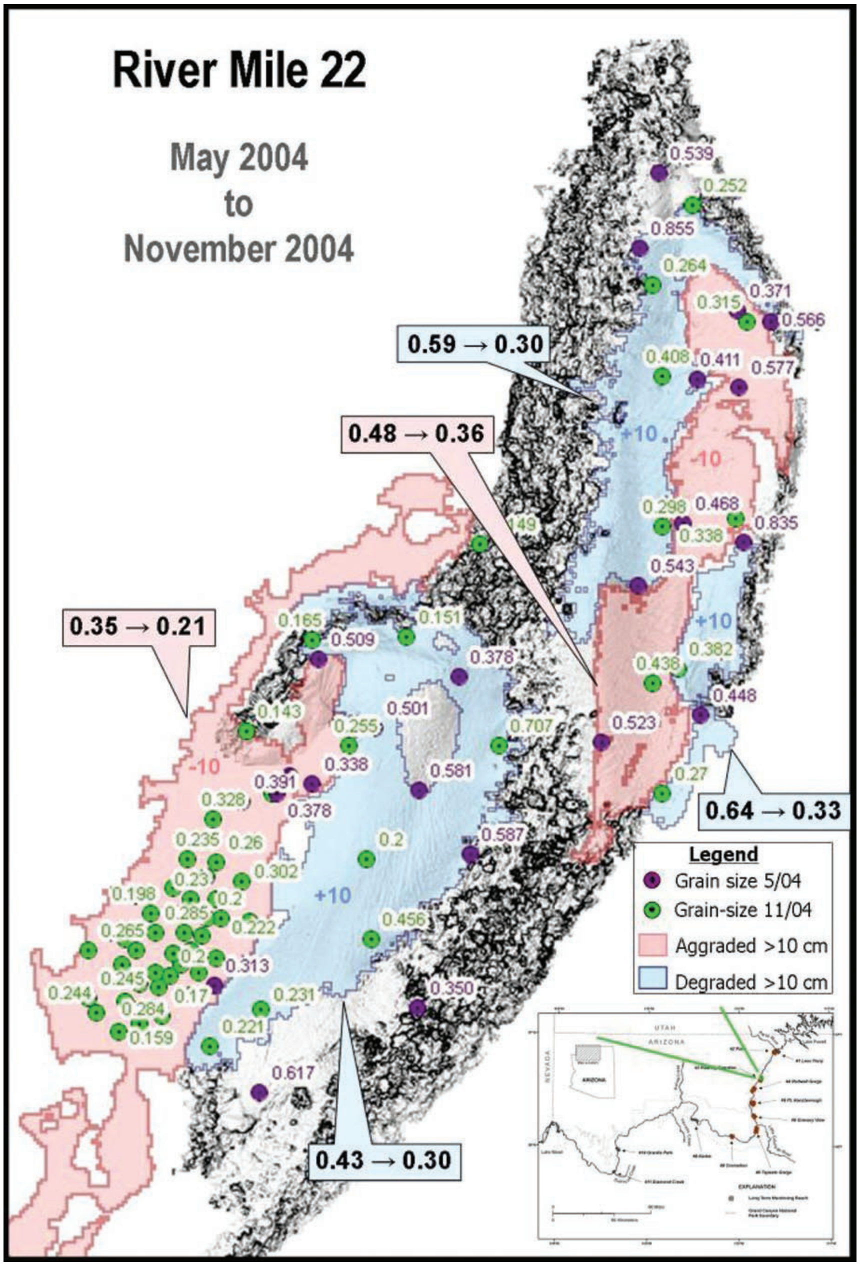


Figure 2. Regions from study reach 3, near river mile 22, that aggraded (blue) and degraded (red) from May 2004 to November 2004, a period of large sediment inputs, overlain with point grain-size measurements (in millimeters) and labeled with the change in mean grain size from survey to survey.

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

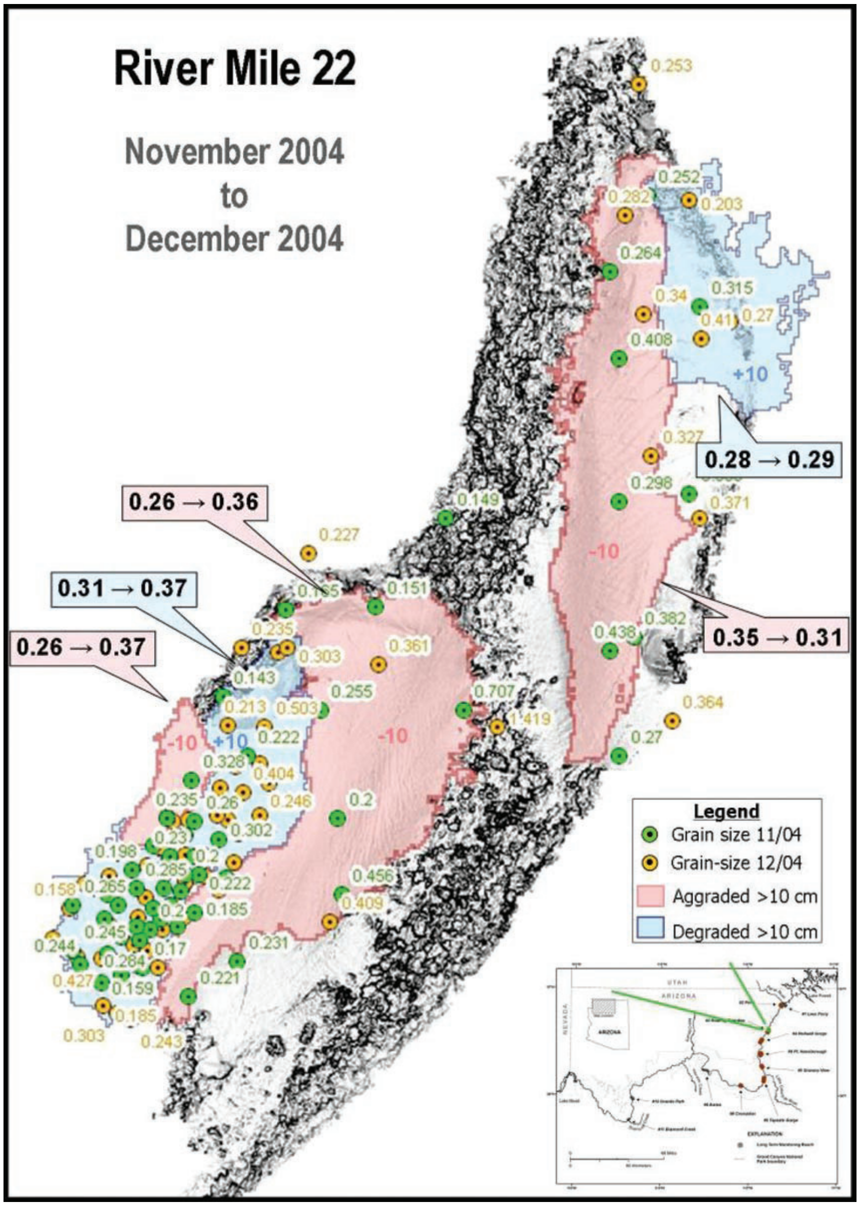


Figure 3. Discrete regions from study reach 3, near river mile 22, that aggraded (blue) and degraded (red) from November 2004 to December 2004, during which the 2004 beach/habitat-building flows (BHBF) experiment was conducted, overlain with point grain-size measurements (in millimeters) and labeled with the change in mean grain size from survey to survey.

