MONITORING REPRODUCTION, RECRUITMENT, AND POPULATION STATUS OF RAZORBACK SUCKERS IN THE UPPER COLORADO RIVER BASIN

By

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EXECUTIVE SUMMARY

Decline of endangered razorback sucker *Xyrauchen texanus* has been attributed to alterations of physical habitat and negative effects of non-native fishes. In the Upper Colorado River Basin, stream flow reduction due to storage of spring runoff in reservoirs, and effects of channelization and levees reduces frequency and duration of floodplain wetland inundation, a critical habitat needed for recruitment. Non-native fish effects include predation, competition, and potentially, hybridization with razorback sucker. In the Upper Colorado River Basin, population declines were substantial enough to extirpate wild stocks of razorback suckers and management efforts included substantial stocking of captive-reared razorback suckers beginning in 1995. To date, survival rates of stocked fish are low but some individuals are reproducing in both the Green and Colorado River systems. Populations are substantial enough to warrant monitoring of razorback sucker populations in the Green and Colorado River systems and their tributaries; this report details efforts needed to understand population status of razorback suckers and measure their potential response to management actions including flow management activities that are ongoing.

Early life history stage sampling for razorback sucker in the Green River is determined suitable in its present form for those regularly reproducing populations. Those efforts include light trap and seine sampling in the lower and middle Green River, which will provide information on first spawning and duration of reproduction as well as annual reproductive effort. Timing of spawning and appearance of larvae is an important real-time signal to trigger flow releases from Flaming Gorge Dam, which is hypothesized to provide connections at appropriate times to important floodplain wetland nursery habitat. Additional sampling effort recently added by the Recovery Program for Endangered Fishes in the Upper Colorado River Basin (Program) will provide additional information to evaluate the efficacy of the larval trigger, where first presence and duration of presence of razorback sucker larvae will guide timing and duration of releases from Flaming Gorge Dam, and was included as a component of this monitoring program. Annual presence and abundance of larvae is also a potential metric of recovery because reproduction must be adequate to sustain self-recruiting populations. Reproduction is less regular in the Colorado River, Utah and Colorado. As such, ongoing sampling is more widespread in order to determine potential distribution and abundance of larvae and to identify possible spawning areas. Modifications to that program will occur once more regular reproduction is observed and when larvae are more abundant. Nevertheless, findings will be useful to evaluate population status and response to management actions, including flow management. We suggest additional sampling and research to better understand fish distribution and entrainment into floodplain wetlands, capture efficiency, and genetic verification of identified larvae.

Valuable capture-recapture data for large juvenile and adult razorback suckers in the Green and Colorado River systems is available from existing sampling programs, particularly that for Colorado pikeminnow *Ptychocheilus lucius*. That sampling occurs on a three-year on, two-year off cycle and will form the foundation for monitoring larger-bodied razorback suckers. However, that sampling by itself is insufficient to effectively monitor survival and population abundance of large juvenile and adult razorback suckers and additional sampling and recaptures of tagged fish are needed in those years to increase recapture rates. We analyzed habitat types of captures made in spring in each system, and additional sampling should minimally focus on slackwater channel margin habitat including flooded tributary mouths, backwaters, and eddies in each system. Additional active sampling (e.g., electrofishing) as well as passive sampling (e.g., fyke/trap nets) is proposed. Monitoring of concentration areas, especially those near spawning areas, via PIT tag detector arrays would be especially useful because those may maximize captures when fish are concentrated and reduce potential handling effects and disturbance. A last resort would be more active sampling directly over spawning areas, an option which should be explored only if other techniques do not yield sufficient recapture information. Additional sampling effort should overlap sampling for Colorado pikeminnow closely in time and space so that it can be incorporated into abundance estimation data; data useful for estimation of survival estimates can be collected over a broader time frame and is not as restrictive as that for abundance estimates. Simulations were also conducted to guide the minimum levels of sampling needed to raise probabilities of capture to 1) reduce bias of parameters derived from recapture data, and 2) increase precision to levels that provide useful estimates. We also make recommendations for frequency and type of data analyses to make best use of data gathered in the future. Implementation of this monitoring program should increase the ability of managers to make informed decisions regarding the status of razorback suckers in the Green and Colorado River systems, which should assist in evaluation of conservation and recovery status of the species.

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INTRODUCTION

Endangered razorback sucker *Xyrauchen texanus* was once widespread and abundant throughout the Colorado River Basin but is now rare (McAda and Wydoski 1980; Minckley 1983; Bestgen 1990; Minckley et al. 1991; U. S. Fish and Wildlife Service 1991; 2002). Concentrations occur in Lake Mohave and Lake Mead reservoirs, Arizona and Nevada, and in the upper Colorado and Green rivers, and the San Juan River, Utah, Colorado, and New Mexico, mostly as stocked hatchery fish; the Lake Mead Reservoir population is thought a wild and self-sustaining population (Minckley 1983; Tyus 1987; Bestgen 1990; Minckley et al. 1991; Modde et al. 1996; Holden et al. 2000; Albrecht et al. 2008; 2010 Zelasko 2008; Zelasko et al. 2009; 2010; 2011). In the lower Colorado River Basin, wild fish in Lake Mohave numbered about 4,000 individuals in 2001, an enormous decline relative to historical populations, and has since declined to only about 24 fish (Minckley et al. 1991; Marsh et al. 2003; 2005; Kesner et al. 2010). An active population replacement program is ongoing in Lake Mohave with mixed results, as stocked fish have relatively low survival (Marsh et al. 2003; 2005; Kesner et al. 2010). Wild populations of razorback suckers are mostly dominated by large, old individuals, and recruitment rates in most localities are thought low or non-existent; the Lake Mead population apparently has ongoing recruitment and consists of younger as well as older fish (Minckley 1983; Tyus 1987; Tyus and Karp 1990; Minckley et al. 1991; Gutermuth et al. 1994; Modde et al. 1996; Holden et al. 2000; Bestgen et al. 2002; Marsh et al. 2005; Albrecht et al. 2008; 2010).

In the Upper Colorado River Basin, wild fish were extirpated from the San Juan River but stocked hatchery fish are surviving and reproducing annually (Platania et al. 1991; Brandenburg and Farrington 2009; Bestgen et al. 2009; pers. comm., S. Platania., American Southwest Ichthyological Researchers, Albuquerque, NM,). Wild razorback suckers in the Upper Colorado River were thought extirpated many years ago (Bestgen 1990; Bestgen et al. 2002). Abundance of wild adult Green River razorback suckers was estimated at about 300 to 950 during the 1980 to 1992 period (Lanigan and Tyus 1989; Modde et al. 1996) but declined to less than about 100 fish by 2000 and that population was likely extirpated soon after due to mortality of old fish (Bestgen et al. 2002).

To bolster populations of razorback sucker in the Upper Colorado River Basin, hatchery-reared fish were stocked beginning in 1995; relatively large numbers were stocked beginning in 2000 and size of stocked fish increased over time (Burdick 2003; Zelasko et al. 2010; 2011).

Survival of relatively large (> 250 mm TL) hatchery-reared razorback suckers released into the Green and Colorado rivers has increased population abundance and some are now reproducing, including in the middle and lower Green rivers and for the first time, in the lower White River in 2011 (Burdick 2003; Modde et al. 2005; Zelasko 2008; Osmundson and Seal 2009; Zelasko et al. 2009; Zelasko et al. 2010, Bestgen et al. 2011, A. Webber, USFWS, Vernal, Utah). Razorback sucker in the Colorado River, Utah and Colorado, are also reproducing (Osmundson and Seal 2009). Thus, a program to monitor population distribution, reproduction, recruitment, and status in the Upper Colorado River Basin is appropriate at this time.

Our objective was to detail a procedure to monitor status and trends for razorback sucker populations in the Green and Colorado River systems of the Upper Colorado River Basin and specifically to 1) compile literature and sampling data relevant to understanding early life and adult razorback sucker distribution and ecology, 2) conduct analyses appropriate to understanding sampling intensity, and 3) make recommendations for sampling. Data gathered will be useful to document important life history parameters and vital rates, identify roadblocks to conservation, and ultimately, quantify measures of population success that will indicate when recovery has been achieved. We consider sampling programs for early life stages, juveniles, and adults, using existing studies and associated data as well as suggestions for obtaining new information. We also detail potentially problematic issues which will enable better parameter estimation and monitoring to detect trends.

STUDY AREA

The main study area was the Green River from the confluence of the Yampa River downstream to the confluence with the Colorado River, and warmwater reaches of the Colorado River and lower Gunnison River from upstream of Grand Junction downstream to Moab, Utah (Figure 1). Future sampling may include downstream areas including the Lake Powell inflow.

METHODS

We used literature, existing information, and personal experience of the authors and others who have conducted field sampling to first identify information that is known about razorback sucker. Key information was understanding the history and reasons for decline of razorback sucker, what life history stages are relatively well-known based on sampling, which stages are not, and what life history stages are thought to be the primary impediments to recruitment and survival in wild populations. Thus, we took a life cycle approach, information that was summarized in a conceptual model for razorback sucker (Zelasko 2009; Valdez et al. 2011) and was similar to that for endangered Colorado pikeminnow *Ptychocheilus lucius* (Bestgen et al. 2007). We also considered the literature and personal experiences in making recommendations about sampling for each life stage. We also modeled aspects of this plan after portions of another life cycle monitoring program that seems robust, that for Colorado pikeminnow. Existing programs (e.g., Bestgen et al. 2007; 2010; Zelasko et al. 2010) provided information about levels of sampling needed to produce various levels of accuracy and precision for abundance and survival estimates. Those estimates were also used to simulate levels of sampling needed to produce recommendations for obtaining robust estimates for razorback sucker. Simulations in Program MARK (details below) used best available information to guide initial parameters regarding population size and capture rates of razorback suckers.

RESULTS AND DISCUSSION

Early life stage razorback sucker data needs and use

Early life stage sampling for razorback suckers has a legacy of success in both the Lower Colorado River Basin (lower basin) as well as the Upper Colorado River Basin (upper basin). In the lower basin, larvae sampled in Lake Mohave were useful to track annual reproductive patterns and to understand that recruitment bottlenecks did not occur at the spawning phase, but rather later in the larval phase due to absence of older larvae (Minckley 1983; Minckley et al. 1991). In the Upper Colorado River Basin, successful sampling for razorback sucker larvae began as early as 1984, when several larvae were captured in the vicinity of Ashley Creek, not far downstream of known spawning areas (those larvae were verified as bona fide razorback suckers rather than questionable in June 2011, based on re-assessment of their identifications by the senior author and D. E. Snyder, Larval Fish Laboratory, Colorado State University).

Early life stage sampling has several main goals and information will be useful in a number of assessments. First, presence of larvae can alert biologists to the presence of adult fish;

without adults there can be no larvae in the vicinity. This is important because larvae are often easier to sample than rare adults. Second, presence of larvae in samples has verified that certain areas are used by adults for spawning. This has recently been the case for the Yampa River, as well as the White River (spring 2011), and is suspected in the San Rafael River and perhaps the Green River upstream of the Yampa River in Lodore Canyon (unpublished data based on samples identified at the Larval Fish Laboratory).

Information that documents first occurrence of razorback sucker larvae in spring light trap samples has also been used in the middle Green River to describe and potentially optimize the timing and duration of flow releases from Flaming Gorge Reservoir and availability of floodplain wetland habitat (Bestgen et al. 2011). The goal of releases is to time flow to coincide with first presence of larvae in spring so that they can be transported into productive floodplain wetlands and subsequently, grow and survive at relatively high levels. Such light trap sampling will continue to be used as managers incorporate presence of native suckers in light trap samples as a trigger to release flows from Flaming Gorge Dam. Presence of larvae has also been used to determine duration of flow releases from Flaming Gorge Dam, which extends connection with floodplain wetlands, and such use is expected in the future as well. Thus, sampling to detect presence of larvae is important now and into the future to determine critical dam management operations.

Abundance patterns of larvae captured in light traps will also be useful to assess population status of razorback suckers. For example, captures of larvae will continue to provide information about survival and reproductive status of razorback suckers stocked into various locations in the Upper Colorado River Basin. Increased captures of razorback sucker larvae in the middle Green River (Figure 2 and Table 1) since at least 2004 has given managers clues about increased spawning success by stocked adults. Such information will continue to be useful there as well as in other localities where adults are now colonizing or are established. Metrics such as captures per night at sentinel locations will be useful to delineate trends over time.

Further, abundance levels on an annual basis will give investigators clues about the relative effects of different flow and water temperature patterns on reproductive success. This may be especially important as investigators proceed with experiments to determine flow timing, magnitude, and duration needed to effect recruitment of early life history stages of razorback sucker in floodplain wetlands of the middle Green River. This is the case because without an initial assessment of levels of reproduction by razorback suckers, estimates of entrainment and, hopefully, recruitment, will have little basis for comparison relative to environmental conditions.

Finally, presence/abundance of larvae captured in light traps, along with adequate numbers of juveniles and adults, is a potential metric for evaluating whether status of razorback sucker is sufficiently strong to permit down or delisting. This is true because without consistent and strong reproduction at various locations, population stability and recovery is not assured. It should be noted that strong reproduction does not necessarily ensure recruitment, a process that may depend on factors other than number of larvae present.

Larvae and juvenile life stage sampling

Middle Green River.—Sampling programs specifically tailored to assess reproductive success of razorback suckers in the Green River began in 1992 (Muth et al. 1998; 2000) and have continued to date (Bestgen et al. 2002; Bestgen et al. 2011). Early sampling techniques included drift netting, seining, and light trap sampling, then a relatively new technique. Studies and subsequent field sampling verified the efficacy of light trap sampling as an efficient sampling technique, although drift nets and seining have also been used in specific instances and seining is the sole and successful technique used in San Juan River monitoring (Snyder and Meismer 1997; Hedrick et al. 2009; 2010; Osmundson and Seal 2009; Bestgen et al. 2011; pers. comm., S. Platania). Razorback sucker larvae have been detected in the middle Green River in every year of sampling since 1992 (Table 1 and Figure 2, in part; Bestgen et al. 2011).

Early life stages of razorback sucker have been captured at most locations sampled in the middle Green River since sampling began in 1992 (Figure 3), but most fish were captured at relatively few localities including the Escalante reach (e.g., Cliff Creek) and Ouray reach (Greasewood Corral, Old Charley Wash) in the period 1993–1999; large numbers occasionally were captured near the Stewart Lake inlet or outlet. Capture locations were more widespread in the middle Green River from 2000–2010, reflecting higher abundance and broader distribution of both larvae and sampling effort (Bestgen et al. 2002; Bestgen et al. 2011). We also note that a razorback sucker larva was captured on 2 July 2000 in the lower Yampa River, and another three (and two more of slightly uncertain taxonomic identity) were captured from 28 June to 4 July 2008 during drift net sampling that targeted Colorado pikeminnow larvae. Those small sucker larvae (9–13 mm TL) suggested relatively late spawning in a location where razorback sucker was not common in recent years (KRB, unpublished data).

In the middle Green River, a set of sites that are regularly available for sampling even when flows are extremely high or low are sampled each year. These include Cliff Creek, Stewart Lake Outlet, Old Charley Wash inlet or outlet, and Greasewood Corral. Those sites reliably provide first-capture-of-the-year information across the longitudinal extent of the middle Green River, especially at Cliff Creek, which is just downstream of known spawning areas (Hedrick et al. 2009; 2010; Bestgen et al. 2011). That site is especially important given the emphasis on using real-time information to dictate timing of annual releases from Flaming Gorge Dam, which are used to provide access of larvae to floodplain wetlands. Several other sites are added annually based on availability of habitat including Baeser Wash, Walker Hollow, and other sites near or in Ouray National Wildlife Refuge. Those sites provide additional information on spawning success of adult razorback suckers and may also indicate presence and abundance of larvae in the vicinity of floodplain wetland breaches. That information may be critical to understanding efficacy of various wetland types to entrain razorback sucker larvae to enhance recruitment, information which is in turn important in experiments to understand timing and duration of flow events needed to effect recruitment of razorback suckers in floodplain wetlands in the middle Green River. Sampling in or near entrances to floodplain wetlands is a key component of the Larval Trigger Study Plan (LaGory et al. 2012), which describes a sampling program and approach to evaluate efficacy of using first appearance of sucker larvae to initiate higher flow releases in spring-time from Flaming Gorge Dam.

We recommend continuation of the existing program of light trap sampling for early life history stage of razorback sucker in the middle Green River. Especially important is early monitoring of upstream sites at Cliff Creek and Stewart Lake to ascertain presence of native sucker larvae in the middle Green River to trigger spring flow releases from Flaming Gorge Dam in spring. We are confident that those sites and others will provide that information accurately. We base those assertions on recaptures of marked razorback sucker larvae released in the Green River in differing batch sizes and under different flow regimes (Hedrick et al. 2009; 2010; Bestgen et al. 2011). Even with relatively small batches of larvae (2004), or under relatively high flows (2005), sampling with light traps at various sentinel stations detected marked larvae within 18 to 48 hours of release, and as much as 90 km downstream from release sites. Nevertheless, because of the importance of detection of reproduction by native suckers with light trapping techniques currently in use to accurately trigger flow releases, we outline below a small, low-cost study with several components as an additional data need to ensure adequate detection of larvae in the Green River (see "*Light trap sampling efficiency and colonization studies*").

Sampling at sentinel stations in addition to other more irregularly sampled sites will continue to provide information on reproductive success of adult razorback suckers in the middle Green River. Those adults, which were almost certainly all stocked fish, have successfully reproduced in the middle Green River likely beginning in the late 1990's. This assertion was based on two pieces of information. First, sampling detected stocked hatchery fish in ripe condition on or near spawning areas, with the few remaining wild fish, during the period 1996–1999 (Bestgen et al. 2002). Increasing abundance trends indicated by captures of razorback sucker larvae, especially since about 2004 and after the time when most wild razorback sucker adults were thought extirpated, also provides evidence that light trap sampling can be effective to monitor adult reproductive success.

Sampling in other areas, such as the Yampa River at the mouth of Yampa Canyon, also occurs in the Green River Basin in conjunction with drift net sampling for Colorado pikeminnow larvae (another element of Upper Colorado River Endangered Fishes Recovery Program Project 22f, sampling to document reproduction by Colorado pikeminnow and razorback sucker). That area was an historical site for razorback sucker reproduction, and in fact, was the first such spawning site described in the Upper Colorado River Basin (McAda and Wydoski 1980, Bestgen 1990). Sampling in project 22f has detected occasional reproduction by razorback sucker in the Yampa River based on captures of larvae, even though sampling is relatively late in the season, usually beginning in mid-June, compared to mid-May in the middle Green River, Utah. Increased frequency of occurrence of razorback sucker larvae in those Yampa River samples, presence of larvae or juveniles in seine samples collected just downstream in Whirlpool Canyon or Island-Rainbow Park (Recovery Program Project 115), or indication of increased abundance of adult razorback suckers (based on non-native fish removal sampling) in the Yampa River, may signal the need for additional monitoring for larvae earlier in the year. However, until such information is in hand, we recommend only a modest additional sampling effort for larvae in or near the mouth of Yampa Canyon at this time.

