BIOLOGICAL ASSESSMENT OF A ONE TIME TEST OF BEACH/HABITAT-BUILDING FLOW FROM GLEN CANYON DAM SPRING 1996

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BIOLOGICAL ASSESSMENT OF A ONE TIME TEST OF BEACH/HABITAT-BUILDING FLOW FROM GLEN CANYON DAM SPRING 1996

INTRODUCTION

This biological assessment was prepared in compliance with Section 7 of the Endangered Species Act of 1973, as amended. The Bureau of Reclamation proposes to conduct a one time test of a beach/habitat-building flow from Glen Canyon Dam (test flow) in the spring of 1996 to allow for collection of data for use in determining future dam operations.

Six species identified as threatened or endangered are addressed in this assessment: Kanab ambersnail (Oxyloma haydeni kanabensis), razorback sucker (Xyrauchen texanus), humpback chub (Gila cypha), bald eagle (Haliaeetus leucocephalus), peregrine falcon (Falco peregrinus), and southwestern willow flycatcher (Empidonax trailii extimus). The list of species is based on discussions with the Fish and Wildlife Service (Service) and previous consultations. Impacts of the test flow on endangered species may result from: physical displacement, injury, or death; loss or alteration of habitat; reduction in food availability; or alteration of interactions with other species (Bureau of Reclamation 1995). This biological assessment summarizes the distribution and abundance, life requisites, and potential impacts of the test flow on these species and their habitats.

Background

Glen Canyon Dam is located in north-central Arizona in Coconino County one mile west of Page, Arizona (See Fig. 1). Glen Canyon Dam regulates the Colorado River's flow through lower Glen Canyon and all of Grand Canyon, and is managed in accordance with the Colorado River Compact and the Colorado River Storage Project Act (1956).

In the Colorado River corridor, flow regulation has resulted in erosion and depletion of lateral sand deposits, particularly in narrow reaches of the river, loss of 50% of the native fish species and other changes in the river ecosystem. These impacts focused public and scientific attention on the need for modified operation of Glen Canyon Dam, resulting in a recently completed Environmental Impact Statement (GCD-EIS, Bureau of Reclamation 1995).

The Grand Canyon Protection Act of 1992 (P.L. 102-575) requires the Secretary of the Interior to operate Glen Canyon Dam "...in such a manner as to protect, mitigate adverse impacts to, and improve the values for which Grand Canyon National Park and Glen Canyon National Recreation Area were established, including, but not limited to natural and cultural resources and visitor use."

[Section 1802(a)]. In concert, the GCD-EIS recommends a Preferred Alternative that preserves a mass balance of sand, and therefore fisheries habitats, in the main channel.

The GCD-EIS recommends a modified low fluctuating flow alternative (MLFF) as the Preferred Alternative. The MLFF has a minimum release of 5,000 cubic feet per second (cfs) between 7:00 p.m. and 7:00 a.m. (at night), and 8,000 cfs as a minimum release from 7:00 a.m. to 7:00 p.m. (day time flows). The maximum release is 25,000 cfs, with a maximum daily flow range of 5,000 cfs, 6,000 cfs or 8,000 cfs, depending on the monthly release volume (during low, medium or high volume months, respectively). Ramping rates under the Preferred Alternative are 4,000 cfs/hr up and 1,500 cfs/hr down. The Preferred Alternative also provides for a 1 to 2 week-long beach/habitat-building flow. This high, steady, release will be used to evaluate the applicability of high flows as a strategy for future adaptive management of the Colorado River.

Flooding was an essential characteristic of the pre-dam river. Current dam operations have virtually eliminated this dynamic element of the natural ecological process. Although a program of reduced daily flow fluctuation (Interim Operations) was implemented in 1991 to conserve a mass balance of tributary-derived sand, occasional high flows are thought to be required to restore high elevation sand deposits and characteristic aquatic habitats, such as backwaters. Backwaters are used as rearing habitats by young native fish. The proposed test flow will improve understanding of how flooding influences ecosystem geomorphology, biological and cultural (archeological and traditional land use) resources, and ecosystem processes. Data collected during the test flow will demonstrate the extent to which planned flooding can be used as an ecosystem management tool.

DESCRIPTION OF THE TEST FLOW

Description of Flow

The test flow will begin on or about 22 March, 1996 (see Attachment A). The first 4 days will consist of a constant 8,000 cfs flow. On 26 March, 1996, discharge will be increased at an upramping rate of 4,000 cfs/hr until a flow of 45,000 cfs is reached. Flows will be held essentially constant at 45,000 cfs for 7 days (until 2 April, 1996), with flow changes less than +/- 1000 cfs. Discharge will then be decreased to 8,000 cfs in the following manner: (1) Between flows of 45,000 cfs and 35,000 cfs, the down-ramping rate will be 1,500 cfs/hr; (2) Between flows of 35,000 cfs and 20,000 cfs, the down-ramping rate will be 1,000 cfs/hr; and (3) Between 20,000 cfs and 8,000 cfs, the down-ramping rate will be 500 cfs/hr. Discharge will be maintained at +/- 8,000 cfs for 4 days (through 7 April, 1996). It is believed that this staggered down-ramp will more closely mimic the reduction of flows after a natural flood. The 8,000 cfs constant flows preceding and following the 45,000 cfs release will permit aerial photography, and, on-the-ground evaluation of sedimentation patterns and impacts to river resources. Interim Operations will resume on or about 8 April, 1996.