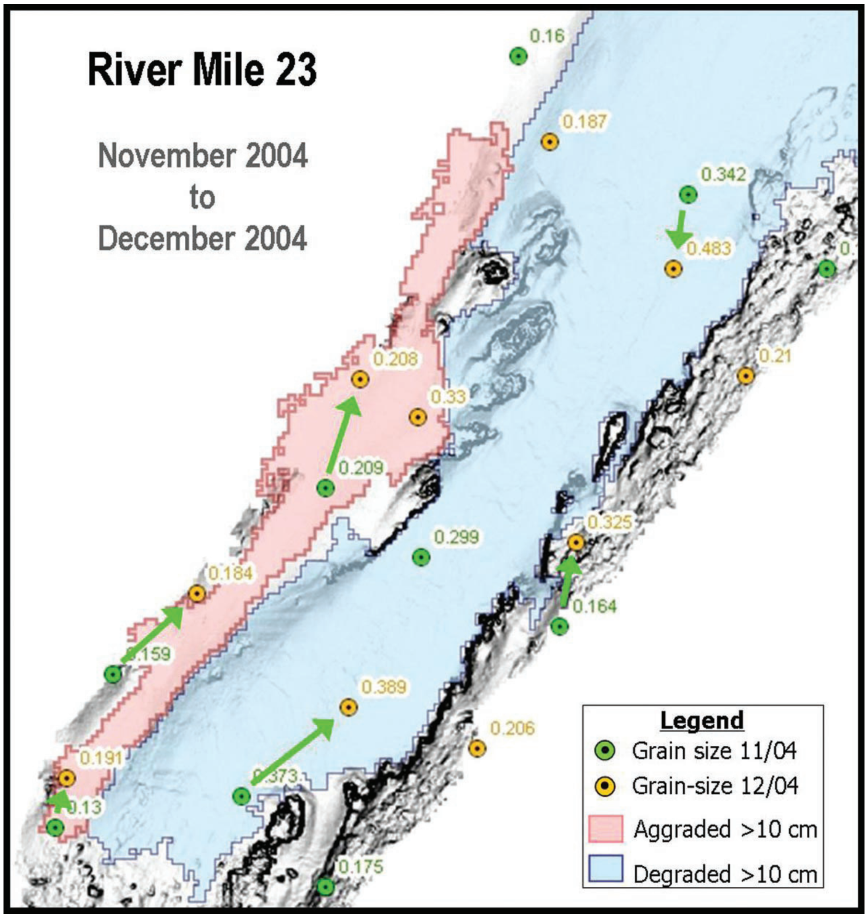


Figure 4. Point grain-size measurements from back-to-back surveys were plotted in a geographic information system (GIS). The nearest subsequent-survey neighbor to each previous-survey point was identified, and the change in grain size was calculated. Also extracted was the change in bed elevation at 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

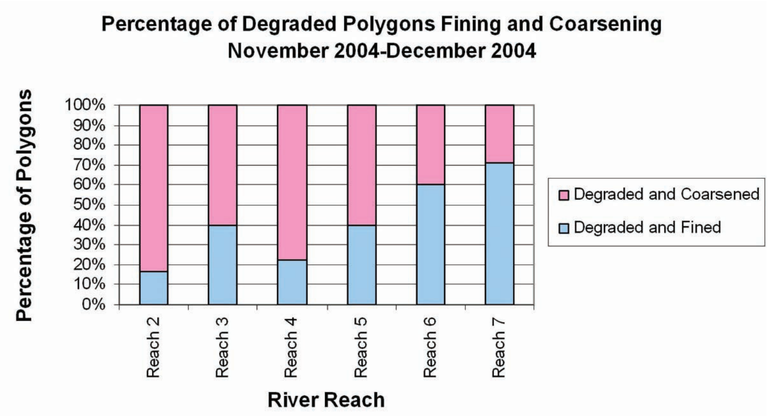
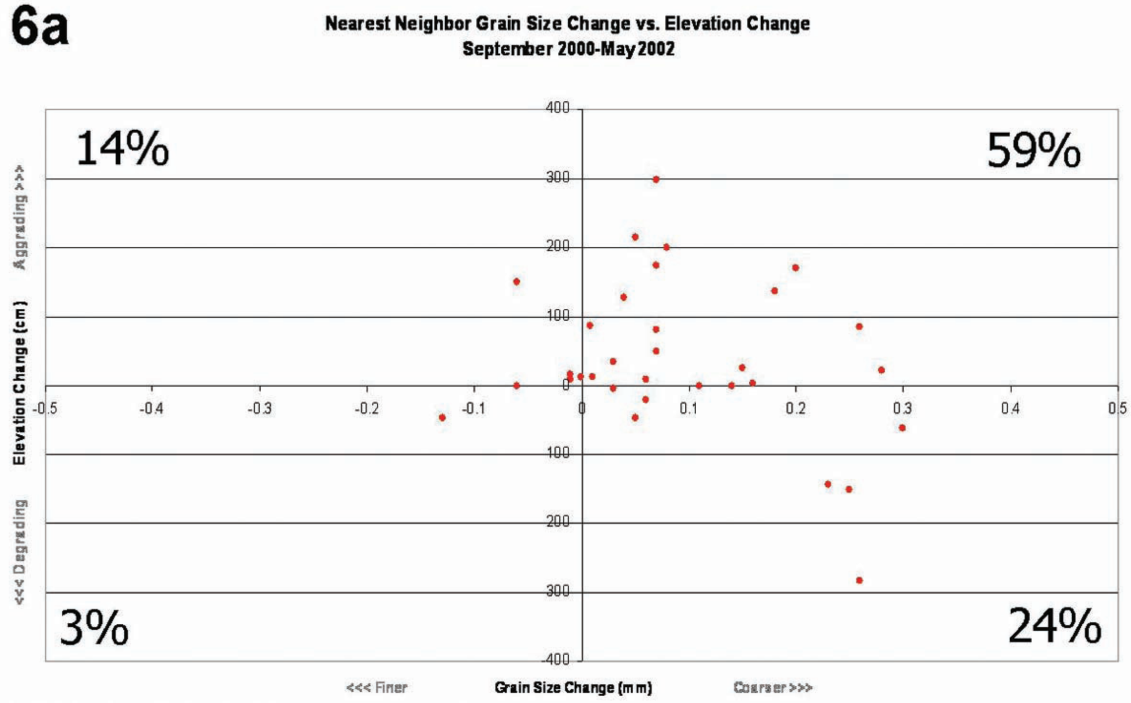