During all sampling programs that occur in mainstem rivers, careful attention should be given to collection and identification of juvenile suckers, with an eye towards capturing razorback suckers. Although rare, and especially in the main channel, juvenile razorback suckers about 30 mm TL and about 100 mm TL have been captured in backwaters or the main channel of the lower Green River by seining or electrofishing, respectively (Gutermuth et al. 1994; P. Badame pers. comm., detailed in Bestgen et al. 2011). Care should be taken when identifying such specimens and minimally, high quality and close-up digital photographs should be taken that include the lateral view of the body as well as the ventral view of the mouth. Any sampling mortalities should be preserved immediately, preferably in 100% ethanol, so that additional verification and other analyses (e.g., otolith increment counts) can be conducted.

Additional sampling of early life history stages of suckers was proposed in the Larval Trigger Study Plan (LaGory et al. 2012) and we offer some guidelines to be used for razorback sucker sampling in that program. Most early life stage sampling will occur under existing study 22f, and additional studies by the U. S. Fish and Wildlife Service and Utah Division of Wildlife Resources (each of Vernal, Utah), and will involve sampling in or near the mouths of up to eight target wetland breaches per year during the flow connection period (late May and June), as well as in the immediate post-connection period (late June or July, depending on the flow level and connection duration). The early sampling will attempt to document the entrainment and early survival of larvae in wetlands as a first step to evaluate use of the Larval Trigger for flow releases. That sampling will begin after first occurrence of larvae is documented downstream of spawning areas, perhaps at Cliff Creek, during the standard sampling effort conducted under Project 22f.

Sampling in the post-connection period is designed to document survival of larvae after connections with the river cease. This sampling should be conducted in all target wetlands identified for sampling that year, not just the ones where larvae were found in or near breaches. That is recommended because larvae could be missed with light trap sampling during high water river-floodplain connection times. Larval presence information in wetlands in the post-connection period will then guide which backwaters and wetlands will be sampled in autumn and then again in spring the following year. Such interval-based sampling will allow assessment of survival in the critical summer to autumn and overwinter periods. Wetland selection and timing of sampling each year will be guided by anticipated flow level and duration; wetlands with lower breach elevations (e.g., more frequent inundation) will be sampled in many or most years and those with higher breach elevations only during higher flow years. Sampling in some of those wetlands in the post-connection period was conducted in autumn 2011, and wild-produced (not hatchery origin) juvenile razorback suckers were detected in Wyasket Lake and Leota wetlands (pers. comm., A. Webber, USFWS, Vernal, Utah).

Post-connection summer sampling should employ a variety of gears including light traps to document larvae presence. However, active sampling gear such as seines, push-nets, or even small trawls or plankton samplers that can be hand-retrieved or towed or used from a canoe or boat are other potential sampling techniques. All effort (light trap hours, seine haul number and area, volumetric measures of towed gears) should be carefully documented to aid in making comparisons of catch per unit of sampling effort from year to year and among wetlands, as absolute estimates of abundance will be difficult unless marked fish are used. Based on our knowledge of light trap sampling, minimal levels of effort per wetland to consider are two or three nights of sampling spaced over a 2–3 week period with a minimum of ten (perhaps as many as 20) light traps, supplemented with other gears as conditions permit. Additional effort should be considered, especially in large wetlands, because the effective radius of light traps may be relatively small (e.g., about 3 m, Falke et al. 2010, see below for additional experimental evaluation of light trap effective radius). Sampling locations for all gear types should minimally include the wetland area adjacent to river breach and include nearshore and offshore areas.

Post-connection autumn (late September, October) and subsequent spring (March–April) sampling will require different gear types (see next paragraph) because of the larger juvenile (age-0) and other razorback sucker that may be present from other year-classes. Selection of wetlands to sample in autumn should be dictated, in part, by presence of larvae in the post-connection sampling efforts, by the likelihood that larvae may or may not have been detected in summer, and by wetland size. If larvae were detected in summer, the wetland should be sampled in both the autumn and subsequent spring periods. If sampling effort to detect post-connection larvae in summer was low, or if similar and nearby wetlands supported larvae in summer, the target wetland should be sampled in autumn and in spring. If efforts in summer were deemed insufficient to detect larvae in very large wetlands, additional autumn sampling is warranted. In all cases, presence of juveniles in autumn warrants subsequent sampling in spring to document if overwinter survival occurred.

Sampling in wetlands in 2011 successfully employed trammel nets and trap or fyke nets and those gears are recommended for future sampling. Mesh sizes for such gear should be sufficient to capture juvenile razorback suckers that are 75 to 125 mm TL, the size range of juvenile razorback suckers previously captured in floodplain wetlands in autumn (Modde 1996;

1997). A minimum of three sampling events per wetland in autumn and spring, including overnight trap or fyke net sets, should be employed. Trammel nets should be used, but judiciously, and perhaps with only short soak times (2 hr), given the potential presence of other endangered fishes and mortalities that may occur with longer sets. If conditions permit and efficiency is potentially high, other sampling gears including electrofishing or seining could also be used.

Biological sampling in target wetlands should be accompanied by sampling for physico-chemical parameters as well. This is important because documented fish kills in floodplain wetlands have been associated with poor water quality conditions in both summer and winter. Minimally, measurements should include dissolved oxygen and water temperature, but other parameters including pH, salinity and conductivity measurements may also be useful, and each should be collected several times per day and year-round. Some wetlands are also known to have elevated levels of selenium (e.g., Stewart Lake), which is being monitored sporadically through a different U. S. Fish and Wildlife Service program. It may also be prudent to collect some minimal level of information on the size and depth of the wetland habitats available. It may be possible to monitor this relatively easily if staff-gauge depth measurements at known elevations are linked with bathymetric maps of wetlands, if those data still accurately portray the physical characteristics of the area (e.g., minimal sedimentation over time). These types of data, when collected over several years, should yield insights into limiting parameters during various flow (spring peak vs. baseflow) and climatic (warm summers, snowy winters which may induce hypoxia) conditions.

Recognizing that equipment to continuously monitor floodplain wetlands is expensive and that all wetlands in the middle Green River may not be suitable in all years to entrain and support early life or older stages of razorback suckers, some areas should be given priority. Selection of wetlands for monitoring should first focus on those that have supported early life stages of razorback suckers in the past, and those with relatively low-elevation breaches that flood relatively frequently (e.g., various Leota wetlands, Old Charley Wash, Stewart Lake; Bestgen et al. 2011). Other locations that flood less frequently but have supported razorback suckers in the past (e.g., Wyasket Lake, 2011), should also be monitored when fish are known to inhabit the system to assist with interpretation of patterns of growth and survival. Other wetlands should be monitored as equipment is available and appropriate conditions warrant (e.g., known entrainment of larvae, presence of other life stages). More comprehensive bathymetric assessments and water quality monitoring in wetlands will assist the Program to define which wetlands and wetland types (flow-through vs. single-breach) should receive priority for razorback sucker recovery, which ones are in need of modification to be better suited for such (e.g., deepening Thunder Ranch), and which ones are not likely to be useful to support razorback suckers at all.

As with all sampling programs, the life stages, sampling effort, timing, gear types used, and many other factors should be flexible and reflect the best information available at the time. This will require annual evaluation and periodic summaries of information to ensure the best information is collected and utilized in a timely fashion.

Lower Green River.—Efforts to detect early life stages of razorback sucker were less frequent in the lower Green River than the middle Green River since sampling began in the early 1990's. Sampling was conducted from 1993–1999, and was re-initiated in 2009 and continues through 2011, with plans to continue sampling into the future (Figure 3). Earlier sampling in the lower Green River occurred from Green River State Park, Utah (RM 120) downstream to Holeman Canyon (RM 28) which is consistent with the area designated for sampling that started again in 2008 (Bestgen et al, 2002; Bestgen et al. 2011). The broader distribution of larvae noted since sampling was reinitiated in 2008, compared to the earlier period, is likely a combined effect of higher abundance and broader distribution of larvae and broader sampling effort. Three specific sampling areas within the reach were chosen for more recent sampling based on success with past sampling and include the Green River Valley area near RM 120, the San Rafael River confluence area (RM 97) and Millard Canyon (RM 33.5). Those sites were attractive for sampling based on presence of off-channel habitats such as tributary streams, flooded washes, or backwaters but additional sites are sampled as available.

We recommend continued sampling of those areas as sentinel sites, similar to those used in the middle Green River. Because distribution and abundance of razorback sucker larvae are less well known in the lower Green River than in the middle Green, we encourage continued additional opportunistic sampling to add to information. Flexibility to change or add sites as needed is important because discharge, accessibility, and habitat conditions may change at each site. We especially encourage additional sampling effort in or just downstream of the confluence of the San Rafael River. This area was noted as a concentration area for larvae through 1999 (Bestgen et al. 2002; 2011). Additional sampling may be helpful to understand if the San Rafael River inflow area is a collection point for larvae produced upstream or if adult razorback suckers are spawning in the tributary itself. It is noteworthy that sampling in the San Rafael River confluence area produced some of the largest razorback sucker larvae captured from 1993–1999. Better information on distribution and abundance of larvae may assist with understanding reasons for the occasional larger age-0 or juvenile-sized razorback suckers found in that reach (Gutermuth et al. 1994, three 100+ mm fish captured in 2008, P. Badame pers. comm., Utah Division of Wildlife Resources).

Colorado River Basin.—Adult razorback suckers have also been heavily stocked in the Gunnison and Colorado rivers, Utah and Colorado, beginning in 1994 (Burdick 2003; Osmundson and Seal 2009; Zelasko et al. 2011) but reproductive success has apparently been limited. Seine sampling from 2002–2007 in the lower 57 miles of the Gunnison River detected presence of nine positively identified razorback sucker larvae and 33 larvae identified as “razorback sucker?” (larvae whose taxonomic identity is slightly uncertain) among over 66,000 fishes examined from 1,032 samples (Osmundson and Seal 2009). Razorback sucker larvae were captured from just downstream of Delta, Colorado, downstream to near the confluence with the Colorado River.

Sampling in the Colorado River from 2004–2007 from upstream near Government Highline Canal downstream to Westwater, Utah, yielded 23 positively identified razorback sucker larvae and 1 larva identified as “razorback sucker?” among 26,000 fishes examined from 670 samples. Razorback sucker larvae were widespread in samples in spite of low abundance (see Osmundson and Seal 2009 for details). Abundance of adult fish may be low; an estimated 1,066 adult razorback suckers > 400 mm TL inhabited the Colorado River in 2005 in spite of stocking nearly 80,000 larger juvenile and adult fish through 2005 (Osmundson and Seal 2009, Zelasko et al. 2009; 2011). Ripe females, which may indicate locations of spawning adults, were distributed over a broad area from Fruita, Colorado, downstream to Moab, Utah. Low abundance of adults combined with widespread distribution may explain the widespread pattern of razorback sucker larvae.

As was suggested by Ryden et al. (2011, Colorado-Gunnison River sampling scope of work, Recovery Program project 163), we support continued sampling of razorback sucker larvae in the Gunnison and Colorado rivers. Sampling needs to be widespread based on broad distribution of running ripe females and larvae detected in an earlier study (Osmundson and Seal 2009) and should include light traps in appropriate locations including gravel pit ponds (e.g. Maggio Pond). Since samples and large numbers of all sucker larvae are easy to obtain but relatively expensive to identify, perhaps some stratification of sampling or samples could occur to obtain broad-based information. However, the first couple years of effort (e.g., 2012–2013) should not stratify or reduce sampling so that a more recent and intensive view of reproduction can be obtained. After that, more targeted sampling could be considered and sentinel sites established if larvae occur reliably in one or more locations.

Because larvae are relatively rare in the Gunnison-Colorado River system, we do not make recommendations for sampling of juveniles at this time. When larvae are relatively more abundant, and perhaps on par with capture rates observed in the Green River, more intensive sampling for juveniles will then be warranted. In the meantime, investigators should be on the lookout for juvenile razorback suckers captured in other sampling programs including Colorado pikeminnow abundance estimation, smallmouth bass removal, three-species and native fish community monitoring in the Gunnison and Colorado rivers, and all other sampling projects.

Additional early life history data needs

Light trap sampling efficiency and colonization studies.—In spite of ability to detect razorback sucker larvae from small releases of marked hatchery reared individuals (Hedrick et al. 2009; Bestgen et al 2011), and the apparent success of light traps to capture razorback sucker larvae, we feel it important to better understand probabilities of detection and colonization of larvae in low-velocity shoreline habitats and especially in large floodplain wetlands of the Green River. Thus, we describe several experiments that could add to that information.

The first experiment would be lab-based and assess attraction distances of larvae and capture efficiency of light traps in a controlled setting. Sampling efficiency of light traps is thought to be high (Snyder and Meismer 1997) but larvae and light traps in those experiments were in close proximity in small circular tanks in the lab, and did not incorporate features of the natural environment such as would be present in Green River backwaters. Thus, to better assess attraction distance, larvae could be placed in linear troughs at varying distances (1–4 m or more) from light traps. With interior room lights off and light trap lights on, small numbers of larvae of different life stages released at various distances from the trap and captured over a set time (e.g., 2–4 hours) would allow estimation of detection (present or not) and sampling efficiency (% captured). Turbidity could also be included as a test variable, since turbidity levels vary in Green River backwaters during spring light trap sampling. Such tests could also be conducted in ponds that simulate Green River backwaters, which could be used to determine attraction distances at even greater lengths than are possible with relatively short and restricted troughs.

A field test of attraction and capture efficiency of razorback sucker larvae would follow. This second experiment would be relatively easy to accomplish if a hatchery could provide relatively small quantities of larvae for marking, as has been done in prior experiments (Hedrick et al. 2009), and would inform aspects of sampling effort and spatial coverage in a natural setting to accurately monitor larvae presence with an appropriate level of effort. The assessment would proceed in two parts. First, we recommend assessing capture efficiency of larvae released within backwaters sampled with standard light trap gear. We propose a staged release of batches of larvae over a three-day period. On the first day, a small number of tetracycline-marked larvae (n = 10?) would be released in the morning, with subsequent light trap sampling in the evening and overnight. Traps would be emptied the following day, and a larger number of similarly marked larvae would be released (n = 100), followed by subsequent sampling. The final day would use a larger batch of released larvae (n = 500) followed by similar sampling effort. Such an effort, completed in 4-6 different backwaters, with similar-age fish but different backwater size, would allow for estimation of sampling detection probabilities by trap (each trap as a sampling unit) and by backwater night (traps combined) over a range of habitat types, and importantly, would allow for estimation of density effects of larvae and backwater size on capture success.

The standard number of traps set in a backwater typically depends on backwater size, with up to 5–10 traps set in each. Traps are set far enough apart such that the halo of light from each does not overlap with others. In large backwaters, larvae and traps would be placed in about the same proximity. A main complication would be availability of similarly aged fish over a long enough period to complete the experiments, because the number of light traps available is limited enough to not allow for simultaneous sampling in 4–6 backwaters. Sampling could occur prior to known wild razorback sucker larvae occurrence, which may negate the need to use marked fish, but use of unmarked fish is not a preferred technique. Sampling before presence of wild fish would not then interfere with or dilute other sampling efforts, and is possible because hatchery-produced larvae are typically available before wild larvae are present.

A third complementary experiment conducted simultaneously would be to release relatively small batches of larvae that are differently marked (e.g., single and double-tetracycline-marked larvae, [n = 25,000–50,000]) into the Green River at or near spawning areas to further test dispersal times to backwaters. Some tests of this were conducted during the wetland entrainment study from 2004–2006 (Hedrick et al. 2009; 2010) with larger releases of larvae; smaller releases would be especially useful to test the efficacy of presently used sentinel sampling sites such as Cliff Creek and Stewart Lake outlet to attract larvae for light traps to detect. Light traps set in backwaters to test light trap efficiency would also be used to capture larvae released in the river. Batches of fish could be relatively small to provide an effective test of detection likelihood. Similar sampling at sites downstream would add to information regarding detectability of larvae and dispersal rates under different flows.

A fourth experiment would measure rates of colonization of larvae into floodplain wetlands once larvae are in the vicinity of the river-wetland interface. Entrainment of larvae into floodplain has been documented and is relatively easily accomplished in flow-through wetlands because relatively weak-swimming larvae are swept into such places by river currents (Hedrick et al. 2009; 2010). Colonization of single-breach wetlands by razorback sucker larvae may rely on relatively lower velocity pulses of water flowing into wetlands caused by daily river flow fluctuations of snowmelt or simply increases in river stage (described in Bestgen et al. 2011). It is also possible that larvae are able to colonize single breach floodplain wetlands by simply swimming into them once they are in the river-wetland interface. To answer questions regarding whether larvae are able to colonize such wetlands, experiments with marked larvae could be used. Assuming larvae are transported to river-wetland interfaces from spawning areas similar to that observed with marked fish releases (Hedrick et al. 2009; 2010), marked larvae could be released at the wetland interface and their dispersal rates into the wetland tracked with daily light trap sampling. Relatively small batches of larvae (n = 1,000?) would be released and traps (n = 10) would be set in a transect at increasing distances from the wetland mouth. Flow conditions into the wetland (incoming, exiting, stable) and river stage (rising, declining, stable) would be monitored via measurements at USGS gauges as well as with staff gauges placed in wetland mouths. Samples would be collected daily to document progress of larvae and wild-produced and released larvae would be differentiated by presence of a mark. Releases of single and double-marked larvae could be used to increase the number of trials to estimate colonization rates over the season and under different hydrologic conditions (increasing and decreasing flows). Sampling after breach connections cease could also be used to document later season survival of different batches of larvae. Such releases would have to be coordinated to not confound studies of natural colonization of wild larvae being conducted under the Larval Trigger study (LaGory et al. 2012)

Yet another aspect of sampling and analysis could be formal occupancy analysis of sampling data. This involves multiple samples collected from the same location (could be in concert with releases described above) and at several locations. The advantage is occupancy estimation analysis allows for estimation of probability of occurrence (here, larvae) in the face of imperfect detection information (i.e., the idea that larvae may not be detected even when present, with inherent methods to correct for that; see Falke et al. [2010] for more information) and is especially useful if number of larvae increases over time. This would be especially useful for Larval Trigger study assessments, which will rely on knowledge of occupancy of larvae in wetlands in late spring-early summer to guide sampling later in summer or autumn.

Taxonomic verification of larvae.—Identification of larvae with traditional morphological and pigment characteristics works well for most specimens captured in light traps (Snyder and Muth 2004). This is true because razorback sucker larvae hatch early and at a smaller size than other native suckers and have earlier (at a smaller size) development of key structures through the early life history stage. Compared to the commonest non-native, the white sucker *Catostomus commersonii*, larvae of razorback sucker have key morphological development event differences (gut coiling), transitions to various larval life stages (e.g., flexion to post-flexion mesolarvae) occurs at a smaller size, and pigmentation patterns are quite different than the aforementioned species. However, with the increase in abundance of non-native catostmids (e.g., white sucker) and increased hybridization rates among various native and non-native catostomids in some portions of the basin, verification of specimens determined to be of questionable taxonomic identity could be very useful. A previous study using allozymes documented that traditional techniques are very accurate to identify razorback suckers (pers. comm., D. Probstel, formerly of Colorado State University) since nearly all razorback suckers and those identified as questionable were in fact bona fide razorback suckers. More recently and using other techniques, a few fish identified as razorback suckers or razorback suckers where identity was slightly questionable ("razorback sucker?") were identified using genetic techniques as some other taxa (J. Wood, Pisces Molecular, Boulder, Colorado); sample size for study specimens and reference fish species was low, reference fish identity may have been questionable, and the number of molecular markers resolved was small so the results are still under review. Regardless, reliable genetic techniques based on samples of known razorback sucker specimens and those of other reference species to benchmark results, would verify efficacy of traditional taxonomy, determine whether traditional techniques are useful to identify hybrid specimens, and thus, could also be used to monitor hybridization rates among native and non-native taxa.