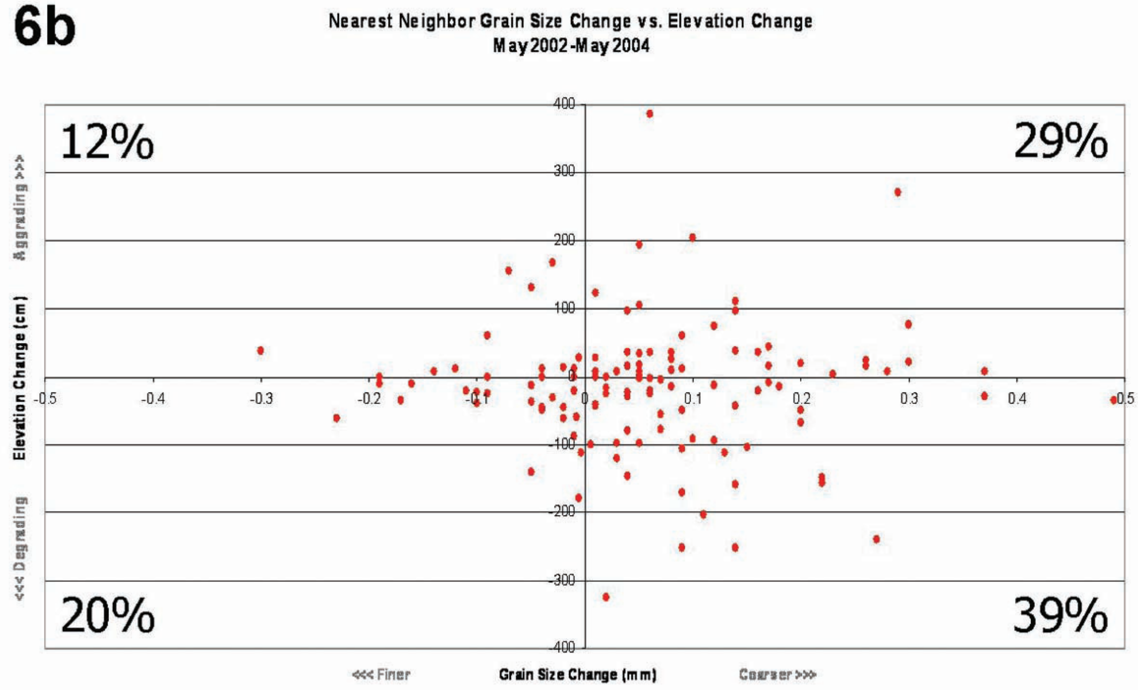
Figure 5. During the 2004 beach/habitat-building flows (BHBF) experiment, areas that degraded were more likely to coarsen in the upstream reaches and more likely to fine in the downstream reaches.

6a



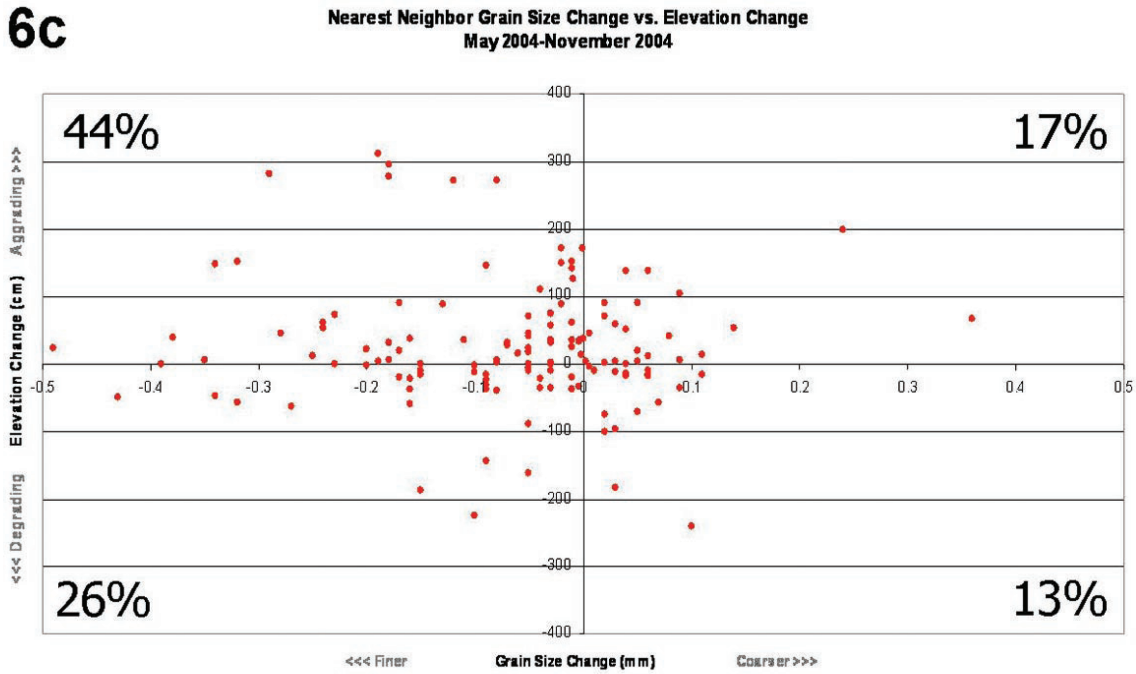
•59% of points both coarsen and aggrade

6b

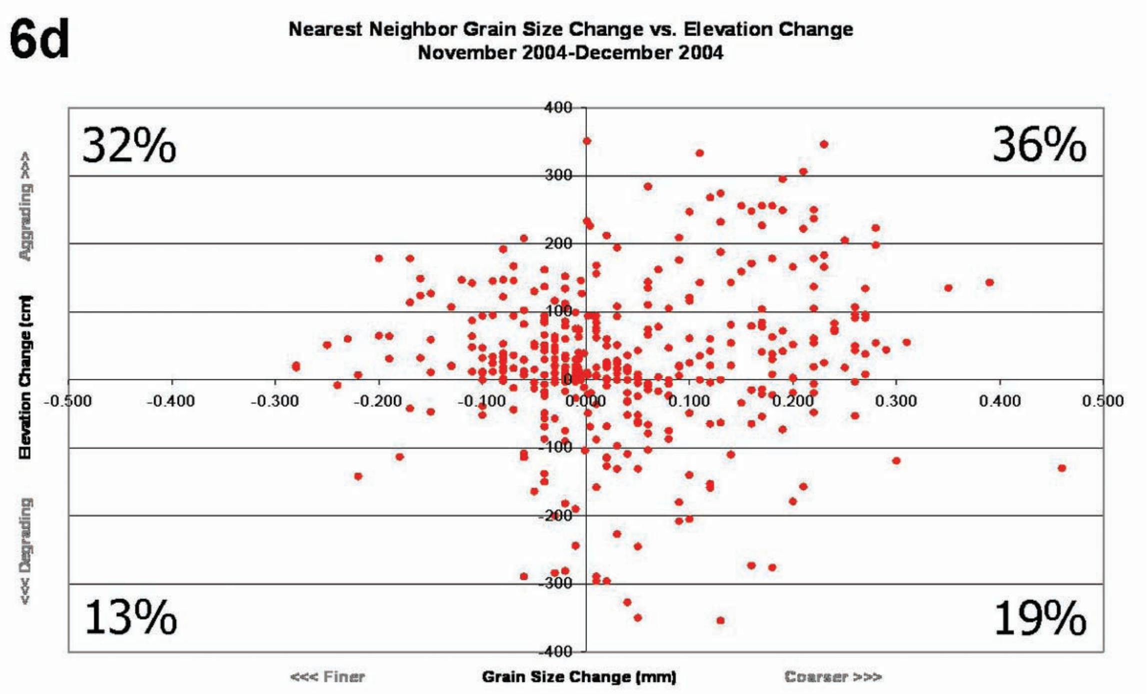


•Coarsening dominates fining, and degrading is more common than aggrading

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).

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.

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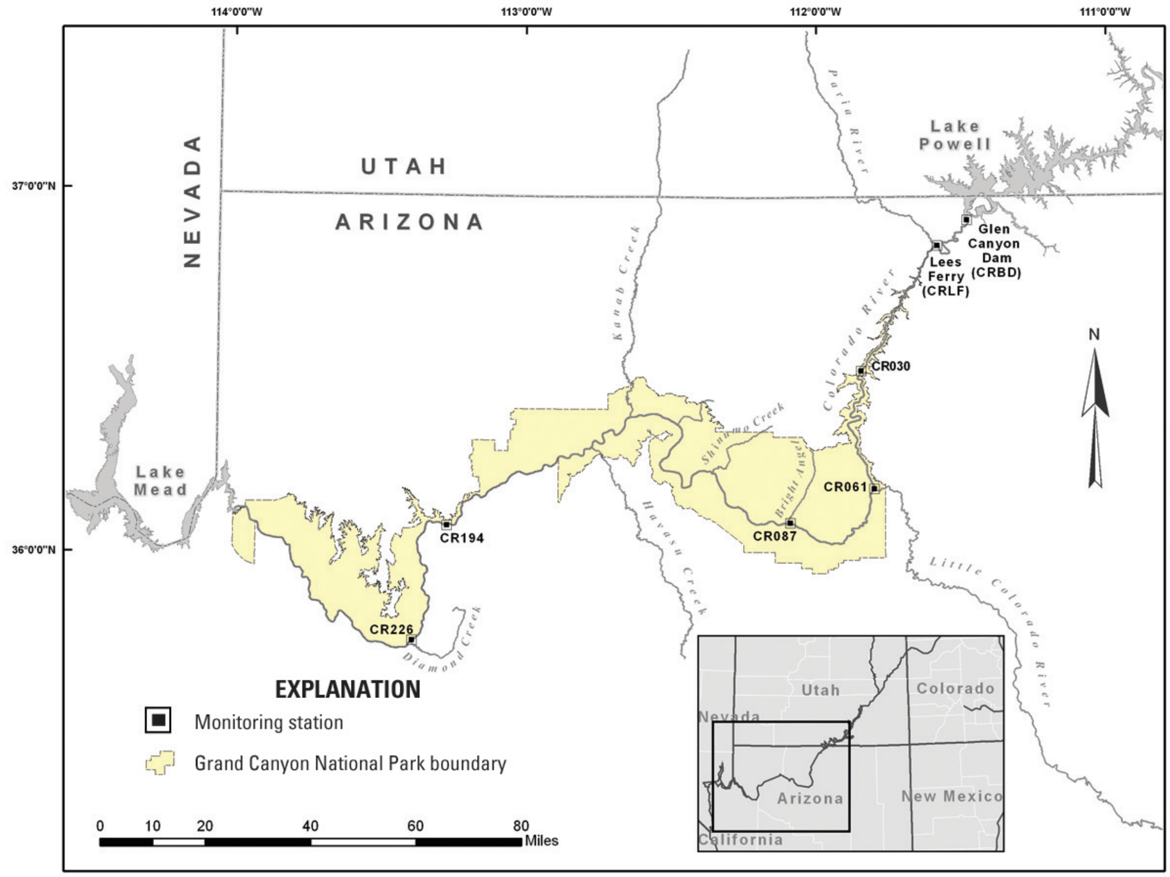

Figures 1. The Colorado River between Lake Powell and Lake Mead, northern Arizona, and specific-conductance monitoring stations.

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-

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[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,

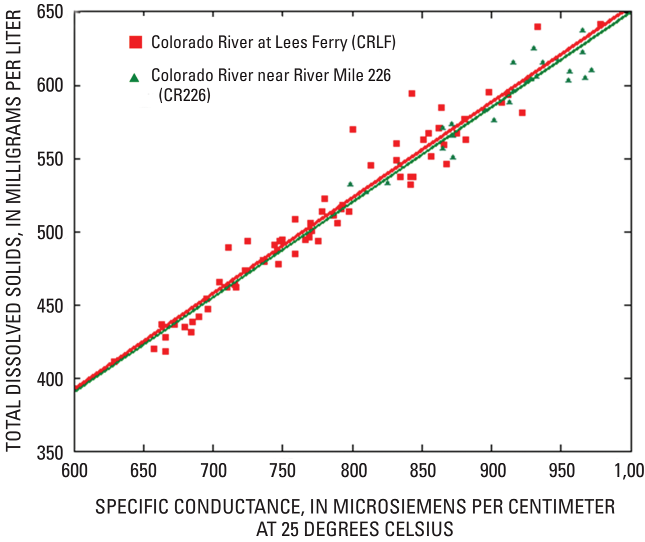

Figure 3. Relation between total dissolved solids and specific conductance of the Colorado River at Lees Ferry (CRLF) from 1991 to 2006 and of the Colorado River near river mile 226 (CR226) from 2002 to 2006. (Refer to figure 1 for the location of the two stations.)