A substantial downside is the high startup costs to determine appropriate markers for species, and subsequent high costs per individual fish. Nevertheless, the information would be very valuable to guide non-native catostomid removal strategies, which is likely to become more important as more species and hybrids spread through the basin. Especially problematic is hybridization with various taxa and endangered razorback sucker.

Large juvenile and adult life stage data needs and use

Similar to early life history data, capture data gathered for adult razorback suckers has multiple uses, and will be particularly important for assessing Recovery Goals (USFWS 2002). Those goals are based on quantifiable population abundance levels, as well as metrics for population stability including survival and recruitment rates. Specifically, the U. S. Fish and Wildlife Service (2002) requires that each of the Upper and Lower Colorado River basins maintains two "genetically and demographically viable, self-sustaining populations" for a five-year period before downlisting the razorback sucker to threatened status. In the UCRB, one population is required for the Green River subbasin and the other is to occur in either the upper Colorado River subbasin or the San Juan River subbasin, and abundance of adults in each population is to exceed 5,800 individuals. Population stability and abundance levels must be sustained for another three years after downlisting as minimally sufficient conditions for delisting to occur. Each cooperative program includes multiple management strategies addressing habitat, instream flow needs, and nonnative species. However, without recruitment, protection of remnant adult populations and associated habitat would not be sufficient to prevent extirpation of razorback sucker. Therefore, the required self-sustaining populations can only be achieved with the aid of hatchery augmentation (U.S. Fish and Wildlife Service 2002), until sufficient recruitment is achieved and maintained.

Razorback suckers are typically found in the largest numbers in impoundments, reservoirs (Lake Mohave, Lake Powell inflow of San Juan River, Platania et al. 1991; recent captures, D. Elverud, Utah Division of Wildlife Resources, Moab), off-channel ponds (Grand Valley gravel pits prior to their extirpation, Bestgen 1990) or other still water habitat, as well as in streams (Razorback Bar sampling, Bestgen et al. 2002), and are often detected in largest numbers during the reproductive season. This was particularly evident in the middle Green River, when the few remaining wild razorback suckers present in the late 1990's could typically be found only during spring on or near shallow spawning areas or in or near off-channel habitats (Bestgen 1990; Bestgen et al. 2002). Those seasonal patterns may indicate that fish in other seasons are spread out, difficult to detect because of the habitat they use (e.g., water depths that do not allow efficient electrofishing), or both. Perhaps for those reasons, adult life stage razorback suckers have sometimes been described as difficult to sample. Support for relative difficulty of sampling comes from relatively low probabilities of capture for 1st year and subsequent life stage fish (Zelasko 2008; Zelasko et al. 2009; 2010; 2011), which were generally < 0.04 (median = 0.0385, 0.002–0.128). Razorback suckers in the San Juan River had higher capture probabilities (Bestgen et al. 2009), which may be indicative of the prevailing shallow habitat through much of that river, and depth-related differences in capture probabilities.

Life history parameter estimation based on tag-recapture data was useful to assess survival of stocked razorback suckers, and to refine stocking goals and procedures. Those studies found survival of fish recaptured in a relatively short time after stocking (9–12 months) was very low and was size-dependent, but after that survival was much higher and rates were similar to that for wild fish (Bestgen et al. 2002; Zelasko et al. 2010). A main conclusion of those analyses that were especially relevant for monitoring was that probabilities of capture (p's) were quite low, and much lower than was typically found for species like Colorado pikeminnow or razorback suckers in different systems (Bestgen et al. 2007, Zelasko et al. 2009, Bestgen et al. 2009) even though data were derived from some of the same sampling programs. For example, Colorado pikeminnow and many razorback sucker captures were from the same multi-pass, multi-year (e.g., robust-design, Pollock 1982; Pollock et al. 1990) sampling programs in the Green and Colorado rivers. Zelasko et al. (2010) identified low p's as a concern and we excerpt that here:

> "*Ultimately, increasing capture probability must become a priority if more precise parameter estimation is desired. In mark-recapture studies, one aims to capture the most individuals from a released cohort on the first occasion after initial marking (stocking), which equates to high recapture probability. Although this study improved on that aim compared to the previous analysis, data were still collected from a variety of sampling programs where effort was sometimes low after stocking substantial numbers of fish, and very few efforts specifically targeted stocked razorback suckers. In contrast, species-specific, Colorado pikeminnow abundance estimate sampling produced recapture probabilities ranging from 0.01 to 0.20 in the Green River subbasin, 2000–2003*

(Bestgen et al. 2007a) and 0.07 to 0.19 in the Colorado River subbasin, 1991–1994 (Osmundson and Burnham 1998). Future recapture probability estimations would be aided by more consistent sampling efforts targeted specifically at razorback suckers, particularly in years when other intensive sampling, for studies such as Colorado pikeminnow abundance estimation, is not occurring. Not only would recapture probabilities likely increase, but a uniform protocol would better meet the underlying assumption that recaptures are made within brief time periods relative to intervals between tagging. Additionally, remote PIT tag stations placed near known spawning areas would provide valuable encounter data with little effort. Recapture rates of razorback suckers stocked in the lower Colorado River, 2006–2008, were 9% or less with electrofishing and trammel netting, but increased to 39% when remote PIT-tag scanning was employed (Schooley et al. 2008). A capture-recapture study on Lost River suckers in Oregon estimated low recapture probabilities (0.02–0.15) when using only physical recaptures, but 0.91 or higher after employing a remote detection system (Hewitt et al. 2010). Furthermore, the increased encounters improved precision of parameter estimates to such a degree that CIs became negligible."

Thus, a main emphasis in this plan will be to suggest means to increase probabilities of capture for razorback suckers without unduly increasing sampling effort, fish handling, and costs associated with monitoring. Increased probabilities of capture will permit better estimation of population parameters such as abundance and survival rates, which will enable better decisions regarding conservation and recovery status.

Large juvenile and adult sampling and analysis

An extensive and intensive set of programs exists for large-bodied fish sampling in the mainstems of the Green River and its major tributaries, the White and Yampa rivers, as well as the mainstem Colorado River and lowermost Gunnison River. Those programs include non-native fish removal, native fish monitoring (e.g., 3-species sampling), but particularly, sampling to estimate demographic parameters for endangered Colorado pikeminnow. That latter sampling program follows a robust-design procedure (Pollock 1982; Pollock et al. 1990) where multiple sampling passes (2–6) are conducted within a year in spring, and sampling is conducted over consecutive years, usually three. Then sampling is suspended for two years so as not to stress fish and also because of monetary limitations. That robust-design sampling procedure allows for annual abundance estimates using tag-recapture data from within year sampling in closed capture-recapture models. Such data also permits survival estimation between years of sampling using open population model (no assumption of demographic closure of populations) Cormack-Jolly-Seber estimation procedures. It is also possible to obtain estimates of annual transition rates of fish between sampling reaches if the sampling design supports a multi-state model analysis. Additional sampling between primary three-year sampling blocks sometimes allows for estimation of population rates of change (λ) that determine if population abundance (not estimated directly) is increasing, decreasing, or staying the same by essentially comparing survival and recruitment rates.

Conveniently, population centers for razorback suckers in the Upper Colorado River Basin are mostly overlapped by sampling areas for Colorado pikeminnow. This is particularly true in the Green River Basin, because substantial captures of adult razorback suckers have occurred mainly in the mainstem Green River, with a few in the lower White River (e.g., 2011). Substantial numbers of razorback suckers do not occur in tributaries such as the San Rafael River, or the Green River upstream of the Yampa River, or the Yampa River itself, even though scattered individuals are occasionally found in those places. Instead, most captures have been in the mainstem Green River from downstream of Split Mountain boat ramp downstream to the confluence with the Colorado River. A similar situation exists in the Colorado River, where Colorado pikeminnow sampling areas broadly overlap with known distribution of stocked adult razorback suckers from just upstream of the confluence of the Gunnison and Colorado rivers downstream to the confluence with the Green River (Zelasko 2008; Osmundson and Seal 2009; Zelasko et al. 2010).

*Green River data, fish distribution, and macrohabitat use***.** —Because of broad overlap of sampling areas, monitoring population status of razorback suckers will borrow heavily from sampling for Colorado pikeminnow, since many razorback suckers are captured during that sampling (Table 2, captures by sampling program, 2006–2008 data). For example, of 1,177 razorback sucker captures in the Green River Basin during 2006–2008, 1,079 (92%) were made during sampling for Colorado pikeminnow abundance estimation. Spring Colorado pikeminnow sampling is also conducted when razorback suckers may be most susceptible to capture, because they are in relatively shallow water near or at spawning areas.

Additional razorback suckers were captured in the 2006–2008 period in the Green River during humpback chub *Gila cypha* sampling (n = 11), and northern pike *Esox lucius* and smallmouth bass *Micropterus dolomieu* removal sampling (n = 16 and 71, respectively), but those contributions are small relative to captures during pikeminnow sampling; such additional sampling data could certainly be used to supplement data collected during Colorado pikeminnow sampling, depending on timing and the specific use of the data. Data for abundance estimation requires closely-spaced sampling occasions to fulfill assumptions of demographic and spatial closure, compared to data for apparent survival estimation which can use data collected in a less restricted fashion and over a broader time period, consistent with models that assume an open-population status. The 1,177 recaptures resulted from stocking events ranging over the period from 1996–2007; 1,062 of those (90%) were from stocking years 2004–2007 but recaptures of fish stocked in the 1990's indicated some longer-term survival (Table 3). We report further analyses of razorback sucker data gathered during Colorado pikeminnow sampling to evaluate its utility as the basis of a razorback sucker monitoring program.

We binned all the captures of larger juvenile and adult razorback sucker riverwide during 2006–2008 Colorado pikeminnow sampling into 10-mile increments to detect potential fish concentration areas. Distribution of razorback suckers captured in the Green River was uneven (Figure 4). Most (n = 771, 71%) recaptured razorback suckers obtained during Colorado pikeminnow sampling were from the lower Green River, with fewer from the middle Green River (n = 170, 16%), and Desolation-Gray Canyon (n = 138, 13%) reaches.

In the lower Green River (RM 0–120), razorback suckers were widely distributed but most common just downstream of the main stocking location at Green River, Utah, especially from RM 120–95. We further binned all captures in those uppermost 25 RM's into 1/10th mile increments to identify potential concentration areas and found four (Figure 5, locations 1–4). The most upstream location (1) was at RM 119.4–119.5 near Brown's Wash on river left and Saleratus Wash on river right, the second-most upstream site (2) was at RM 114.9, at or near Little Grand Wash on river left, the second-most downstream site (3) was at RM 105.5 near Salt Wash on river left, and the most downstream site (4) was at RM 101.6 and Anvil Bottom and associated with Dry Lake Wash on river right. We requested information from Mr. Paul Badame (formerly Utah Division of Wildlife Resources, Moab) about razorback sucker distribution in general in the lower Green River reach, and about specific locations 1–4 and observations of fish in them: those useful thoughts were received on 28 February 2012 are placed below to assist others in the future.

> *"I'll start with a few general observations and then hit each site. First thought, razorbacks are stocked at RM 120 so I would expect to see more concentration areas within the upper 25 miles of this reach, in addition this portion of the reach*

is dominated by riffles and cobble/gravel substrates while the most of the lower 95 miles dominated by runs and sand/silt substrates. All of the noted concentration areas are flooded washes/tribs and in some years they can be backed up for more than a mile; 2011 is a great example of this, we actually had to turn back in several flooded washes because we had gone up them over a mile and would simply run out of time.

Site 1: This is actually two flooded washes Brown's on river left and Saleratus on river right. Both are flooded washes with Brown's being significantly shorter when flooded and Saleratus having flow in the summer from agricultural irrigation returns. There are many riffles and shallow pool-tails that could be potential spawning bars but nowhere specific near this location was noted between 2006–08. During each of those years a few tuberculated individuals would be captured in the washes, but I don't recall fish running ripe. We have light trapped in both of these washes since 2009 and captured larval razorbacks in both. Brown's wash goes dry in the summer while Saleratus holds water all year...except maybe in 2007.

Site 2: This is Little Grand Wash, which floods 1/4 mile back in wetter years. This wash has a mix of large boulders and small gravel/course sand. I don't recall seeing tuberculated or ripe fish in there. We have also had some success light trapping larval razorbacks in this wash. This wash typically only holds water until Early July.

Site 3: This is Salt Wash, which floods about 1/2 mile back in wet years. It is wider and shallower than most others in the reach, also there's not much shoreline vegetation, combine that with its shallow depth and I would say it's the warmest flooded wash we sample. The substrate is mostly course sand and silt. We have caught a few tuberculated individuals there but no ripe or expressing fish. This wash typically holds water until Early July.

Site 4: This is Dry Lake Wash, which floods about 1/2 mile back in wet years. This wash is defined by high cut sand banks, deep water 2–3 meters, and large cottonwood snags in the water. The habitat is very complex and often holds water throughout the year. As in other sites we have captured tuberculated fish but no ripe ones that I recall. We also light trap in this wash and have had larval captures. This wash is located 2 miles downstream of a site we believe to be a spawning bar. The bar is located at RM 103.8; it's a side channel riffle with a small cobble substrate. This bar is the only site in the reach that we have consistently seen ripe and expressing razorbacks congregated in.

In terms of concentration areas, flooded washes are consistently where we find the largest numbers of razorback suckers. Most of the flooded washes look like little wetlands or flood plains with no flow, warmer clearer water, and high productivity. I've always thought the flooded washes in this reach fill a similar role as flood plains and flooded bottoms by taking in lots of course organic material and turning it into primary production, invertebrates, and ultimately fish. I should also note that the two shallower sites (2 & 3) both have large debris *fans at their mouths, which as the spring flows recede, block off a side channel which become large deep backwaters holding water all year."*

Observations of ripe fish in the reach are supported by database information (USFWS, Grand Junction database, summarized in part by Zelasko et al. 2010; 2011) for those same 2006–2008 razorback suckers captured during Colorado pikeminnow sampling in the lower Green River. Although only 48 fish were noted as ripe, 42 (87.5%) were from that same upper 25-mile reach. Similarly, 47 of 58 (81%) razorback suckers noted as tuberculate were from the same reach. Recall that the ripe fish captured between the two lower areas at RM 103.8 in 2008 (pers. comm., P. Badame, Utah Division of Wildlife Resources, Moab) were the same as previously reported in Bestgen et al. (2011); that location should be viewed as a known spawning area for razorback suckers per the above information.

An additional concentration area was in the vicinity of Mineral Bottom (RM 53.8) boat ramp from RM 60–40 in the lower Green River reach. Because no fish were ever stocked there (K. Zelasko, unpublished data), concentrations may be due to additional sampling or habitat features that attract fish.

We analyzed macrohabitat used by razorback suckers in lower Green River in each of two reaches, the upstream reach (RM 120–95) where fish were concentrated and many were in reproductive condition, and a downstream reach (RM 95–0) where fish were more spread out and few were in reproductive condition (Figure 6). Primary and secondary habitat types were taken from data entered on field sheets and input into the main database. Primary habitats included main channel or side channel, and secondary habitats within each primary one included backwater, eddy, embayment, island tip, riffle, run, shoreline, and tributary (a combination of perennial or intermittent stream mouths, and flooded mouths of washes or side canyons).

In the lower Green River, about equal numbers of fish were present in upstream and downstream reaches, but were more concentrated upstream due to the shorter reach length (25 vs 95 RM's). In each reach, most (>93%) primary macrohabitat used by razorback suckers was located in main channel, and in each reach and in increasing percentages, runs, shorelines, and tributary mouths were the dominant secondary habitat types.

Side channels were only infrequently used in the lower Green River, because they are rare (P. Badame, UDWR, Moab). Runs and shorelines were the most commonly used secondary macrohabitat type in side channels. Even though side channels are not often used, their importance should not be discounted given that a suspected spawning area at RM 103.8 occurs in a side channel, and that a side channel is a main feature of the middle Green River spawning area, Razorback Bar. Such areas should be considered for monitoring with passive PIT tag detector antennae, which may substantially boost numbers of recaptured fish in each area.

The importance of tributary habitat in downstream and upstream reaches (nearly 35 and 40% use, respectively) of the lower Green River is notable, because those habitat types are mostly unavailable in seasons other than spring when those places are inundated with snowmelt runoff flows. Investigators should target such habitats to increase capture rates of razorback suckers, especially in the upper reach of the lower Green River where more fish occur, both with active sampling gear, and perhaps, with passive gear such as hoop nets, which may be deployed and checked less frequently. Hoop nets deployed in low-velocity channel margin areas were an effective gear to sample razorback suckers in the middle Green River from 1996–1999, and captures made with those gears greatly supplemented fish captures compared to those made with just electrofishing, which occurred mainly over Razorback Bar (Bestgen et al. 2002). Such locations should also be evaluated for use of PIT tag detector antennae. In spite of their high initial cost, high potential detection rates of fish in concentration areas such as flooded tributary mouths may warrant their use, and in the longer-term, be cost effective compared to exclusive use of active sampling gear. Flooded tributaries are also typically low-velocity environments, which may be advantageous when deploying detection gear compared to the swifter flowing main or side channels.

In the Desolation-Gray Canyon reach, razorback sucker capture numbers were relatively low and were more uniformly distributed. The most upstream 10-mile section, which was just downstream of the Sand Wash boat ramp and the main access for stocking hatchery fish, was where the most fish were captured, but lack of concentration areas precluded more detailed mapping of such. Fish macrohabitat use in that area was dominated by main channel captures, with a smaller number of backwater and very few tributary (mainly Price River mouth) captures (Figure 7). That area is sampled mostly earlier in the year during Colorado pikeminnow abundance estimation (Table 5), so the importance of tributary habitat may be lower in that lower flow period.

In the middle Green River, razorback sucker captures were also relatively uniform and low relative to the lower Green River. Also similar to the Desolation-Gray Canyon reach, most fish were captured in the two most upstream reaches and just downstream of the Split Mountain Boat Ramp where most fish were stocked. We plotted the fine-scale distribution of razorback sucker captures in the uppermost 20 miles of the reach to examine distribution related to known spawning areas (Figure 8, locations 1 and 2). Concentrations were associated with or near spawning areas at Razorback Bar (#1, RM 311) and Escalante Bar (#2, RM 306.8); a third concentration at RM 313.8 was also noted (perhaps Cub Creek mouth). Each had a relatively low number of fish captures, likely because extensive sampling in the vicinity of spawning areas was discouraged. Only five fish were noted as ripe (n = 167 had no status recorded), and all those were from the upper reach, RM's 320–310, where spawning bars are located.

We also analyzed macrohabitat used by razorback suckers in middle Green River in each of two reaches, the upstream reach (RM 320–300) where fish were concentrated and the few reproductive fish were noted, and a downstream reach (RM 299.9–246.1) where fish were more spread out and the few fish in reproductive condition were only tuberculate but not ripe (Figure 9). In the middle Green River, Utah, fish were slightly more concentrated in the upper reach than downstream. In both reaches, most razorback suckers were captured in main channel primary habitat; only three fish were found in side channels in each reach. In the upper reach, main channel secondary habitat use was mainly shorelines (95%), but in the more downstream reach, more backwater and tributary secondary habitat was used, because it is more available there.