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,

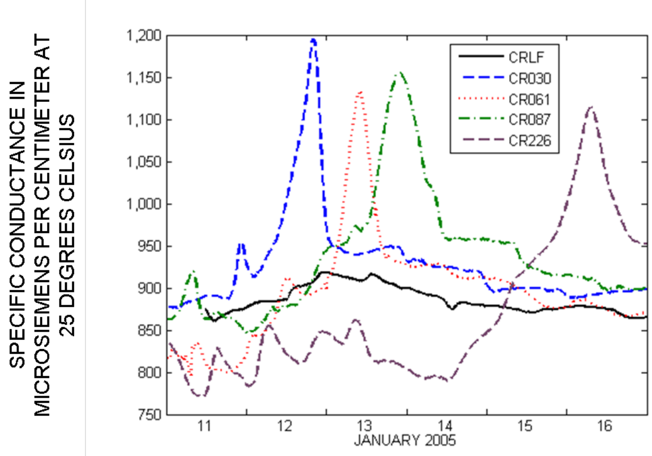

Figure 4. Specific conductance at five of the monitoring stations on the Colorado River from January 11 to 16, 2005. (Refer to figure 1 for the station locations.) The specific-conductance spike at the four stations on the Colorado River downstream from the Paria River is the result of a large Paria River flood.

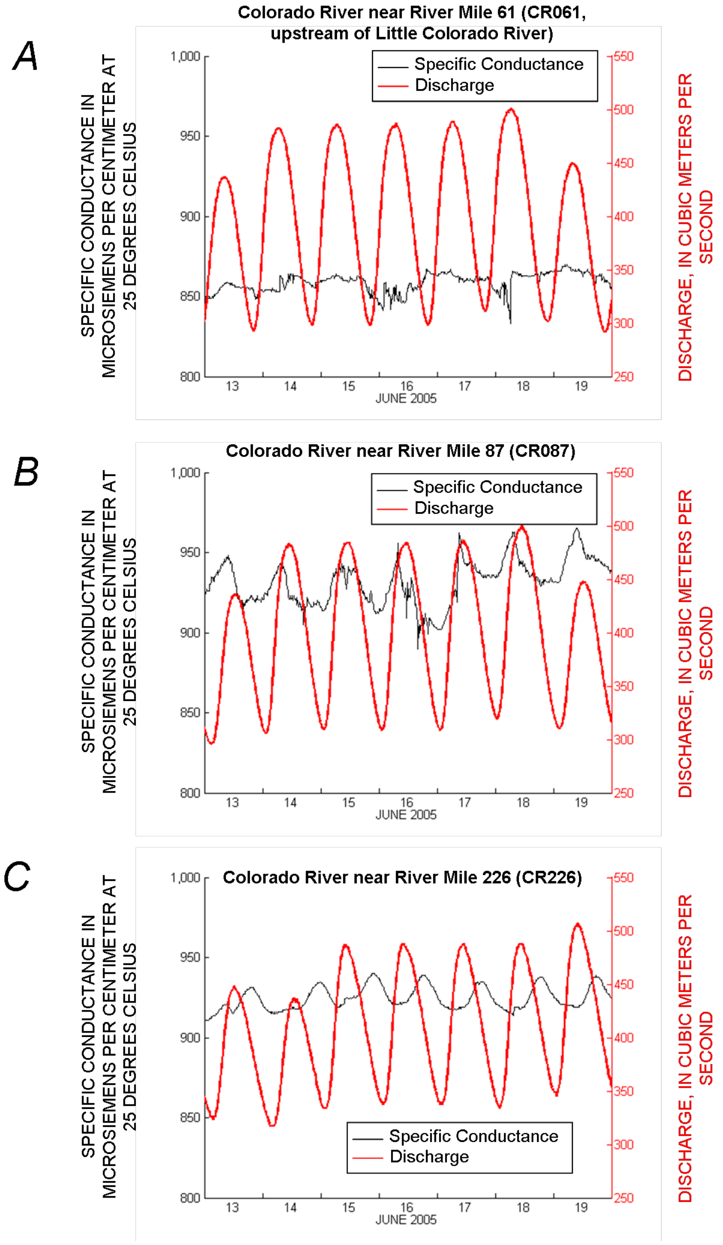

Figure 5. Specific conductance and discharge at three monitoring stations on the Colorado River from June 13 to 19, 2005. (Refer to figure 1 for the station locations.) The Little Colorado River was at base flow, contributing high-specific-conductance water to the Colorado River.

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.

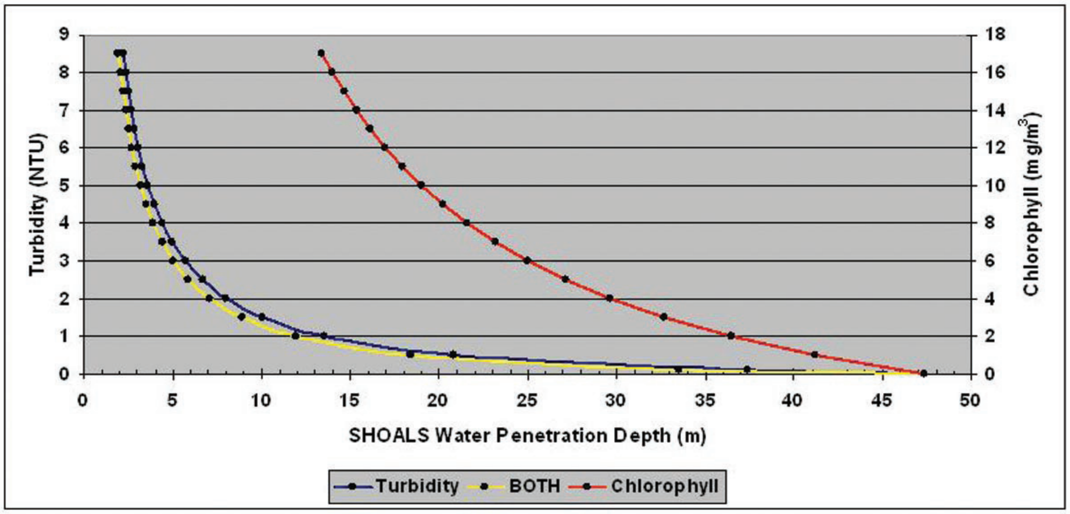

Figure 1. Theoretical water-penetration depth of SHOALS green-wavelength laser as a function of turbidity and chlorophyll concentrations, based on integrated absorption/scattering equations and reported parameter values (Gallegos, 1994, and references therein). NTU is nephelometric turbidity unit; BOTH shows the combined effect of turbidity and chlorophyll.

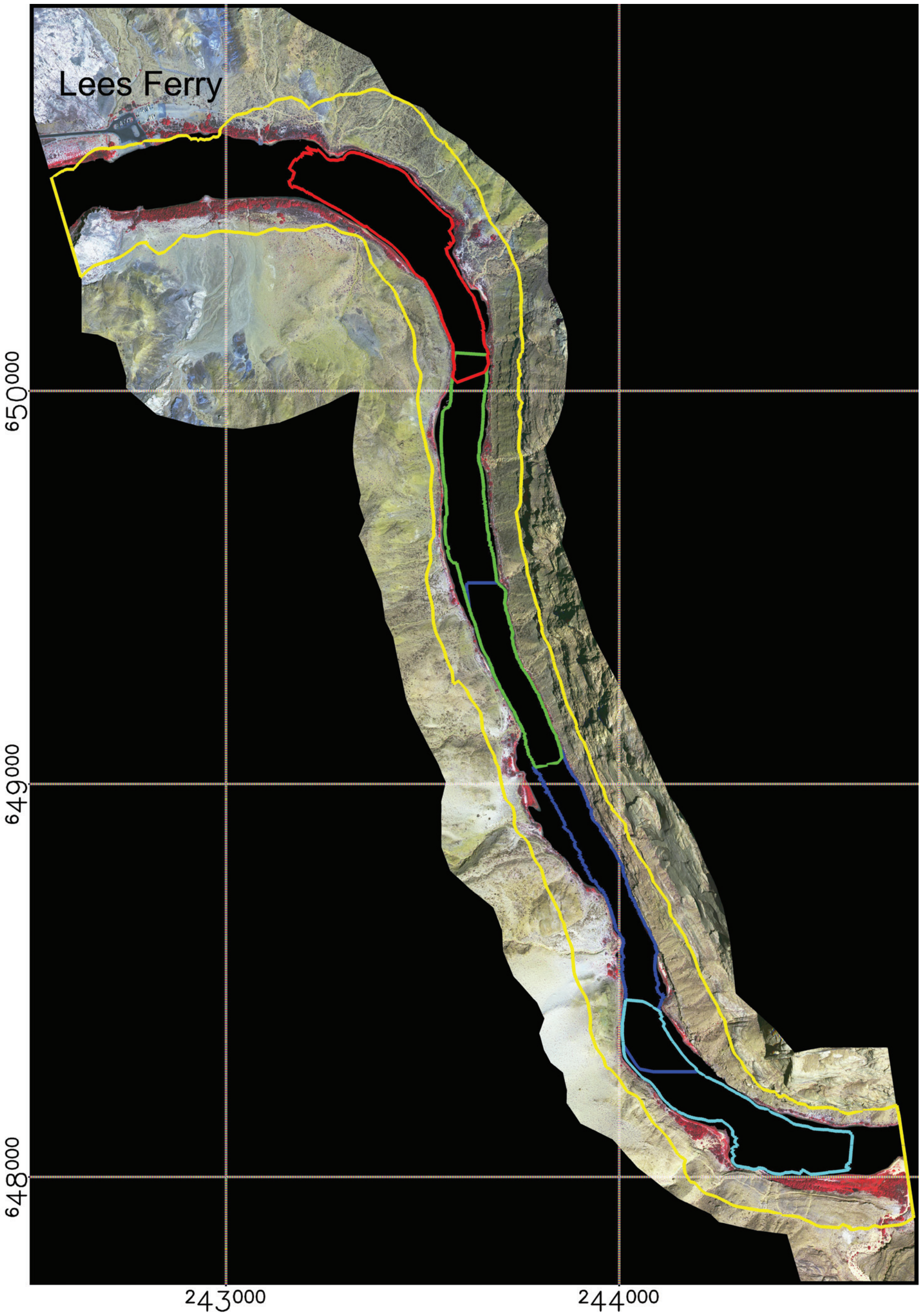

Figure 2. Color-infrared image of the Lees Ferry study area showing the SHOALS data-collection area (yellow polygon), the real-time kinematic (RTK) multibeam collection areas (red, green, and cyan polygons), and the OmniSTAR multibeam collection area (blue polygon). Image is in State Plane (Zone 202) 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