Green River abundance, survival, and capture probabilities. —In all three Green River reaches, 404 razorback suckers were captured in 2006, 285 in 2007, and 390 in 2008 (Table 4). Patterns of capture among passes varied among reaches and years. In the middle Green River in 2006, the most fish were captured on pass 1 and the fewest in pass 3, whereas an opposite pattern prevailed in 2007 and 2008. Not included in those capture numbers were three age-1 juveniles (119–120 mm TL) captured between river miles 18 and 44 in 2008 in the lower Green River (pers. comm., P. Badame).

In Desolation-Gray Canyon, the fewest fish occurred on the last pass in all years. In the lower Green River, the most fish were captured on pass 1 or 2, and the fewest always during pass 3, regardless of year. Pass 3 was generally associated with high or increasing flows in the lower and middle Green River reaches; sampling in Desolation-Gray Canyon occurred earlier which may be responsible for relatively low captures in those lower flow, but colder water, conditions (Table 5).

The 1,079 razorback sucker captures obtained during Colorado pikeminnow sampling from 2006-2008 were from 1,004 unique fish. Discounting multiple captures of individuals during the same sampling pass, 933 fish were recaptured only once, while 71 unique fish were recaptured 146 times (67 of which were recaptured twice and 4 were recaptured three times). Of those 1,004 individuals, 118 (11.8%) were ≥ 400 mm TL at stocking (an adult fish, USFWS 2002), but 374 (37.2%) were ≥ 400 mm TL upon recapture, indicating growth of some individuals between time of stocking and last recapture but relatively few were stocked as adults. Maximum change in length over the 2006–2008 period was 74 mm TL; that 350 mm TL individual was captured on the first sampling occasion in 2006 and was 424 mm TL when recaptured on the last sampling occasion in 2008.

Change in length data were somewhat suspect as 269 of 1,004 recaptured fish (27%) were apparently shorter than when released by an average of 9.6 mm TL (1 to 60 mm); that total and mean value did not include two fish where a recording error was assumed (e.g., 440 at release, 342 at recapture). In addition, 17 of the 1,004 recaptured fish (1.7%) did not have a length recorded at the time of stocking. Given the value of length and change in length data to inform size-dependent changes in capture or survival rates, to estimate growth rates between recapture intervals, or to estimate condition indices, and the considerable expense associated with culture of fish in hatcheries and recapture of stocked fish in the wild, we encourage additional precautions to obtain the most accurate length data possible. This might be in the form of more training, consistency checks, quality control in data recording, or other measures.

Below we use the 2006–2008 razorback sucker capture data to estimate recapture and survival rates, and abundance. We do this mainly to illuminate expectations for the utility of the razorback sucker monitoring data that will be gathered in future Colorado pikeminnow sampling efforts; the reader should exercise caution in interpreting the estimates provided because they are unreliable in most cases and instead focus mainly on the message that the type of data gathered in those three years, in and of itself, may be insufficient to adequately estimate demographic parameters of interest for razorback suckers in the Green River Basin, and perhaps the Colorado River, Colorado and Utah, as well. It is recognized that additional 3-year-segments of data (e.g., 2006–2008 and 2011–2013) will allow for more precise estimates of survival, because data are then potentially borrowed across all years to estimate probabilities of capture and survival. However, estimates of abundance and more importantly, their potential for bias and low precision, are not likely to improve without addition of sampling occasions, or more recaptures within occasions.

We fit a multi-state robust-design model to the 2006–2008 data, where states were the three sampling reaches: the upstream middle Green River, the intermediate Desolation-Gray Canyon reach, and the most downstream lower Green River reach, as defined per Bestgen et al. (2007; 2010). Models were built that included parameters to estimate transition rates between reaches among years; preliminary model runs demonstrated that no transitions occurred so transition estimates were set to zero. Further, no razorback suckers from Desolation Canyon were ever captured after first capture in 2006-2008, whether they were first stocked there, or were first recaptured there, and then seen again in a different year (i.e., none captured in 2006 were seen in 2007 nor 2008, and none captured in 2007 were seen in 2008) indicating very low survival, a noteworthy management finding. Thus, we set survival = 0 for that reach.

The final model we chose to interpret had the following parameters: survival rates for each of the lower and middle Green reaches (*n* = 2 total), capture probabilities for each reach and year (*n* = 9 total), and a length-dependent (slightly positive) effect for capture probability (*n* = 10 parameters). Derived parameters included abundance estimates for each reach and year (*n* = 9 total). More complicated models were fit that included various combinations of probabilities of capture by reach, year, and pass, and a length effect on survival rate, but those were rejected because one or more parameters yielded incomprehensible estimates (very small) or because parameters had even wider confidence limits than the already imprecise ones shown. Other models, including one with reach-specific survival rates and reach-specific capture rates without year effects had little or no support.

Survival rate of razorback suckers in the lower Green River (0.51, 95% confidence limits 0.32–0.70) was moderate (Table 6). That survival rate was likely influenced by a mix of fish that included just stocked ones that had low, often less than 10% first-year survival rates, and older fish which have higher survival rates (Zelasko et al. 2010, 2011). Survival rate of razorback suckers in the middle Green River (0.69, 95% confidence limits 0.08–0.98) was higher and may have reflected presence of a few additional older fish, but the estimate was likely biased and the confidence limits nearly spanned the possible range of 0 to 1, thus limiting inferences from that estimate. As mentioned earlier, absence of recaptures of any razorback suckers across years in the Desolation-Gray Canyon reach from 2006–2008 precluded estimation of survival rates, and the most parsimonious explanation is that few or no razorback suckers survived in that reach from one year to the next.

Low precision of survival rate estimates reflected the low probabilities of capture of fish in each reach, but particularly for the middle Green River, where capture rates ranged from 0.004–0.02 over the three-year period (Table 7). Capture rates for the lower Green River were slightly higher and more fish existed in that reach, so *p*'s were slightly higher at 0.01–0.07 over the 2006–2008 period. Those rates consisted of the following across-year captures: 2 fish captured in 2006 and 2007, 9 fish captured in 2006 and 2008, 11 fish captured in 2007 and 2008, and 1 fish captured in all three years. Capture rates exist for the Desolation-Gray Canyon reach only because within year recaptures were observed; those ranged from 0.008–0.04 and were slightly higher than those in the middle Green River but lower than those in the lower Green River. In the middle Green River, only five razorback suckers were recaptured across years: 3 from 2006 to 2007 and 2 from 2007 to 2008. Overall, capture rates for all reaches were highest in 2006 (0.02–0.07), very low in 2007 (0.008–0.01), and intermediate in 2008 (0.01–0.04). Capture rates for Colorado pikeminnow in the same reaches and years were also highest in 2006, but tended to be higher in 2007 than 2008 (Bestgen et al. 2010). River flow rates were low in 2007, which may have made sampling more difficult, and in at least one reach, effort (in terms of days of sampling) was relatively low.

Abundance estimates of razorback suckers varied dramatically across years in each reach (Table 8, abundance estimates). Abundance was highest in the lower Green River, ranging from nearly 1600 fish in 2006 to 5153 in 2007, and then declining to 2597 in 2008. Razorback sucker abundance was lowest in the Desolation-Gray Canyon reach of the Green River, ranging from nearly 474 fish in 2006 to 3011 in 2007, and then declining to 836 in 2008. Abundance was intermediate in the middle Green River reach, ranging from nearly 600 fish in 2006 to 3146 in 2007, and then declining to about 1200 in 2008.

Razorback sucker abundance estimates were relatively imprecise in all reaches and years. Abundance estimates for Green River razorback suckers in the period 2006–2008 were the most precise in the lower Green River, with CV's of 22–37% among years, and reflected the relatively higher capture rates in that reach. The confidence limits for abundance estimates in other reaches were very wide and the CV's ranged from 49 to 81%, which limits any meaningful inference to those estimates. However, relatively precise estimates of large-river fish abundance are possible. For example, estimates of Colorado pikeminnow in the middle Green River reach from 2000–2003 had CV's of 9–18% (Bestgen et al. 2007). The higher recapture probabilities

(0.05–0.12) and large numbers of recaptured fish were responsible for higher precision of estimates.

Relative imprecision of estimates, particularly for Desolation-Gray and Middle Green River reaches, was easy to understand given the lack of recaptures within sampling years, which are used to estimate abundance. Relatively more precise (but only marginally useful) abundance estimates from the lower Green River resulted from within-year recaptures of 18, 3, and 11 fish in 2006, 2007, and 2008, respectively. Within-year recaptures of razorback suckers in the Desolation-Gray Canyon reach was limited to 4 in 2006 and none in other years. Within-year recaptures of razorback suckers in the Middle Green River reach was limited to 2 in 2008 and none in other years. Estimates were possible in reaches and years without within-year recaptures only because data were borrowed across years to estimate reach-effects and because the length effect on captures was present across all reaches and years. Nevertheless, imprecision of estimates in those years without within-year recaptures was extreme and the estimates themselves are likely biased high.

We compared numbers of razorback suckers stocked the year prior to sampling for abundance estimates (from Zelasko et al. 2011) to abundance estimates the next year to see if patterns were evident. We used fish stocked the year prior to abundance estimates (rather than the same year) because most of those fish were stocked in summer or autumn and were available for recapture in the subsequent abundance estimation year in spring. In other words, fish stocked in summer or autumn in one year would thus not be available for capture for spring estimates conducted in the same year.

Abundance patterns in each reach did not necessarily match patterns of stocking for the prior year. In the lower Green River, numbers of fish stocked the year prior to estimates in each of 2006, 2007, and 2008 were relatively similar, but abundance increased by over 3X from 2006 to 2007, and then declined by about 50% in 2008. In Desolation-Gray Canyon, fish abundance was relatively low in 2006 because no razorback suckers were stocked there in prior years. Thus, most fish that resided there in 2006 were likely fish that dispersed from upstream reaches, based on movement rates and direction documented in the past (Zelasko et al. 2010, 2011). Higher 2007 estimates did reflect high numbers of fish stocked there in 2006 (n = 10,075), and are also likely the result of the very low p and the resultant biased (high) and imprecise estimate. The dramatic reduction in abundance in 2008 was consistent with high mortality rates of first-year stocked fish, the overall high mortality rate of razorback suckers in that reach in general, and absence of any fish stocked in 2007. Mixed patterns were also evident in the middle Green River, as abundance in 2006 and stocking numbers in 2005 were lowest, and abundance increased in 2007 consistent with higher stocking numbers in 2006. However, abundance declined in 2008 by about 65%, even though stocking numbers were highest in 2007.

The large swings in abundance of razorback suckers among years would not be expected in a stable population of wild fish. Even relatively small populations of wild razorback suckers in the middle Green River in the 1980’s and early 1990’s did not experience dramatic population swings, likely because of lack of recruitment and because the population was influenced mainly by mortality of aging fish. One explanation for large population swings is the low survival rates of fish just post-stocking, as amply demonstrated by Zelasko et al. (2010; 2011). Survival rates must have been very low in the 2007–2008 interval for population abundances to decline so dramatically in each reach. That relatively low flow year in 2007 may have had an adverse influence on survival rates; Bestgen et al. (2010) did not estimate annual survival rates for Colorado pikeminnow in the 2006–2008 period so no comparisons are possible. Another likely explanation for apparently large swings in abundance among years is that high abundances in 2007 may have been biased by very low recapture rates; capture rates biased low would have the effect of increasing population abundance because numbers of fish captured divided by an unrepresentative and low p would increase apparent abundance. It is important to remember that the abundance estimates presented here are for all life stages present; razorback sucker adults (> 400 mm TL) represented only 37% of fish captured in the 2006–2008 period so adult abundance is substantially lower than the estimates portray.

Colorado River data, fish distribution, and macrohabitat use. —The 2005 sampling for Colorado pikeminnow abundance estimation in the Colorado River also resulted in capture of relatively large numbers of razorback suckers (Osmundson and Seal 2009). Distribution of larger juvenile and adult razorback suckers in 2005 in the Colorado River was more concentrated upstream, where about 60% of those fish were from the Palisade-to-Westwater reach (Osmundson and Seal 2009, Figure 10). About 40% of razorback sucker captures were from the more downstream reach Cottonwood Wash to the Green River confluence but < 1% of razorback suckers were captured in the lowermost 46 miles from Potash to the Green River confluence. Low abundance of razorback suckers was also noted in Westwater Canyon (e.g., RM 125–120, in part) and in the most upstream 5-mile reach just below Price-Stubb diversion.

We did not plot distribution of reproductively active razorback suckers in the Colorado River because Osmundson and Seal (2009) plotted locations of running ripe female razorbacks. They indicated presence of a potential spawning area in the Colorado River near Loma, Colorado, at the downstream end of Skipper's Island at RM 154 in 2007.

Habitat use patterns by razorback suckers in the Colorado River were analyzed similarly to that for the Green River. The Colorado River was divided into two sections, one upstream of the head of Westwater Canyon (RM 185–125) and one downstream of there (RM 124.9–0); nearly equal numbers of fish were captured in each (Figure 11). In each reach, most (>78%) primary macrohabitat used by razorback suckers was located in main channel, but side channels were used more widely than in the Green River. In the lower reach, use of runs dominated, but backwaters, eddies, and tributary mouths constituted 49% of habitat where razorback suckers were captured. In the upper reach and in primary main channel habitat, the main secondary habitat used by razorback suckers was runs, but slackwater areas such as backwaters, eddies, and fewer tributary mouths also accounted for 37% of captures.

Side channel secondary habitat use in the lower Colorado River was also dominated by runs, but backwaters, eddies, and tributary mouths also received high use (36%); use of runs predominated in side channel primary habitat in the upper reach and use of backwaters, eddies, and tributary mouths amounted to only 12% of use, remembering that the number of fish captured in side channels was relatively low.

The importance of backwater, eddy, and tributary habitat in downstream and upstream reaches of the Colorado River, Colorado and Utah, is notable, because those habitat types are mostly unavailable in seasons other than spring when they are inundated with snowmelt runoff flows. Such high flow inundated areas are also sampled with high success for Colorado pikeminnow, often by blocking with trammel nets which capture pikeminnow when attempting to leave the habitat. Investigators should continue to target such habitats to increase capture rates of razorback suckers, and perhaps, also with passive gear such as hoop nets, which may be deployed and checked less frequently. Hoop nets deployed in low-velocity channel margin areas were an effective gear to sample razorback suckers in the middle Green River from 1996–1999 (Bestgen et al. 2002), as previously explained. If high concentration areas could be identified, or the presumed spawning area at Skipper's Island continues to be used, installation of a PIT tag detector array at such locations should be considered to increase capture probabilities.

Colorado River abundance, survival, and capture probabilities.—During 2005, sampling resulted in capture of 426 stocked razorback suckers in the Colorado River, of which 145 (34%) were adults (≥ 400 mm TL); a total of 12 adult individuals were recaptured among the five sampling passes. Based on adult capture data, model M_t + length (a model that allows capture probabilities to vary with sampling pass and with fish length) produced an abundance estimate of 1,066 fish, although precision was low (95% confidence limits = 377–3,703) and likely a result of relatively low probabilities of capture of 0.018 to 0.057 for fish that averaged 437 mm TL (Osmundson and Seal 2009). Using the same model, an abundance estimate of 2,137 was derived for all sizes of razorback suckers present (SE = 348, CV = 16%; 95% confidence interval 1,576–2,958).

Zelasko et al. (2009; 2010; 2011) found similarly low capture probabilities for razorback suckers in both the Green and Colorado River systems, with highest capture rates coming from the largest, most easily captured fish. These data demonstrate that Colorado pikeminnow sampling can produce sufficient captures of fish to estimate population abundance but estimates are imprecise and of limited value. Additional data from multiple years of sampling, and additional captures from other sampling programs will likely increase precision of those estimates because data can be borrowed across years to increase efficiency of the estimators. The point here is that abundance and survival estimates can be produced but managers will need to decide how much increased recapture effort is needed to satisfy the need for increased precision and potentially reduced bias of the estimates.

Simulations to guide sampling effort and group stocking size

Simulation methods. —To assist with determining the levels of sampling needed to increase precision and reduce bias of estimators, we simulated various sampling program data in program MARK (White and Burnham 1999). Recall that the overall goal of the monitoring sampling is to obtain relatively accurate and unbiased estimates of abundance, survival, or other parameters of interest, and to estimate those parameters with a relatively high level of precision. Those goals can be accomplished mainly by increasing recapture rates of marked fish, and the simulations can be used to predict effects of variations in sampling programs on accuracy and precision of estimates.

Simulations were accomplished by varying probabilities of capture, number of sampling occasions, and population size. Increasing probabilities of capture and sampling occasions both result in increased recapture rates and precision of estimators. This can be accomplished in reality by increasing sampling efficiency or effort per pass, using more or different gears to increase number of recaptured fish per pass, or adding sampling passes. We used probabilities of capture that varied from 0.02 to 0.20; the lower end of the range reflects capture probabilities similar to those estimated in most Colorado and Green River studies, and *p*'s as high as 0.20 were similar to those estimated for razorback suckers in the more intensively sampled and smaller San Juan River, New Mexico and Utah (0.06 to 0.36; Bestgen et al. 2009). We varied the number of sampling passes from 3–4 in our simulations, because those numbers reflect realized or potential levels of effort implemented in the past during Colorado pikeminnow sampling (Bestgen et al. 2007; 2010). We also varied number of sampling passes because that is a strategy that can be implemented to vary effort and because the monetary effects of changing numbers of sampling passes can be more easily translated than the more abstract effect of increasing *p*'s. In general, increasing number of sampling passes has the effect of increasing recapture rates because given a certain recapture rate per pass, the number of fish recaptures will increase as sampling passes are increased. Varying population size in simulations also has the effect of increasing precision, because even though probabilities of capture may be held constant, more recaptures result from a larger population. Thus, given a static rate of probabilities of capture, populations of a larger size are estimated with higher precision. We used population sizes that were either 1,000, 2,000 (2,500 for abundance estimation simulations), or 5,000, because the lower population sizes reflect an estimate of abundance in the Colorado River in 2005 and estimates in some reaches and years in the Green River (see previous estimates) and the upper limit is similar to Recovery Goal levels (n = 5,800) for certain river reaches of the Upper Colorado River Basin (e.g., USFWS 2002, Osmundson and Seal 2009).

The main estimation parameters of interest in simulations were survival rate and abundance. Ability to detect declines in survival rates is important because Recovery Goals require population stability (recruitment rates similar to mortality rates over the long term, in addition to absolute abundance) to downlist or delist razorback sucker. Ability to detect a true change in rates is often frustrated by imprecise estimates, because changes in trends over time are not obvious. Thus, detecting a true decline in survival rate with a high probability would be important to alert managers to conditions that may not be favoring survival of adult razorback suckers.

Two types of survival simulations were created. The first simulated the ability of a sampling program with certain qualities to correctly detect a true decline in survival rates of 10 or 20% (from 80 to 70% and from 80 to 60%, respectively). In other words, the simulation data generated reflected a real decline in survival at the specified level (1,000 simulations each) and were used to determine what percentage of time a sampling program with specific sampling characteristics (% change, probability of capture, # sampling passes) was able to correctly detect that trend. We started at the 80% survival level, because that is similar to annual survival rates in the Upper Colorado River Basin of both wild razorback suckers and stocked razorback suckers after their first-interval (usually a year) in the river post-stocking (Bestgen et al. 2002, Zelasko et al. 2010; 2011). The 10 and 20% decline levels seemed reasonable as benchmarks for when managers ought to be concerned with a change in population status, as a 15% decline in annual survival rate of adult Colorado pikeminnow in the Green River Basin over a four-year period resulted in a population abundance level decline of about 40% from 2000–2003 (Bestgen et al. 2007). We used a three-year period over which to detect such a decline because that is consistent with the duration of sampling programs for robust-design sampling for Colorado pikeminnow, the program under which much of the data that may be available for analysis would be collected. Three years also seemed like a reasonable duration after which managers would want to take action to reverse such a decline. We used a simulated population size of 5,000 individuals, to reflect the approximate size of populations desired in Recovery Goals, and used constant probabilities of annual capture of 0.02, 0.05, 0.10, and 0.20, because those seemed within the bounds of reality, albeit somewhat high for the Green and Colorado rivers, for present sampling programs.

A second set of simulations was created to understand effects of various combinations of differing capture probabilities (same as before, [0.02, 0.05, 0.10, 0.20]) and numbers of fish (1,000, 2,000, and 5,000) stocked annually over a three-year period on ability to correctly detect differences in survival rates of 10 and 20% among groups of fish (same rate differences as before); 1,000 simulations were run for each combination. These simulations depict ability of managers to correctly detect differences in survival of groups of fish, relative to a benchmark (here 80%), stocked into a system under different recapture rates and stocking numbers, criteria that can presumably be manipulated. For example, if the goal was to detect differences in survival rates of batches of fish stocked in the same river reach from different hatcheries, or fish that were fed varying diets, or that had different rearing or acclimation histories, release sizes and recapture probabilities could be determined such that a relatively high probability of detecting a true difference of 10 or 20% survival could be achieved.

Akaike's Information Criterion (AIC; Akaike 1973; Burnham and Anderson 2002) was used to determine which model was correctly chosen among the 1,000 simulated pairs of data. For example, AIC scores evaluated if the time-varying survival model (e.g., the one estimating a true declining population) fit the simulated data better than a time-constant survival model (one where no change in survival was detected even though a decline had occurred) for that particular simulation pair. In other words, AIC selection of the true time-varying survival model over the constant survival model for that simulation would indicate that the specified decline in survival (10 or 20%) was correctly identified. The number of times the correct model was chosen was then summed over the 1,000 simulation pairs that were evaluated. The same procedure was used to determine how often the true difference in survival among two groups of fish was detected, given various simulation parameters of differing probabilities of capture, different levels of survival among the groups (10 or 20%), and differing release sizes, over a 3-year period and 1,000 pairs of simulated of data. Again, the number of times AIC correctly chose the correct varying-group survival model compared to the constant-group survival model over the 1,000 simulated pairs of data is reflected as the % of simulations where the true model was chosen.

Survival simulation results.—Simulation results showed that a change in survival rate of 10% (success rate ranged from 13.2 to 81%) was more difficult to detect than a change in survival rate of 20% (success rate of 37.5–100%, Table 9); this is the expected result because small changes in parameters are more difficult to detect than large ones, under the assumption given here, that capture probabilities and all other conditions are the same. Also evident is the large effect of capture probability on the percentage of simulations where the correct model of population survival rate decline was chosen. If managers wanted to correctly detect a decline in survival rate of 10% over three years at the relatively high rate (e.g., > 80%) in a hypothetical population of 5,000 razorback suckers, capture probabilities would have to be at least 0.20, a rate much higher than is presently occurring in most stream reaches (see previous analyses in this report; Zelasko 2008; Osmundson and Seal 2009; Zelasko et al. 2010; 2011, but see Bestgen et al. 2009). For capture probabilities of 0.02, 0.05, and 0.10, the correct model of declining survival rate would be detected less than 50% of the time, and sometimes much less; recapture probabilities in the Upper Colorado River basin are often in the range of 0.02–0.05, so the likelihood of being able to correctly detect a declining survival rate of 10% is often < 25%. This is not a desirable attribute of a high-quality monitoring program.

Correct model selection occurred at higher rates when the decline in survival rate was increased to 20%, and especially so, when recapture probabilities were 0.05 or higher. The simulation results could be considered in the framework of a power analysis, whereby the statistical power to correctly detect a true trend is equal to the percent of the simulations that are correctly chosen. On the other hand, the proportion incorrectly chosen is the chance that the incorrect model will be chosen even when the state described in the simulation, here a decline in survival rate, is true. For example, in Table 9, where detection of a 20% change is survival rate is desired and recapture probability is 0.05, data would incorrectly identify that no change in survival occurred 22% of the time (100% – 78% = 22%); thus, managers would have about a 1 in 5 chance of not detecting the true decline in survival rate with the given sampling and population characteristics in place. It should also be noted that these are likely the most optimistic levels of correct model selection, and that situations encountered in the wild (e.g., highly variable recapture rates, other factors that reduce tag detection, and sampling error) will almost certainly reduce the likelihood that a decline in survival rate at the specified level will be detected; this is true for all simulated data.

The second set of simulation results that tested for group differences in survival rates with varying numbers of released fish over a 3-year period were consistent with the ones just discussed for time-varying survival rates in that a smaller decline in survival of 10% is more difficult to detect correctly than a decline of 20%, and that increasing recapture probability increases the chances of correctly detecting a true group difference (Table 10). Number of fish stocked in the group annually has an effect similar to recapture rates, in that larger numbers of stocked fish result in a higher chance of correctly detecting a decline of the specified magnitude under a given set of sampling conditions. For example, if managers wanted to be able to detect a 10% difference in group survival rate over a three-year period with a sampling program sufficient to obtain a 0.10 recapture rate, they could expect correct detection of group survival differences less than half (47%) of the time if 1,000 fish were stocked annually. The correct rate of detection of such a decline in survival rate increases to 84% when 2,000 fish were stocked annually and to 99% when 5,000 fish were stocked. In general, smaller batches of fish (n =1,000) should be used only when detecting larger (e.g., 20%) changes in survival rates and when recapture rates are 0.10 or higher. Groups of 2,000 fish gave good results when capture probabilities were high (0.10 or higher) or survival rate differences were large (e.g., 20%). Large batches of 5,000 fish give good results under all circumstances. The various combinations of conditions also allow comparisons of expectations when stocking number is considered a tradeoff with capture probability or effect size. For example, correct predictions of group effects were estimated to be very similar for a stocking group size of 2,000 fish when p = 0.20 and effect size is 20% (98.8% correct), compared to a stocking group size of 5,000 fish when p = 0.10 and effect size is 10% (98.9% correct). Then costs of sampling vs. the cost of raising additional fish can be evaluated; number of fish remaining after first year mortality should also be considered.

Abundance simulation methods. —Another set of simulations was conducted to determine tradeoffs in terms of bias (estimated as the % difference in estimated compared to true abundance, a negative bias reflecting abundance underestimation and a positive bias reflecting abundance overestimation, on average) and precision of abundance estimates (estimated as % coefficient of variation [CV], SE/population size*100), where number of sampling passes (3 or 4), probabilities of capture (0.02, 0.05, 0.10, and 0.20), and population sizes (1,000, 2,500, and 5,000) of animals can vary when estimating population abundance. Higher levels of bias reflect potential to mis-estimate population abundance outright, and high levels of CV reflect poor precision of estimates. Abundance estimates with CV's ≤ 10% are considered excellent, ones with CV's in the range of 10 to 25 or 30% useful, and sampling programs that result in estimates with CV's higher than 30% should reconsider the sampling designs to obtain more precise information; middle Green River reach abundance estimates for Colorado pikeminnow in 2000–2003 had CV's that ranged from 9–18%, while those in more recent years were larger and in the 32–41% range, reflecting lower probabilities of capture for many reaches. Basinwide estimates of adult Colorado pikeminnow abundance had CV's much lower; < 10% in the 2000–2003 period and 13–23% in the period 2006–2008.

Abundance simulation results.—Simulations showed that bias reduction of abundance estimates was achieved by increasing probabilities of capture, increasing sampling passes from 3 to 4, and increasing population size under consideration (Figure 12). With simulated population size of 1,000, 2,500, or 5,000, bias of estimates when p was 0.02 was high, either negative or positive. This was partially due to many simulations failing to converge when p = 0.02; when probability estimates are drawn from near the tail of a sampling distribution as they would be when the mean p = 0.02, very small estimates of p result in very large abundance estimates and lack of convergence. Simulation run results were censored (not a full 1,000 simulations was achieved because some failed to converge or abundance was estimated at greater than 10X the simulated level) at some reasonable level of abundance relative to the simulation parameters and often reduced bias relative to what is reported, but the main point here is that estimating abundance when population size is small and *p*'s are low is not advisable because highly biased estimates can result, sometimes negative but often positive. Increasing number of sampling passes when population size is 1,000 when $p = 0.05$ is advisable because 3 sampling passes resulted in potential average positive bias of nearly 20%. Relatively low average bias resulted when *p*'s were 0.10 or greater under all population sizes with 3 or 4 sampling passes, or when $p = 0.05$ or higher with population size of 2,500 or 5,000, especially when sampling pass number was increased from 3 to 4. In general, increasing the number of sampling passes from 3 to 4 reduced bias by 50% or more, regardless of the level of *p* or population size.

Similar to reducing bias, increased precision of abundance estimates was obtained by increasing probabilities of capture, increasing sampling passes from 3 to 4, and increasing population size under consideration (Figure 13). The CV's were high under all scenarios when $p = 0.02$, and remained high when $p = 0.05$ and population size was 1,000 or 2,500. When sampling yields *p*'s per pass of 0.10, estimates of abundance are relatively precise for all population sizes ranging from 1,000 to 5,000, especially when 4 sampling passes were employed. In general, addition of a fourth sampling pass was less effective than expected for increasing precision, when compared to addition of a fourth pass for bias reduction.

The CV's generated from simulation results are generally consistent with CV's for abundance estimates generated by field sampling. For example, CV's of abundance estimates (1,600 to 5,200) for razorback suckers in the lower Green River from 2006–2008 were 22 to 37% with *p*'s of 0.01 to 0.07 using 3-pass sampling. Those are similar to CV's for a simulated population of 2,500 animals with $p = 0.05$ and three sampling passes, which averaged 29.6%. Similarly, middle Green River abundance estimates for Colorado pikeminnow from 2000–2003 that ranged from 660 to 1,600 had CV's of 9 to 18% with *p*'s of 0.04 to 0.13 using 3-pass sampling. Those CV's are similar to those for simulated populations where 3 sampling passes were conducted with $p = 0.10$ and population size was 1,000 or 2,500 animals, and resulting CV's were 9 and 19% respectively.

Uncertainties

The issue of how to effectively monitor juveniles is an uncertainty, given that so few are observed in the Upper Colorado River Basin, even with presently high levels of sampling efforts. This has been an issue throughout the history of sampling fishes in the Colorado River Basin (Bestgen 1990), as few juvenile-sized razorback suckers have ever been recovered in sampling efforts, in spite of substantial efforts to do so (Minckley 1983; Bestgen 1990; Minckley et al. 1991; Bestgen et al. 2002, Zelasko et al. 2010; 2011).

An increase in abundance of larger larvae and small juveniles has been noted in annual sampling in the middle Green River, perhaps as a function of increasing abundance of early life stages. And some juveniles have been captured, including those by Modde (1996; 1997) in floodplain wetland Old Charley Wash, those by A. Webber (USFWS, Vernal, Utah) in Leota and Wyasket floodplain wetlands in 2011, and those reported in the Stirrup wetland (Hedrick et al. 2012). A few have also been captured in the main channel lower Green River, including age-1 fish in 2008. Thus, juvenile-sized fish can be captured with conventional sampling gears in regularly sampled habitats in some years.

Increased emphasis on floodplain wetland management and connections during time when larvae are available should increase the likelihood of finding larger wild-spawned juveniles in the wild. Sampling associated with evaluation of the Larval Trigger Study Plan for Flaming Gorge Dam flow releases is aimed directly at that life stage (LaGory et al. 2012) and should assist with determining the efficacy of standard sampling protocols to capture juveniles and build on additional methods that are useful.

In addition, sampling programs should begin to see additional fish in the main channel as larvae grow and recruit in floodplain wetlands and eventually leave those for the main channel. We know those smaller fish are relatively more difficult to capture, but we also know much about size-dependencies of electrofishing captures rates from recaptures of various sizes of stocked fish, including small ones. Thus, we can begin to adjust those recapture rates to assess abundance of small, untagged razorback suckers when more are observed in field sampling efforts.

Ultimately, determination of juvenile fish abundance and actual estimates of recruitment rates could come from at least two different sources. The first might be from tag-recapture sampling and analysis. This approach would be able to directly estimate recruitment rates, based on capture rates of untagged razorback suckers, which is an important part of recovery goals that require recruitment at levels that offset mortality rates (USFWS 2002). However, the restrictive assumption of that approach is that all untagged fish are wild-produced, which we know is likely not true, because of stocking of "excess" fish in various places, as well as tag loss, tag failure, improper tag scanning and other issues that result in a non-detection of a once-tagged fish. Because recapture rates are already relatively low, careful tag detection and data recording becomes even more vital to effective parameter estimation. Zelasko et al. (2010) summed up this issue regarding their use of tagged fish recaptures as follows:

> "*Of the 4,010 total razorback sucker capture events we examined, at least 275 records (6.9%) had PIT tag errors which made them unusable. Another 328 capture records (8.2%) did not have associated stocking data, which could be the result of captures of wild untagged fish, captures of hatchery-reared untagged fish, loss of tags, failure of equipment to detect tags, or failure to scan fish prior to stocking. Accurate tagging, tag detection, and data recording are minimal requirements to understand provenance of captured fish (hatchery or wild) and recruitment rates of razorback suckers*."

Some of those errors or problems can be accommodated in an analysis informed by things like tag loss rates, but presence of large numbers of untagged fish as juveniles, when large numbers of unmarked early life stages are released one to several years prior is problematic and not likely treatable in an analysis. Those factors will frustrate attempts to estimate recruitment rates of wild fish until they are minimized to a much greater level.

Another approach that would aid identifying actual wild-produced juveniles from untagged or otherwise unrecognizable hatchery fish would be to employ trace element analyses of structures such as scales. Investigators have used this technique to differentiate wild from stocked fish of other species and preliminary analyses are underway using scales from San Juan River razorback suckers (Flem et al. 2005, S. Platania, pers. comm.). Preliminary results show clear differences between element concentrations in scales consistent with when fish were in the hatchery and were then subsequently captured in the river after a reasonably long time interval. Verification of the number and, potentially, rates of wild fish recruitment would be very useful and presumably require little additional work other than the analysis of the scales themselves, since non-regenerated scale samples could be collected when untagged fish were noted in other ongoing sampling programs.

Data analysis and reporting needs

Analysis of capture data for early life history sampling is conducted annually for projects 22f, 160, and is anticipated for project 163 as well. Portions of the Green River data will also be responsive to reporting requirements under the Larval Trigger Study Plan (LaGory et al. 2012). Emphasis should be placed on adequacy of sampling to detect timing of hatching, abundance of larvae, and abundance trends over time. Data collection associated with specific tasks to improve early life history sampling should be reported separately and as soon as possible so results can guide additional sampling.

Analysis of large-bodied juvenile and adult razorback sucker data should be completed soon after or in conjunction with Colorado pikeminnow abundance estimates for which data is gathered for three-year intervals. Preliminary survival and abundance estimates have already been completed based just on captures obtained from Colorado pikeminnow sampling data (Osmundson and Seal 2009; this report). Additional abundance estimates should be modeled when the next set of data is available from the Green River in 2013, in an attempt to use data from more years and reaches, which may improve estimates. In the Colorado River, it has been proposed that population estimates of razorback sucker will be developed in 2012 for the years 2008, 2009, and 2010, for which mark-recapture data already exist. Survival rate analyses could be better accomplished using all available information from all sampling programs and all years. Such could be accomplished after each three-year Colorado pikeminnow abundance-monitoring period, but should include data for the previous five years. For example, when Colorado pikeminnow abundance estimation sampling in the Green River ceases for the three-year period 2011–2013 in autumn 2013, analysis of survival data could be completed at that time with data collected from 2009–2013. A similarly formatted schedule (dates are already set) could be followed in the Colorado River. A summary report of all data from each of the Colorado and Green River basins should be proposed for intervals of 3–5 years.

CONCLUSIONS

- Sampling to monitor timing of presence and abundance of razorback sucker larvae in the middle and lower Green is adequate at this time.

- Sampling for larvae in the lower White River where reproduction was documented for the first time in 2011 should continue at levels outlined in Project 22f.
- Sampling in the lower Yampa River for razorback sucker larvae may be warranted but at a low level that may include near shore seine sampling conducted within 1–2 weeks after first appearance of middle Green River larvae.
- Sampling for razorback sucker larvae in the Colorado River should proceed in accordance with plans to evaluate razorback sucker reproduction in the lower Gunnison and Colorado rivers per Project 163. When higher levels of reproduction there are noted, the monitoring program should be revised to incorporate new information on distribution of larvae relative to spawning areas and establishment of sentinel locations that can be used to monitor timing of reproduction and population abundance over time.
- Although early life stage sampling programs are efficient, additional experimental sampling program evaluations are needed to verify what we think sampling data suggests. Those experiments include but are not limited to light trap attraction distances and sampling efficiency studies in the lab and the field, and field releases of marked larvae near spawning areas and wetland breaches to assess dispersal and wetland colonization. Occupancy analyses may also aid in determining colonization probabilities of larvae in wetlands, given that detection probabilities of larvae in wetlands may be less than one.
- Morphological techniques are the benchmark for identification of larvae and are reliable. Verification of identity of larvae via appropriate genetic techniques, especially identities that are questionable, would be useful. Minimally, such a study needs to be conducted with an appropriate design, including development of sufficient taxonomic markers, and reliably identified reference populations from across the ranges of candidate taxa.
- Real-time monitoring requires continued real-time identification of razorback sucker larvae in samples collected in the middle Green River. Annual updates of early life history sampling and data collection is needed to inform management.
- Existing sampling programs, especially Colorado pikeminnow abundance estimation sampling, provides useful data with which to monitor razorback sucker populations in the Green and Colorado rivers.
- Although estimates generated from analyses of razorback sucker recapture data were mostly unreliable, some were noteworthy. Relatively low population sizes were apparent in spite of large numbers of stocked fish in the system, supporting previous analyses that

pointed to low survival of stocked fish. This was especially evident in the Desolation-Gray Canyon reach where survival of stocked fish was apparently very low, because no fish were recaptured among years in the 2006–2008 period.