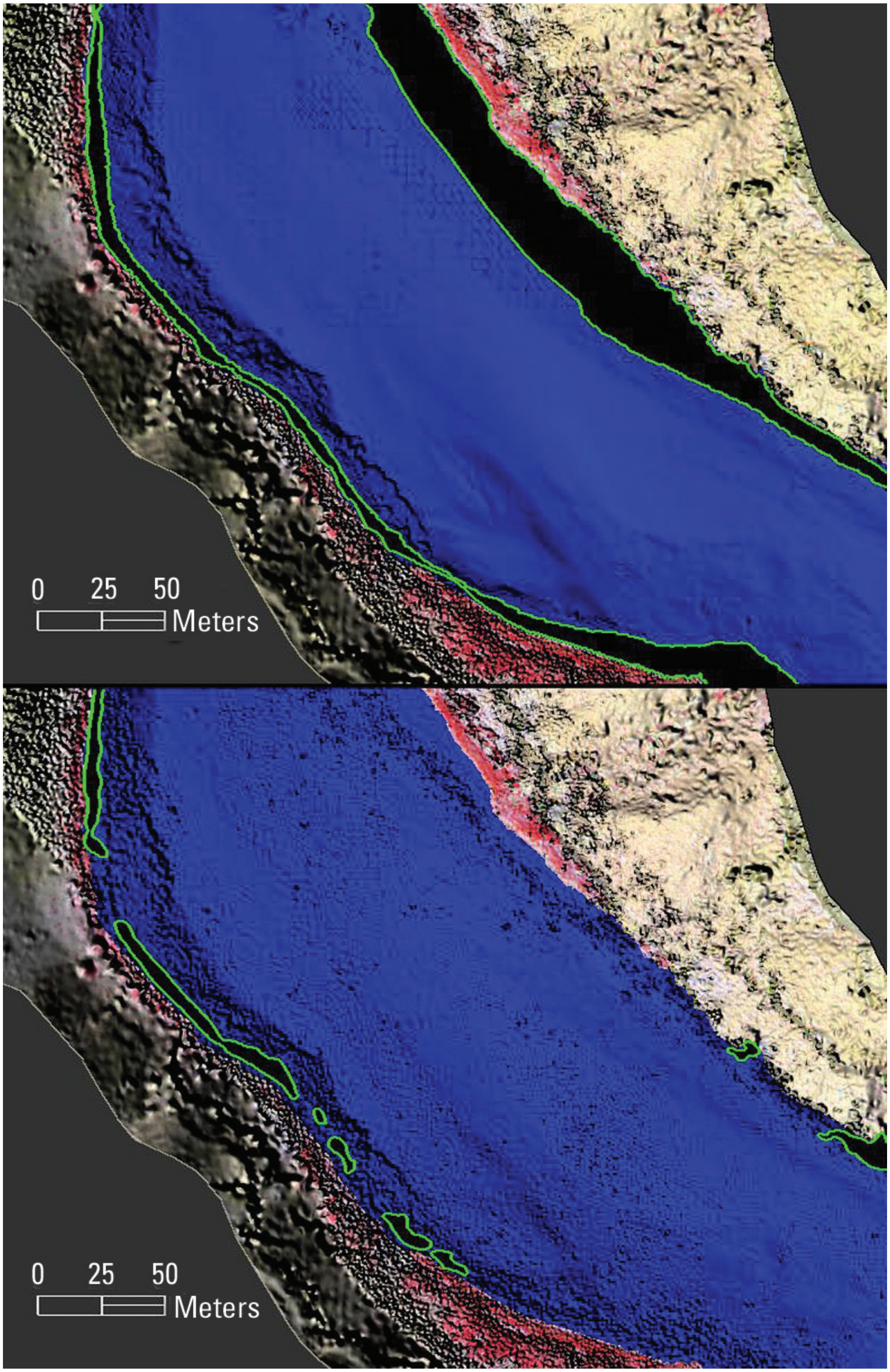

Figure 3. Shaded-relief DEM image of a portion of the Lees Ferry study area showing survey limitations of multibeam (top) and SHOALS (bottom) within shallow-water (<1 m depth) areas. Water is represented as blue, superposed on a shaded-relief, color-infrared image of the study area. Green polygons outline data gaps.

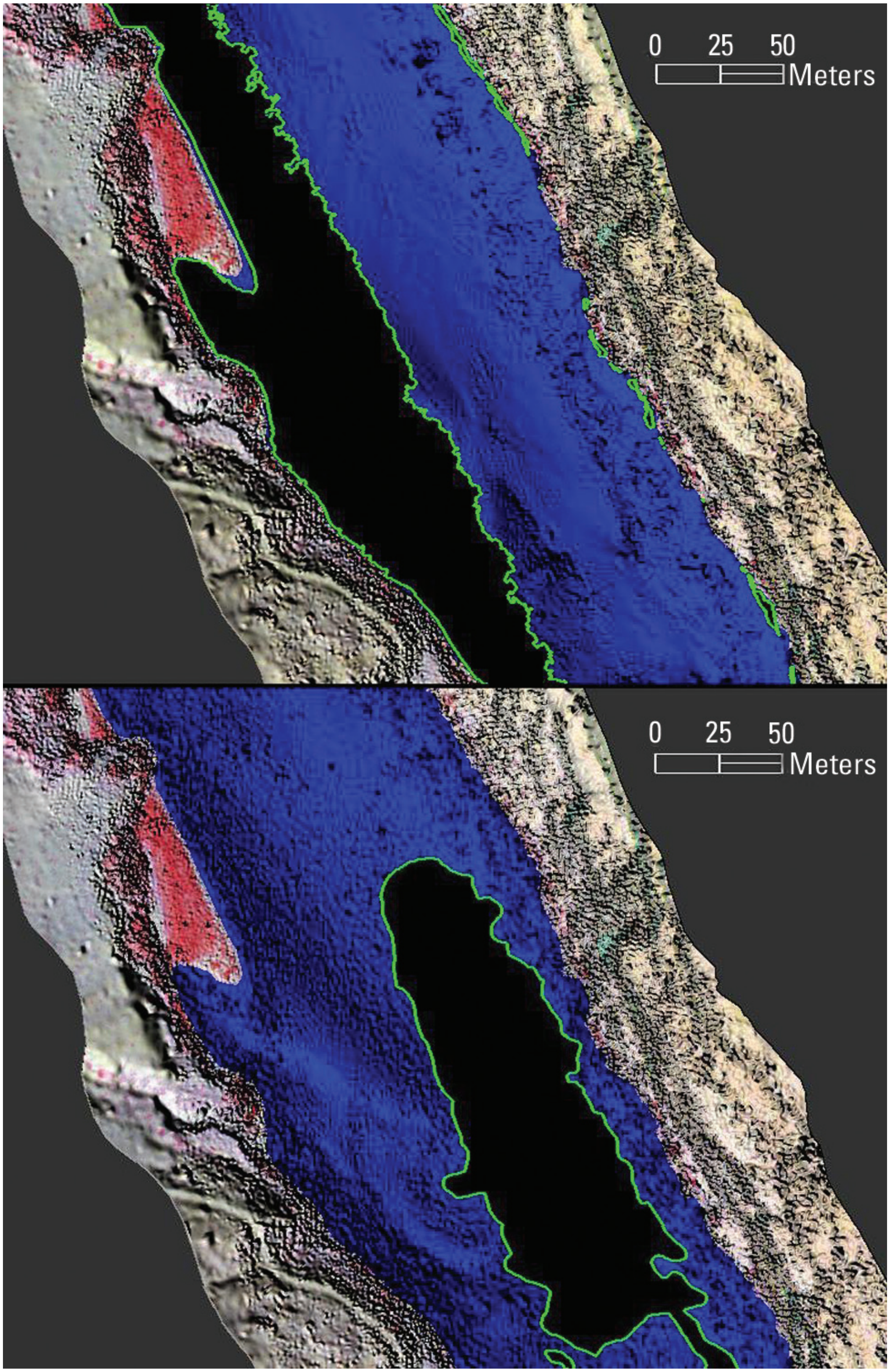

Figure 4. Shaded-relief DEM image of a portion of the Lees Ferry study area showing shallow- and deep-water limitation of SHOALS (bottom) relative to multibeam surveys (top). Water is represented as blue, superposed on a shaded-relief, color-infrared image of the study area. Green polygons outline data 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.

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.