- Sampling effort in existing sampling programs is likely insufficient to produce unbiased and relatively precise (CV's of 10–25%) estimates of survival and abundance of razorback suckers in the Colorado or Green River. Increased recaptures of razorback suckers are needed, particularly among sampling passes within a year, if abundance estimates are desired. Increased sampling efforts would be most useful when conducted in the same years and seasons as sampling for Colorado pikeminnow abundance estimation.
- Increased recaptures of razorback suckers should be attempted by using additional gears and technology, with an emphasis on seasons and places where fish are concentrated. Those would typically involve spawning areas or low-velocity channel margin areas such as backwaters, flooded tributary mouths or washes, and floodplain wetlands that are seasonally flooded. Antennae arrays to detect PIT tags at known or suspected spawning locations are a non-invasive means to obtain additional recapture information, which if employed at times overlapping abundance estimation sampling for Colorado pikeminnow, could be used for razorback sucker abundance estimates. Passive gears such as trap or fyke nets have been used in the past to increase precision of abundance and survival estimates for razorback suckers in the middle Green River. Use of such may be possible in other reaches, including the lower Green River, or the Colorado River. A less desirable means to increase capture rates of razorback suckers is to sample over spawning areas when fish are concentrated in spring.
- Length and change in length data are useful to inform size dependent changes in capture or survival rates, to estimate growth rates between recapture intervals, and to estimate condition indices. Accordingly, inaccurate length data reduces our ability to accurately assess those things.
- The number of untagged razorback suckers captured in the wild is relatively high. It is important to minimize the number of apparently untagged razorback suckers in the wild, which could result from not tagging hatchery fish, loss of tags, faulty scanning, or other tag-related issues, so that true recruitment (e.g., an untagged wild fish) can be better determined.

- Determining origin of untagged fish, whether hatchery-reared or wild, would aid in determining recruitment rates of razorback suckers.
- Simulations are useful to guide assignment of minimal levels of sampling effort to accomplish goals of parameter estimation.
- Rigorous data analysis is needed to evaluate whether sampling efforts for larger juvenile and adult razorback sucker are effective at improving accuracy and reliability of abundance and survival estimates in the future.

RECOMMENDATIONS

- Maintain present sampling to monitor timing of presence and abundance of razorback sucker larvae and juveniles in the middle and lower Green and Colorado rivers. Minimally, high quality and close-up digital photographs should be taken of questionable specimens that include the lateral view of the body as well as the ventral view of the mouth, and in some cases, specimen preservation in appropriate solutions should be considered.
- Maintain sampling for larvae in the lower White River where reproduction was documented for the first time in 2011 at levels outlined in project 22f. Sampling should be evaluated in the future (after 3 more years) and adapted to presence and abundance patterns noted.
- Conduct additional but relatively limited fine-mesh seine sampling in the lower Yampa River for razorback sucker larvae within 1–2 weeks after first appearance of middle Green River larvae.
- Maintain sampling for razorback sucker larvae in the Colorado River in accordance with plans to evaluate razorback sucker reproduction in the lower Gunnison and Colorado rivers per project 163, perhaps adding light trap sampling in gravel pit ponds and other appropriate areas. When higher levels of reproduction there are noted, revise the monitoring program to incorporate new information on distribution of larvae relative to spawning areas and establishment of sentinel sites that can be used to monitor timing of reproduction and population abundance over time.

- Conduct additional experimental early life stage sampling programs to assess capture efficiency with light traps, and dispersal and colonization of wetlands by larvae. Use occupancy analyses to aid in determining colonization probabilities of larvae in wetlands, given that detection probabilities of larvae in wetlands may be less than perfect.
- Verify the benchmark morphological techniques for identification of razorback sucker larvae using appropriate genetic techniques, especially for specimens that are questionable, or potentially of hybrid origin. Use an appropriate design, including development of sufficient taxonomic markers, and reliably identified reference populations from across the ranges of candidate taxa.
- Maintain existing sampling programs, especially Colorado pikeminnow abundance estimation sampling and standardized monitoring programs for fish in backwaters, which provides useful data with which to monitor razorback sucker populations in the Green and Colorado rivers.
- Add sampling effort to existing sampling programs, or increase effort in other programs, to increase fish captures and recaptures to reduce bias and increase precision of survival and abundance estimates of razorback suckers in the Colorado or Green River systems. Increased recaptures of razorback suckers are of particular importance if useful abundance estimates are desired.
- Increase recaptures of razorback suckers by using additional gears (e.g., trap/fyke nets) and technology (PIT tag detector arrays), with an emphasis on seasons and places where fish are concentrated. Electrofishing over spawning areas has been conducted in the past, but has potentially negative effects because of disruption to spawning fish, and is viewed as unnecessary if detector arrays are installed and are effective.
- Investigate reasons for low recapture rates and low apparent survival of razorback suckers stocked into or resident in Desolation-Gray Canyon.
- Obtain more accurate length information on stocked and recaptured fish. This includes both consistent measurement and recording techniques.
- Reduce numbers of apparently untagged fish in the wild that result from fish not being tagged, tag loss, tag failure, or scanner issues.
- Evaluate efficacy of determining origin of untagged fish, whether hatchery-reared or wild, through elemental or other analyses which would aid in determining recruitment rates of razorback suckers.

- Maintain appropriately skilled people to provide real-time identification of razorback sucker larvae in the middle Green River. Continue annual analysis of early life history sampling, especially for the middle Green River.
- Perform rigorous analysis of tag-recapture data for larger juveniles and adults at appropriate intervals (minimally every three years) to evaluate whether additional sampling efforts are effective at improving accuracy and reliability of abundance and survival estimates in the future.

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Table 1.—Mean total length (TL) and standard deviation (SD) of wild razorback sucker larvae collected in the middle and lower Green River, Utah.

Reach	Year	*n*	Mean TL (mm)	SD	Range (mm TL)
Middle Green					
	1993	228	12.9	1.88	10.0–24.0
	1994	1217	11.7	0.84	9.0–18.0
	1995	32	11.9	1.13	10.0–16.0
	1996	174	11.8	1.09	10.2–16.5
	1997	3	11.6	0.40	11.2–12
	1998	58	12.5	1.31	10.7–16.3
	1999	12	-	-	-
	2000	82	11.5	0.98	9.8–16.2
	2001	89	12.1	1.38	11.0–16.0
	2002	93	12.6	1.33	10.0–16.0
	2003	47	10.8	0.82	10.0–13.5
	2004	1047	11.1	1.12	7.1–18.4
	2005	172	12.9	2.42	9.8–21.0
	2006	535	11.3	0.90	9.2–18.0
	2007	2293	11.3	1.41	7.0–19.0
	2008	889	11.9	1.41	10.0–19.0
Lower Green					
	1993	120	12.7	0.90	11.0–16.0
	1994	76	11.9	1.03	10.0–15.3
	1995	5	12.2	0.58	11.3–12.8
	1996	214	11.9	1.40	9.8–18.2
	1997	3	12.9	0.70	12.2–13.6
	1998	57	13.2	2.28	10.8–19.7
	1999	30	12.4	1.55	10.5–15.5

Table 2.—Number of razorback sucker captures in the Green River, Utah, per Recovery Program sampling study, 2006–2008. Sampling programs list below are as follows: CS POP Est = Colorado pikeminnow abundance estimation sampling, HB POP EST = humpback chub abundance estimation sampling, NP REMOVAL = northern pike removal sampling, and SMB REMOVAL = smallmouth bass removal sampling.

	Recapture Year			
Project	2006	2007	2008	total
CS POP EST	404	285	390	1079
HB POP EST	5	6		11
NP REMOVAL	16			16
SMB REMOVAL	15	14	42	71
Grand Total	440	305	432	1177

Table 3.—Number of razorback suckers recaptured in the Green River, Utah, 2006–2008, by year in which they were stocked.

	Recapture Year			
Stocking year	2006	2007	2008	total
1996			1	1
1998	2		2	4
2000		3	3	6
2001	1			1
2002	1	1	2	4
2003	39	29	31	99
2004	121	77	87	285
2005	268	84	111	463
2006	8	103	72	183
2007		8	123	131
total	440	305	432	1177

Table 4.—Number of razorback suckers captured per sampling pass in three reaches of the Green River, Utah, 2006–2008, during the Colorado pikeminnow abundance estimation program. The Middle Green River reach was from river kilometer (RK) 539.5–396.0 (143 RK long), the Desolation-Gray Canyon reach was from RK 395.9–206.1 (189.8 RK), and the lower Green reach was from RK193.2–0 (193.2 RK).

Pass	Recapture Year			
	2006	2007	2008	total
Middle Green River				
1	34			
2	12			
3	6			
1		4		
2		23		
3		29		
1			8	
2			20	
3			34	
total	52	56	62	170
Desolation-Gray Canyon				
1	17			
2	34			
3	10			
1		13		
2		13		
3		12		
1			20	
2			10	
3			9	
total	61	38	39	138
Lower Green River				
1	148			
2	82			
3	61			
1		70		
2		85		
3		36		
1			110	
2			115	
3			64	
total	291	191	289	771
total	404	285	390	1079

Table 5.—Dates of sampling passes in three reaches of the Green River, Utah, 2006–2008, for the Colorado pikeminnow abundance estimation program. The middle Green River reach was from river kilometer (RK) 539.5–396 (143 RK long), the Desolation-Gray Canyon reach was from RK 395.9–206.1 (189.8 RK), and the lower Green reach was from RK193.2–0 (193.2 RK).

Pass	Year		
	2006	2007	2008
	middle Green River		
1	Apr 17 – Apr 27	Apr 11 – Apr 27	Apr 21 – Apr 29
2	May 2 – Jun 1	Apr 30 – May 9	Apr 30 – May 8
3	Jun 5 – Jun 14	May 14 – May 24	May 13 – May 21
	Desolation-Gray Canyon		
1	Mar 27 – Apr 7	Apr 3 – Apr 7	Apr 24 –May 2
2	Apr 10 – Apr 14	Apr 11 – Apr 15	May 4 – May 13
3	Apr 19 – Apr 26	Apr 16 – Apr 26	May 12 – May 21
	lower Green River		
1	May 3 – 10	May 9 – May 17	Apr 22 – Apr 30
2	May 24 – June 1	May 23 – May 31	May 6 – May 14
3	June 14 – 21	Jun 6 – Jun 13	May 20 – May 27

Table 6.—Razorback sucker mean survival estimates from 2006-2008 (*S*), and their associated standard errors (*SE*), lower (L) and upper (U) 95% confidence limits, and coefficients of variation (CV, [SE/estimate]*100). The middle Green River reach was from river kilometer (RK) 539.5–396 (143 RK long), the Desolation-Gray Canyon reach was from RK 395.9–206.1 (189.8 RK), and the lower Green reach was from RK193.2–0 (193.2 RK).

Reach	*S*	SE	L95%CI	U95%CI	CV
lower Green River	0.51	0.102	0.32	0.70	20.1
Desolation-Gray	0				
middle Green River	0.69	0.356	0.08	0.98	51.7

Table 7.—Razorback sucker probabilities of capture (*p*), and their associated standard errors (*SE*), and lower (L) and upper (U) 95% confidence limits by sampling year in three reaches of the Green River, 2006–2008. The middle Green River reach was from river kilometer (RK) 539.5–396 (143 RK long), the Desolation-Gray Canyon reach was from RK 395.9–206.1 (189.8 RK), and the lower Green reach was from RK193.2–0 (193.2 RK).

Reach	Year	*p*	SE	L95%CI	U95%CI
lower Green River	2006	0.070	0.015	0.046	0.105
	2007	0.014	0.005	0.007	0.028
	2008	0.041	0.010	0.025	0.066
Desolation-Gray	2006	0.040	0.020	0.015	0.102
	2007	0.008	0.005	0.002	0.026
	2008	0.023	0.013	0.008	0.068
middle Green River	2006	0.021	0.012	0.007	0.062
	2007	0.004	0.003	0.001	0.014
	2008	0.012	0.007	0.004	0.038

Table 8.—Abundance estimates ($\hat{N}$) for razorback sucker, and their associated standard errors (*SE*), lower (L) and upper (U) 95% confidence limits, and coefficients of variation (CV, [SE/estimate]*100) by sampling year in three reaches of the Green River, 2006–2008. The middle Green River reach was from river kilometer (RK) 539.5–396 (143 RK long), the Desolation-Gray Canyon reach was from RK 395.9–206.1 (189.8 RK), and the lower Green reach was from RK193.2–0 (193.2 RK). The # Stocked represents the number of razorback suckers stocked in the reach specified in the year prior to the estimate.

Reach	Year	$\hat{N}$	SE	L95%CI	U95%CI	CV	# Stocked
lower Green River	2006	1582	344.2	1061	2446	21.8	4231
	2007	5153	1907.0	2588	10460	37.0	5113
	2008	2597	683.1	1595	4359	26.3	8539
Desolation-Gray	2006	474	233.3	207	1217	49.3	0
	2007	3011	2422.3	772	12076	80.5	10075
	2008	836	535.9	280	2677	64.1	0
middle Green River	2006	576	315.2	227	1608	54.8	2917
	2007	3146	1970.2	1039	9764	62.6	5021
	2008	1218	699.3	448	3514	57.4	7749

Table 9.—Simulation results that depict the % of times (n = 1,000 simulations) the true model was chosen given a specified decline in survival rate (10 or 20%) from 80%, and annual probabilities of capture over a three-year period of 0.02, 0.05, 0.10, and 0.20. For example, under an assumed 10% decline in survival rate and a recapture probability of 0.05, the correct model of a true decline in survival rate over three years was chosen only 24.6% of the time, and the incorrect model of constant survival rate over time was chosen 75.4% (100–24.6 = 75.4%) of the time. Population size simulated was 5,000 razorback suckers.

change in survival rate	recapture probability (*p*)	% simulations that chose true model
0.10	0.02	13.2
(0.80 to 0.70)	0.05	24.6
	0.10	46.3
	0.20	81.0
0.20	0.02	37.5
(0.80 to 0.60)	0.05	78.0
	0.10	96.9
	0.20	100.0

Table 10.—Simulation results that tested for group differences in survival rates (true model = difference in survival of 10% or 20% among groups from 80%) with varying numbers of released fish over a 3-year period (n = 1,000 simulations). Annual probabilities of capture over a three-year period were 0.02, 0.05, 0.10, and 0.20.

simulated release (*n*)	decline in survival rate	recapture probability (*p*)	% simulations that chose true model
1,000	0.10	0.02	11.5
	(0.80 to 0.70)	0.05	23.6
		0.10	47.1
		0.20	77.8
	0.20	0.02	38.9
	(0.80 to 0.60)	0.05	75.5
		0.10	95.9
		0.20	100.0
2,500	0.10	0.02	23.8
	(0.80 to 0.70)	0.05	51.8
		0.10	84.2
		0.20	98.8
	0.20	0.02	74.1
	(0.80 to 0.60)	0.05	98.4
		0.10	100.0
		0.20	100.0
5,000	0.10	0.02	95.7
	(0.80 to 0.70)	0.05	85.1
		0.10	98.9
		0.20	100.0
	0.20	0.02	96.6
	(0.80 to 0.60)	0.05	100.0
		0.10	100.0
		0.20	100.0

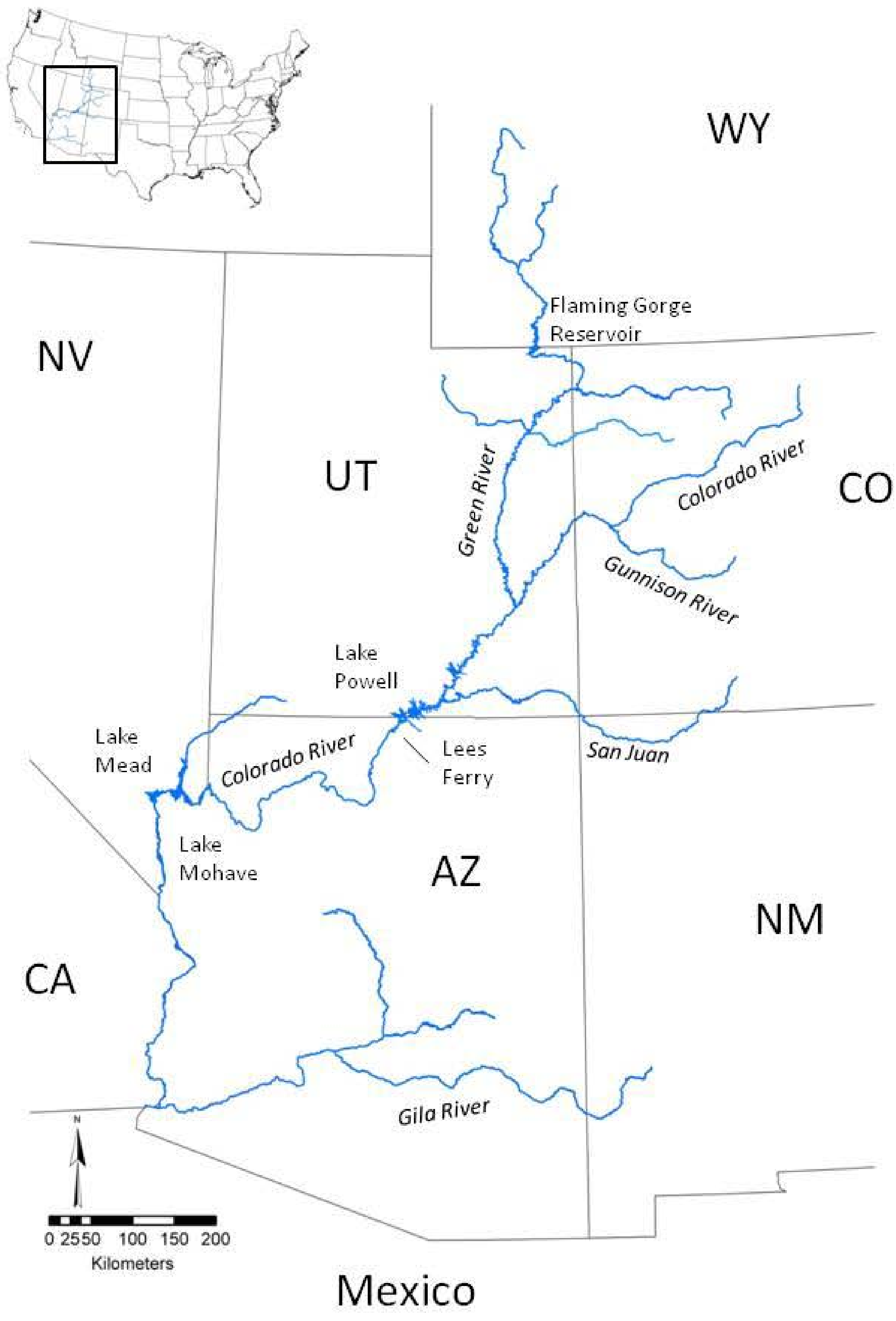

Figure 1. Colorado River Basin with Upper Colorado River study area.

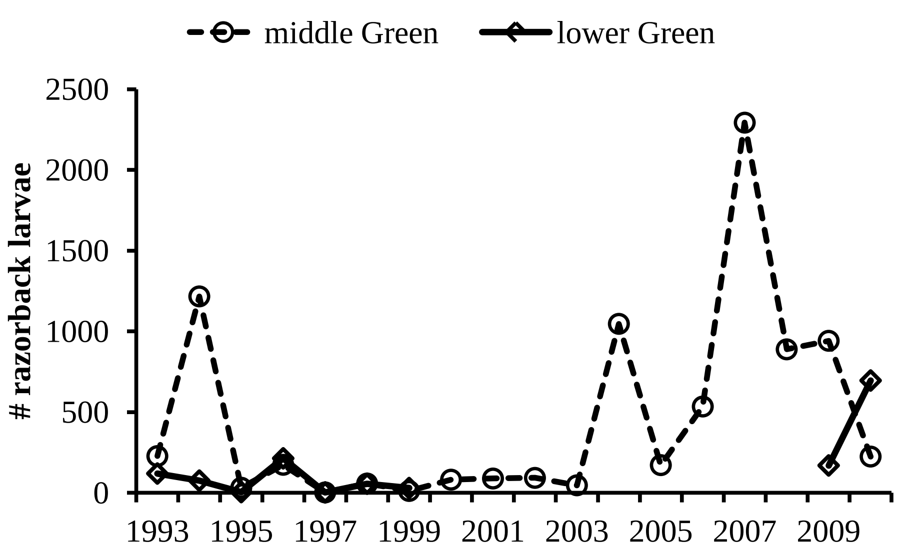

Figure 2. Number of razorback sucker larvae captured in light trap samples the middle and lower Green River, Utah, 1993–2010.