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 • 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 • slope$_{degrees}$ – 3.3 cm]. Although the bathymetric precision measurements plot near the topographic regression line, the decrease in bathymetric precision with increasing channel

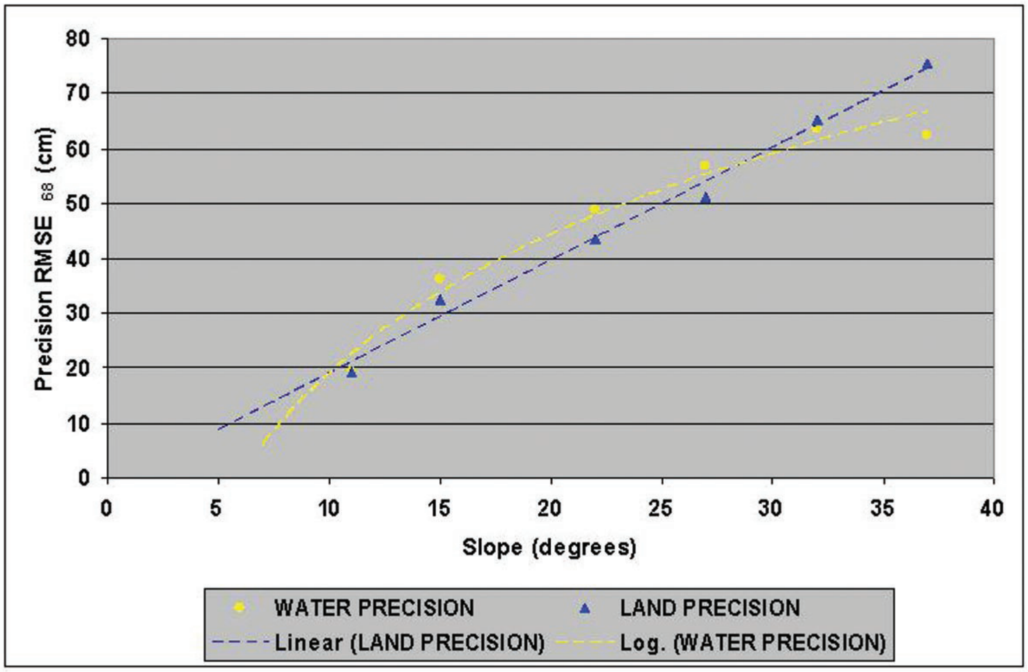

Figure 5. Variation in precision of SHOALS topography (land) and bathymetry (water) relative to surface slope. Dashed blue line represents linear regression of topographic points; dashed yellow line represents logarithmic regression of bathymetric points.

slope is closer ($R^2 = 0.97$) to a logarithmic relation [RMSE_{95} (cm) = 71.5 • ln ($\mathrm{slope}_{\mathrm{degrees}}$) – 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.

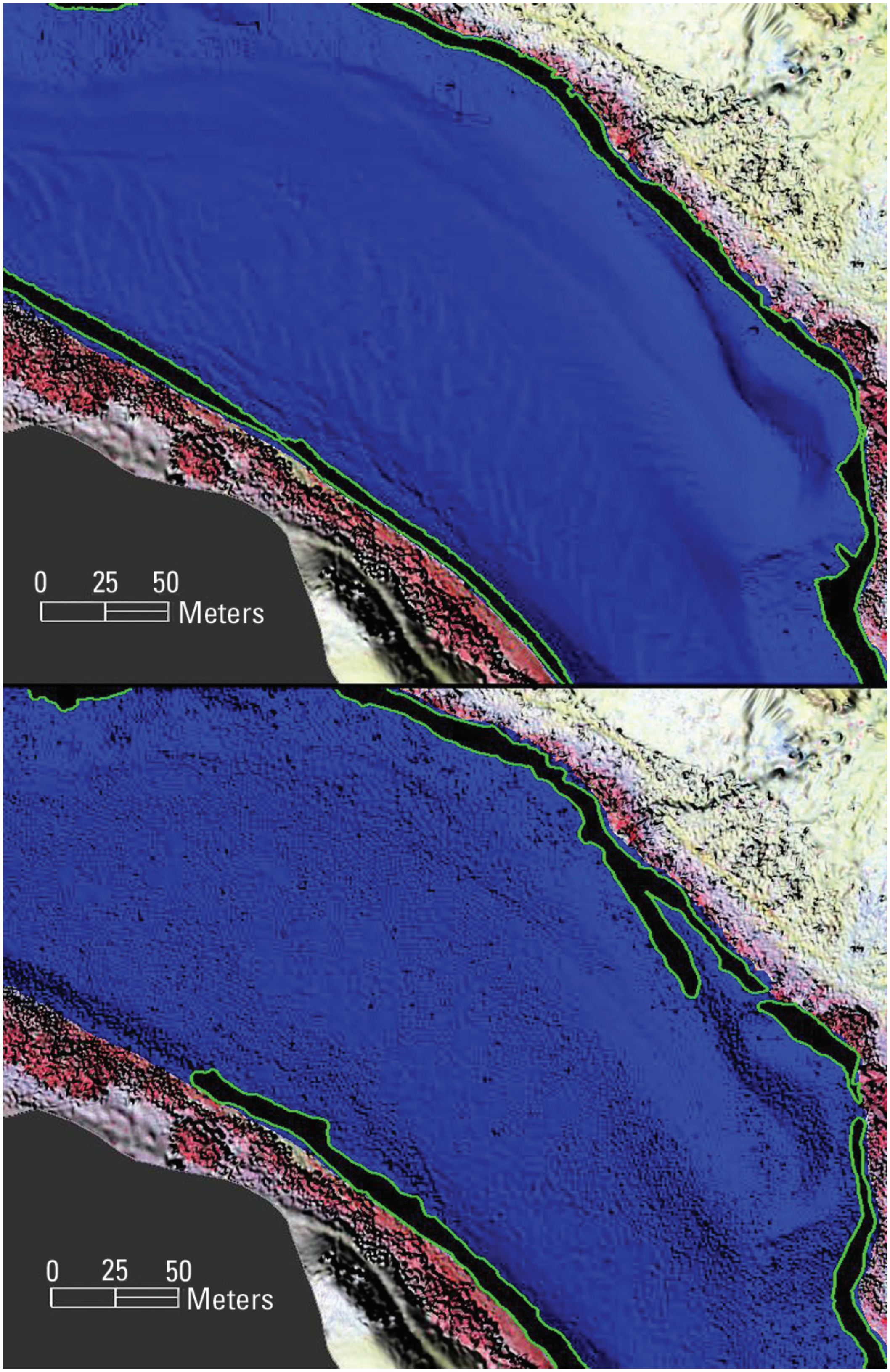

Figure 6. Shaded-relief DEM image (at 0.5-m cell resolution) of a portion of the Lees Ferry study area showing the greater substrate detail provided by 0.25-m multibeam data (top) relative to that provided by 1-m SHOALS data (bottom). Water is represented as blue, superposed on a shaded-relief, color-infrared image of the study area. Green polygons outline data gaps.

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