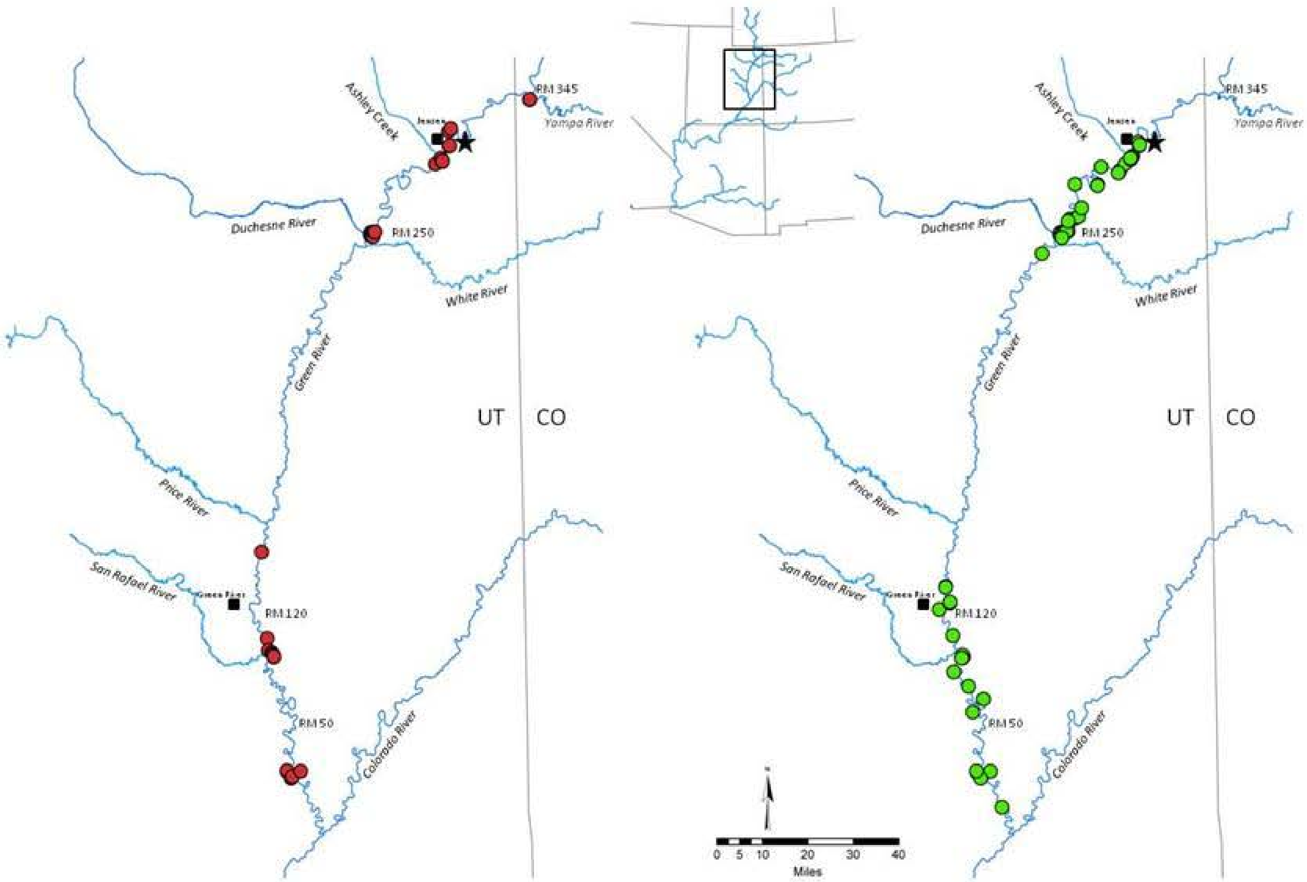

Figure 3. Map of razorback sucker larvae captures, middle and lower Green River, in two time periods, 1993–1999 (left map), and 2000–2010 (right map). That time separation was used because wild adult razorback suckers likely produced most larvae prior to 2000, but few were available for reproduction after 2000 when those fish were rare or extirpated and large numbers of stocked fish were in the system (Bestgen et al. 2002, Zelakso et al. 2010). Data were available in the middle Green River from 1993–2010; data for the lower Green River were available from 1993–1999, and only 2009–2010 in the later period. Razorback sucker larvae captured in the lower Yampa River (e.g., 2000, 2008) are not depicted in the recent map because they were not captured in Project 22f sampling. The star depicts the general location of razorback sucker spawning areas (Razorback Bar and Escalante Bar) in the middle Green River.

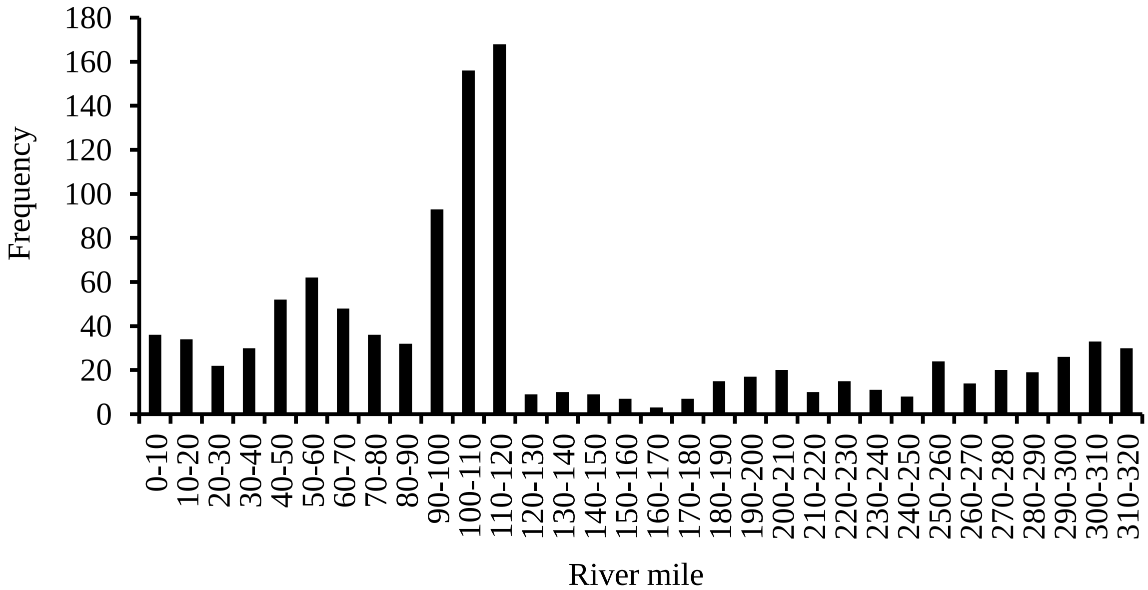

Figure 4. Concentration areas of juvenile and adult razorback suckers (mean TL = 332 mm, 203–505 mm) captured in the Green River, Utah, during sampling for Colorado pikeminnow abundance estimation, 2006–2008. Frequencies are captures summed over the three years for each 10-mile river increment: the middle Green River reach was from river mile (RM) 320–246 (river kilometer [RK] 539.5–396 [143 RK long]), the Desolation-Gray Canyon reach was from RM 245.9–128 (RK 395.9–206.1 [189.8 RK]), and the lower Green reach was from RM 120–0 (RK193.2–0 [193.2 RK]) at the confluence with the Colorado River.

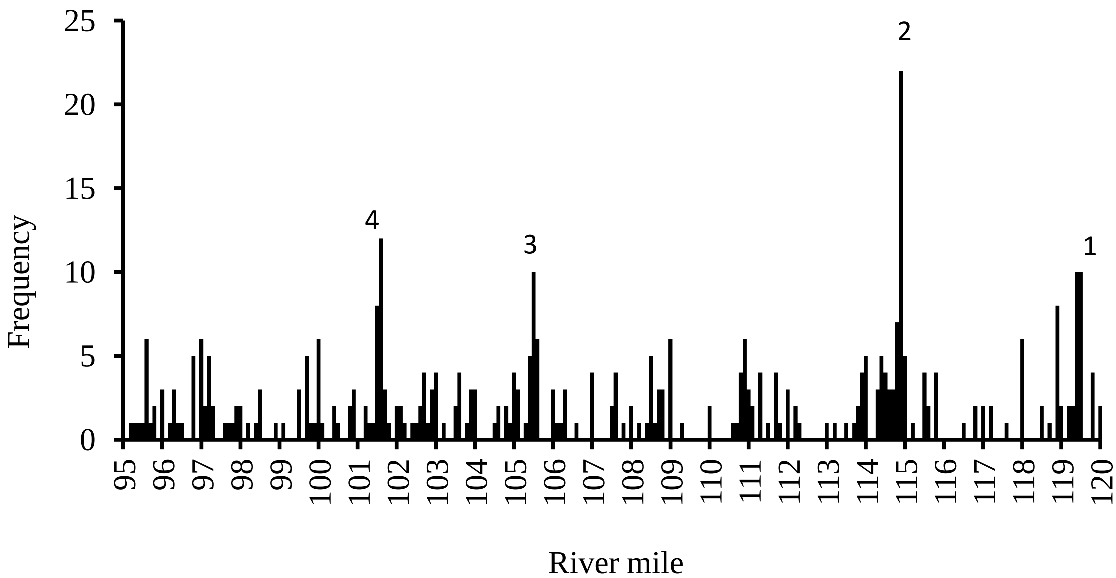

Figure 5. Concentration areas of juvenile and adult razorback suckers captured from river mile 120–95 in the lower Green River, Utah, during sampling for Colorado pikeminnow abundance estimation, 2006–2008. Frequencies are captures summed over the three years for each 1/10-river mile increment; the first and most upstream location was at RM 119.4–119.5, near Brown's Wash on river left and Saleratus Wash on river right, the second was at RM 114.9, at or near Little Grand Wash on river left, the third was at RM 105.5 near Salt Wash on river left, and the fourth was at RM 101.6 and associated with Dry Lake Wash on river right.

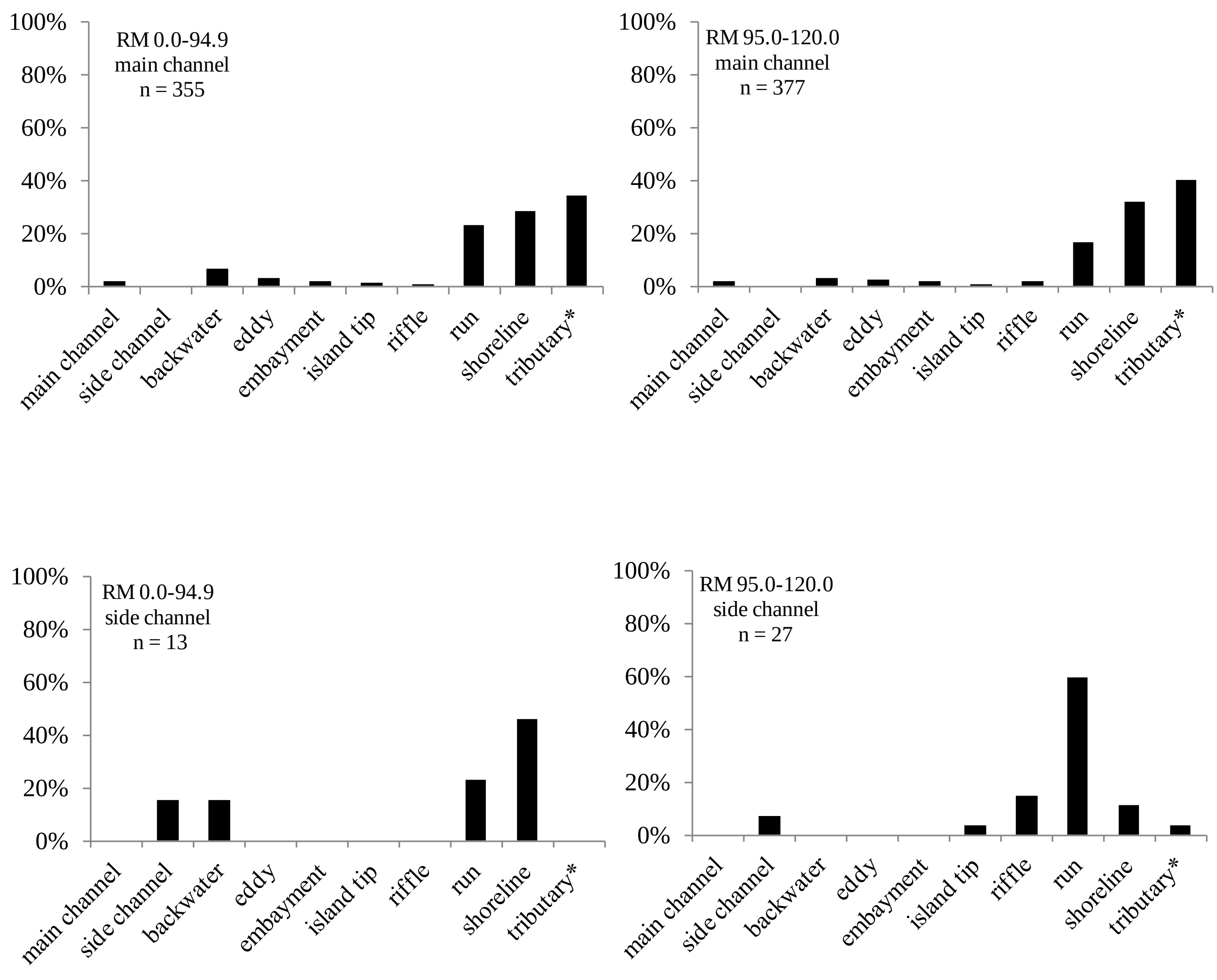

Figure 6. Percent macrohabitat use by razorback suckers captured in the downstream (RM 94.9–0, left panels) and upstream ((RM 120–90, right panels) sections of the lower Green River, Utah, captured each spring during 2006–2008 sampling for Colorado pikeminnow abundance estimation. Primary habitat types in each reach were either main channel (upper panels) or side channel (lower panels), and secondary habitats within each of those, for each reach, are across the *x*-axis of each graph. Tributary (*) includes perennial or intermittent stream mouths confluent with the Green River and flooded mouths of washes or side canyons.

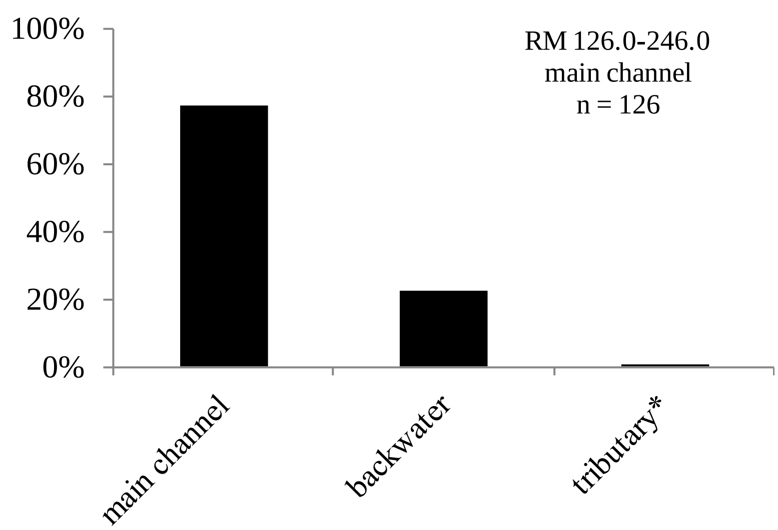

Figure 7. Percent macrohabitat use by primary habitat type for razorback suckers captured in the Desolation-Gray Canyon reach of the Green River, Utah, from RM 245.9–128 (RK 395.9–206.1 [189.8 RK total]). Tributary (*) includes perennial or intermittent stream mouths confluent with the Green River and flooded mouths of washes or side canyons.

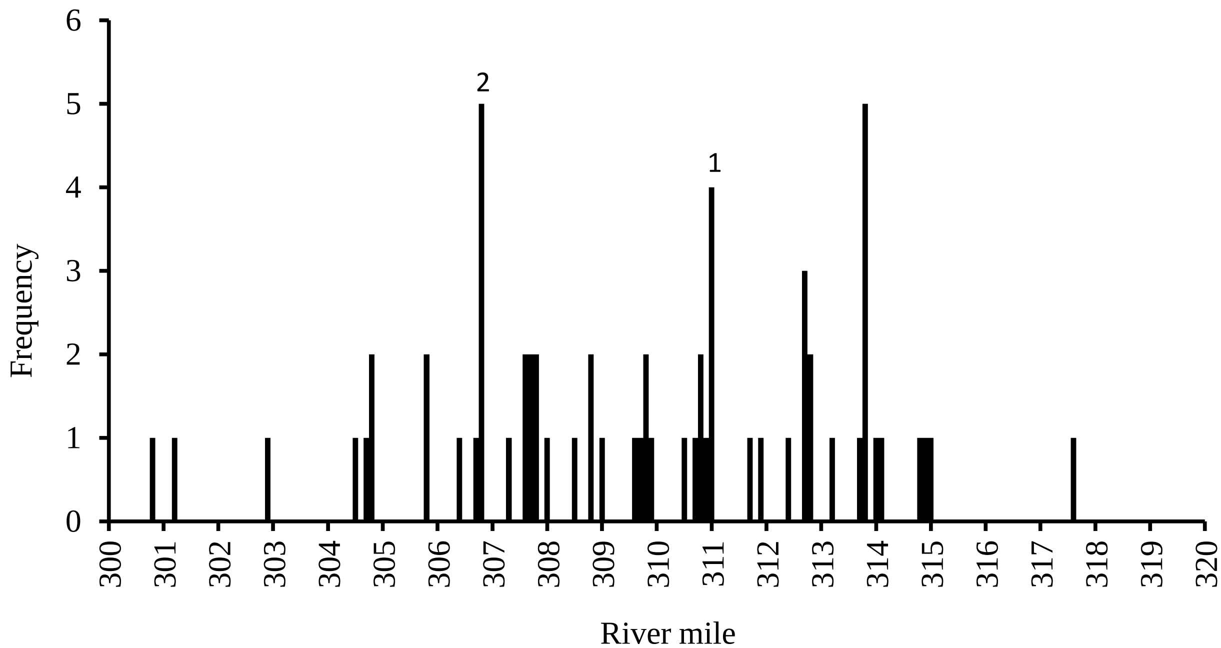

Figure 8. Concentration areas for large juvenile and adult razorback suckers captured from river mile 320–300 in the middle Green River, Utah, during sampling for Colorado pikeminnow abundance estimation, 2006–2008. Frequencies are captures summed over the three years for each 1/10 river mile increment; location 1 is Razorback Spawning Bar and location 2 is the Escalante Spawning Bar; the concentration at RM 313.8 is Cub Creek.

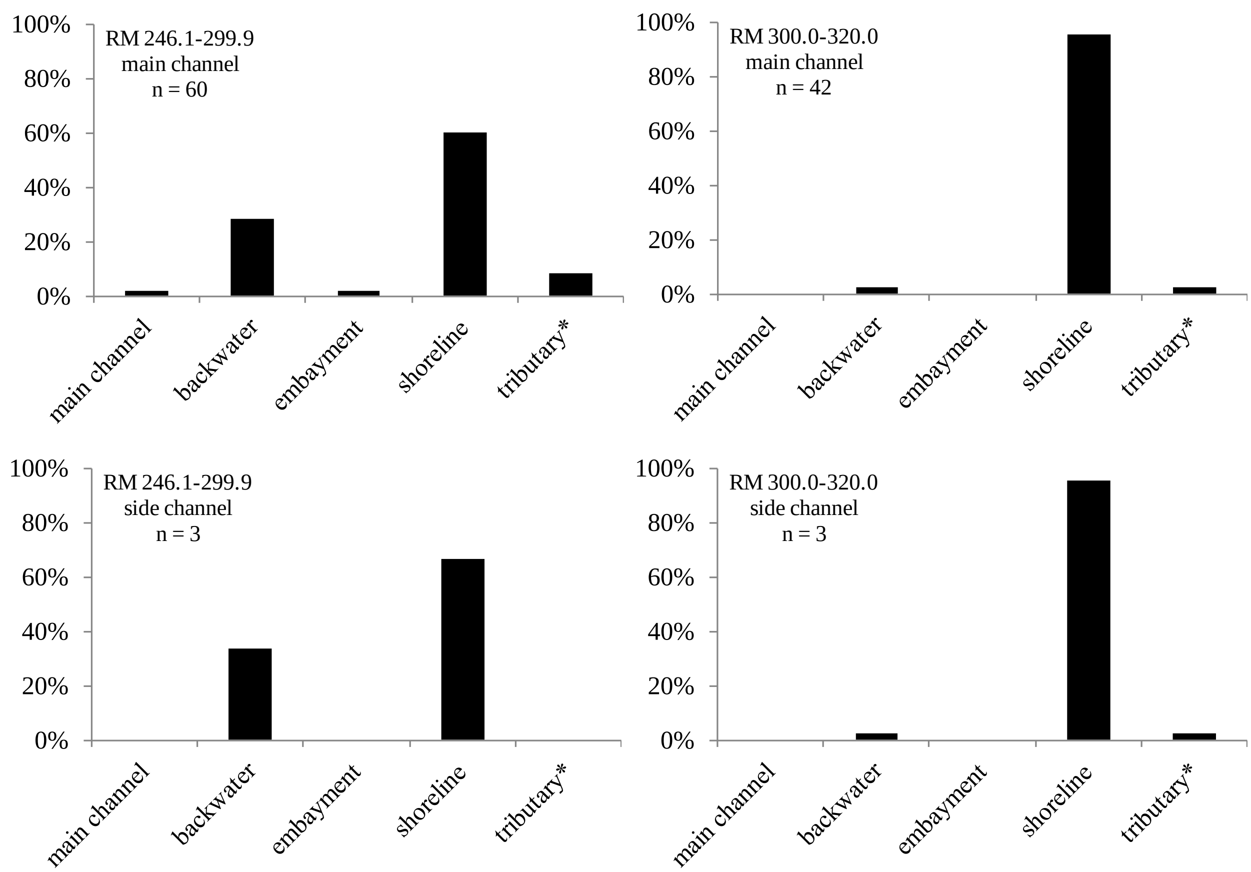

Figure 9. Percent macrohabitat use by razorback suckers in the downstream (RM 299.9–246.1, left panels) and upstream (RM 320–300, right panels) sections of the middle Green River, Utah, captured each spring during 2006–2008 sampling for Colorado pikeminnow abundance estimation. Primary habitat types in each reach were either main channel (upper panels) or side channel (lower panels), and secondary habitats within each of those, for each reach, are across the *x*-axis of each graph. Tributary (*) includes perennial or intermittent stream mouths confluent with the Green River and flooded mouths of washes or side canyons.

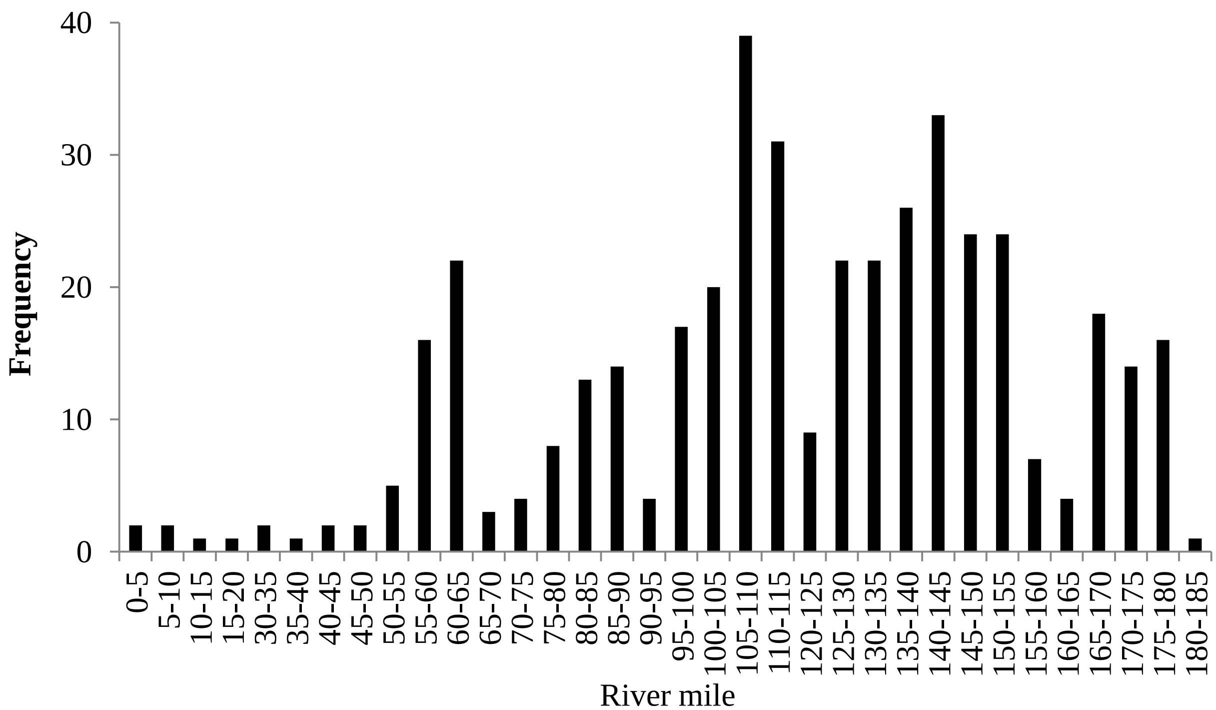

Figure 10. Concentration areas of large juvenile and adult razorback suckers captured in the Colorado River, Colorado and Utah, during sampling for Colorado pikeminnow abundance estimation, 2005. Frequencies are for captures for each 5-mile river increment from RM 185 which is just downstream of Price-Stubb diversion dam, downstream to the confluence of the Green River.

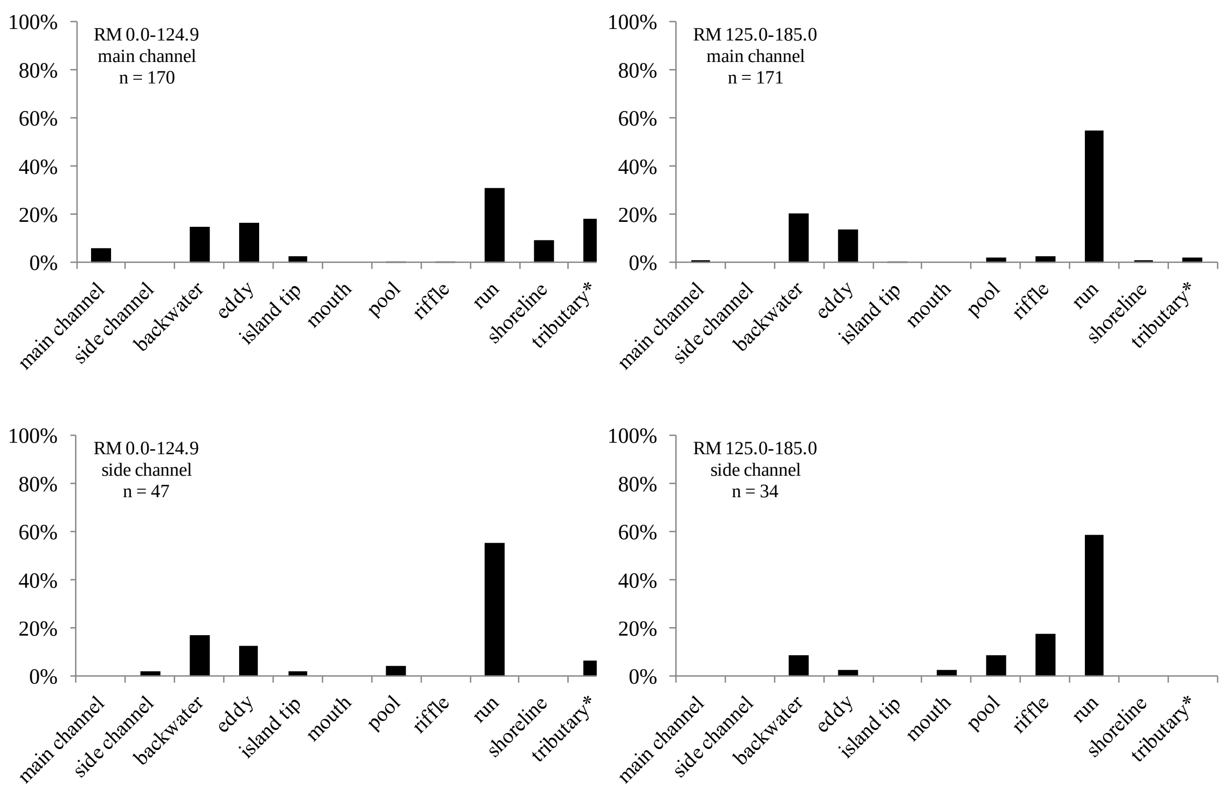

Figure 11. Percent macrohabitat use by razorback suckers captured in the downstream (RM 124.9–0, left panels) and upstream (RM 185–125, right panels) sections of the Colorado River, Colorado and Utah, captured in spring 2005 during sampling for Colorado pikeminnow abundance estimation. Primary habitat types in each reach were either main channel (upper panels) or side channel (lower panels), and secondary habitats within each of those, for each reach, are across the *x*-axis of each graph. Tributary (*) includes perennial or intermittent stream mouths confluent with the Green River, and flooded mouths of washes or side canyons.

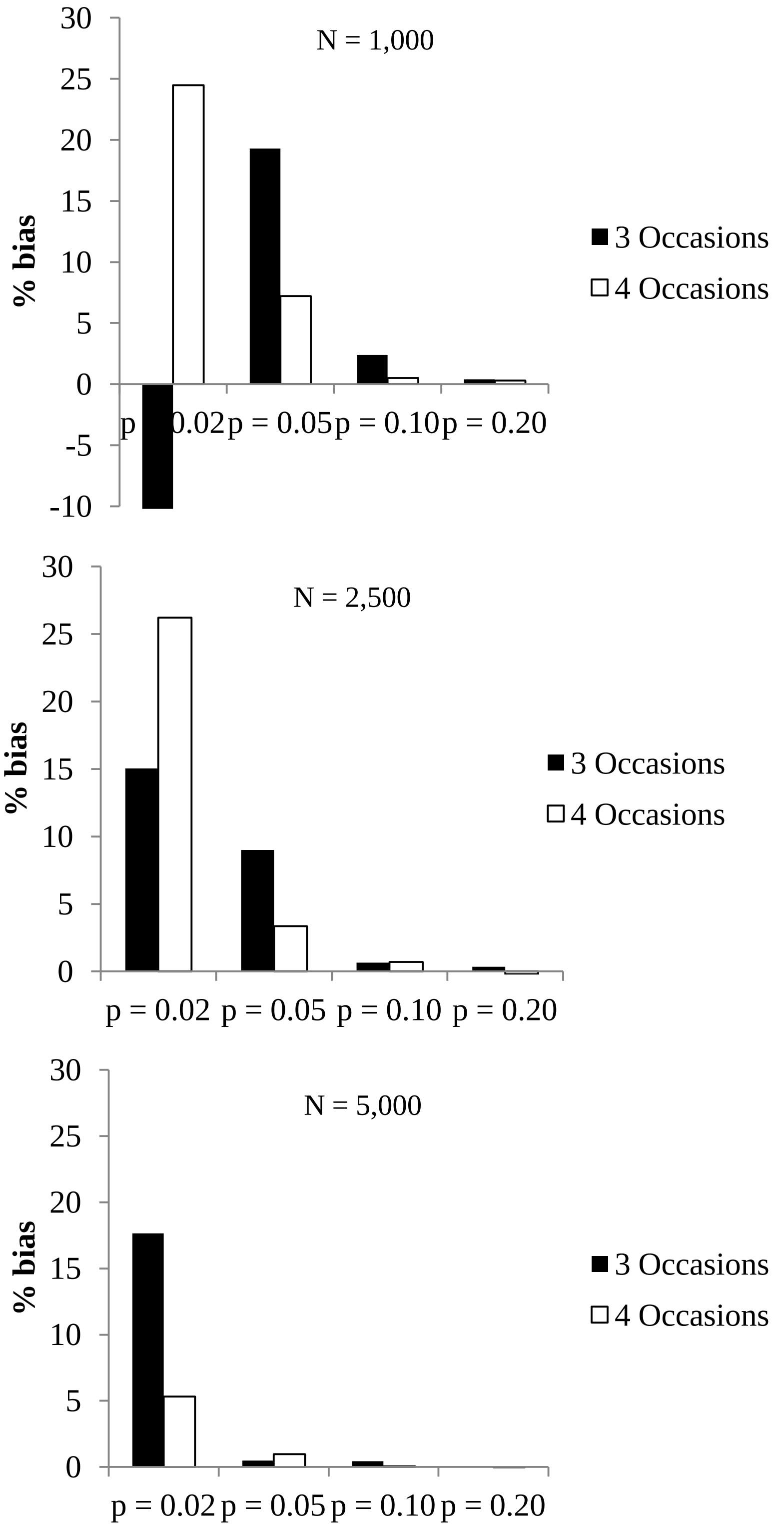

Figure 12. Simulation results that depict the bias of abundance estimates under four probabilities of capture per pass 0.02, 0.05, 0.10, and 0.20, using three or four sampling occasions, where true population size was 1,000 (top panel), 2,500 (middle panel), or 5,000 (bottom) fish (n = 1,000 simulations). Some simulations at $p = 0.02$ did not converge and results were censored when population size was 10x or more higher than the true population size specified in simulations, which sometimes gives non-intuitive results (e.g. lower bias for 3 than 4 sampling passes).

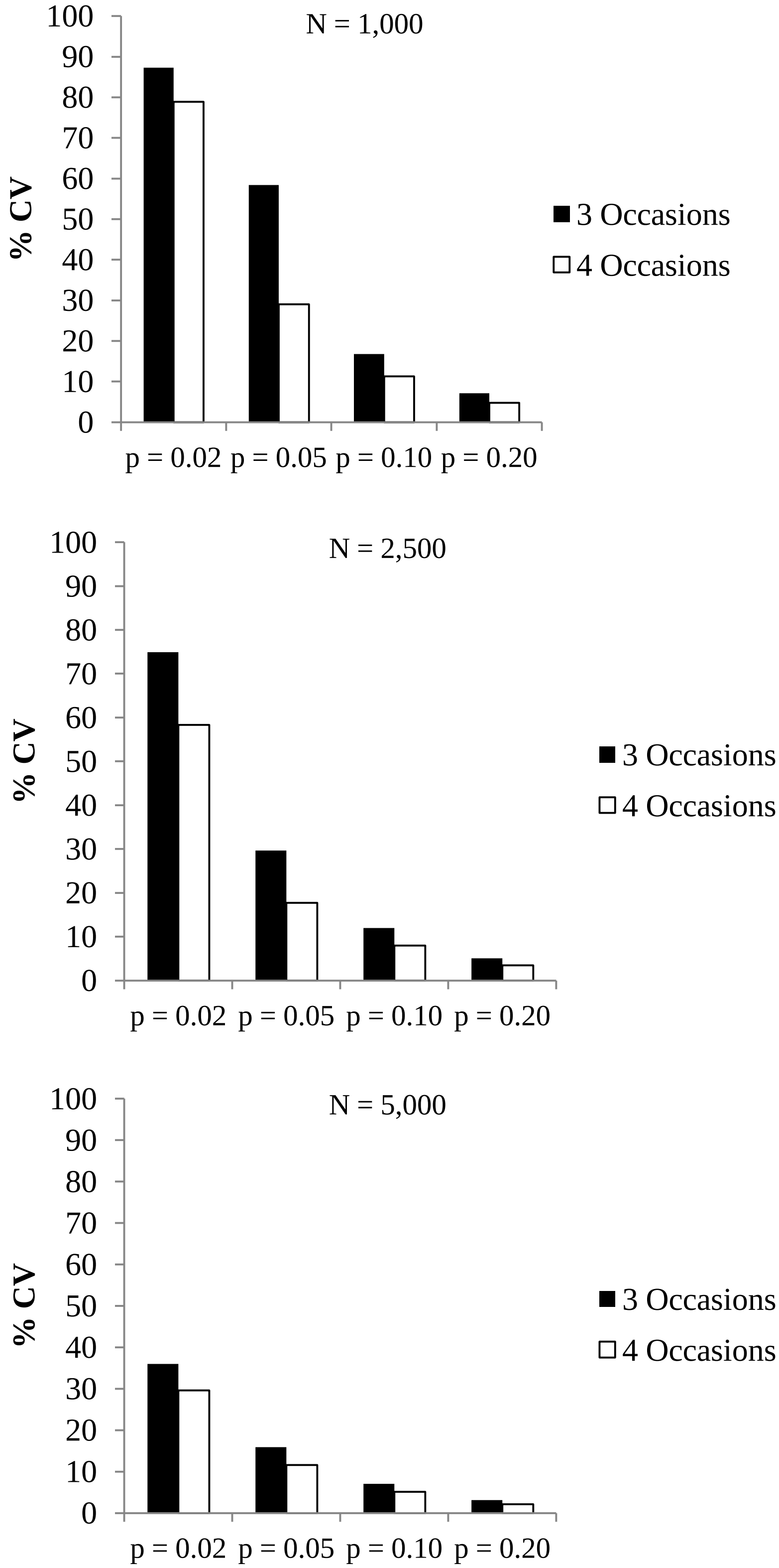

Figure 13. Simulation results that depict the precision (as % coefficient of variation, [SE/estimate]*100) of abundance estimates under different scenarios of probabilities of capture per pass 0.02, 0.05, 0.10, and 0.20, using three or four sampling occasions, where true population size was 1,000 (top panel), 2,500 (middle panel), or 5,000 (bottom) fish (n = 1,000 simulations).