Rational for Conducting the Test Flow in a March/April Time Frame

The timing of the test flow was considered in detail. Specifically, the time frame for the test flow was selected to reduce impacts on river resources, by conducting the test flow: 1) prior to native fish (especially humpback chub) spawning, or larval dispersal; 2) after the peak of rainbow trout spawning at Lees Ferry; 3) several weeks after the peak concentrations of wintering bald eagle and waterfowl 4) prior to the Neotropical bird and waterfowl breeding seasons; 5) during a month in which high flows occurred in the pre-dam era; 6) prior to the peak release of non-native saltcedar (Tamarix ramosissima) seeds in late April and May, to allow riparian soils time to desiccate after the test flow, thereby reducing saltcedar germination; and 7) prior to the recreational river rafting season.

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KANAB AMBERSNAIL SPECIES ACCOUNT

Distribution and Abundance

Kanab ambersnail (KAS; Succineidae: Oxyloma haydeni kanabensis Pilsbry 1948), is a federally endangered landsnail that was proposed for emergency listing (England 1991a, 1991b) and officially listed in 1992 (England 1992). Fossil Oxyloma shells have been recovered from sediments in Grand Gulch (lower San Juan River) that date to 9,200 years ago (Kerns 1993). Living KAS were first collected by J.H. Ferriss in 1909 near Kanab, Utah in seep vegetation (Ferriss 1910, Pilsbry and Ferriss 1911, Pilsbry 1948). Extant populations of KAS are presently known to occur at two southwestern springs: one at Three Lakes, near Kanab Utah, and the other at Vaseys Paradise, a spring at Colorado River Mile 31.5R, in Grand Canyon, Arizona (Spamer and Bogan 1993a, 1993b). Two populations formerly occurred in the Kanab area, but one population was extirpated by desiccation of its habitat. The remaining Utah population at Three Lakes occurs at several, small spring-fed ponds on cattail (Typha sp.; Clarke 1991). The Three Lakes site is privately-owned and the land owner is commercially developing the property.

KAS were first collected at Vaseys Paradise in 1991 (Blinn et al., 1992, Spamer and Bogan, 1993), and an interagency team lead by the Bureau of Reclamation examined KAS ecology there in 1995 (Stevens et al., 1995). Vaseys Paradise is a popular water source and attraction site for Colorado River rafters; however access is limited by the dense cover of poison ivy (Toxicodendron rydbergii) and the nearly vertical terrain (Stevens, et al., 1995). Within Grand Canyon, KAS is apparently restricted to Vaseys Paradise: no KAS were observed at 81 other Grand Canyon springs surveyed from 1991 to 1995. Rematched historic photographs of Vaseys Paradise (e.g. Turner and Karpiscak 1980:58-59) reveal that vegetative cover has increased greatly at lower stage elevations since completion of Glen Canyon Dam.

Stevens, et al. defined primary habitat at Vaseys Paradise as crimson monkey-flower (Mimulus cardinalis) and non-native water-cress (Nasturtium officinale); and secondary, or marginal, habitat is defined as patches of other riparian vegetation that are not dominated by monkey-flower or water-cress, and not used by KAS. Land surveys in 1995 revealed rapid changes in vegetation cover over the growing season, with 5.9% to 9.3% of the primary habitat occurring below the 33,000 cfs stage, and 11.1% to 16.1% occurring below the 45,000 cfs stage. The total area of primary habitat was 0.22 acres, secondary habitat area was 0.22 acres, and the total vegetated area of the spring was 0.44 acres in June, 1995.

The total estimated Vaseys Paradise KAS population rose from 18,476 snails in March up to as many as 104,000 snails in September, 1995 as reproduction took place in mid-summer (Stevens et al., 1995). The proportion of the total estimated KAS population occurring below the 33,000 cfs stage rose from 1.0% in March to 7.3% in September, and that occurring below the 45,000 cfs stage was 3.3% in March, 11.4% in June and 16.4% in September, 1995.

Life Requisites

KAS occurs on cattail (Typha sp.) at Three Lakes, near Kanab, Utah, and cattail is the dominant macrophyte at that locality. In contrast, Vaseys Paradise is a fast-flowing, cool, dolomitic-type spring, with abundant wetland and phreatophyte vegetation, including native crimson monkey-flower, poison ivy and non-native water-cress. Crimson monkey-flower and non-native water-cress are perennial aquatic wetland or hydrophytes (Kearney and Peebles 1960), and KAS was almost completely restricted to those two species along the edges of the Vaseys Paradise stream (Stevens et al., 1995). KAS was rare to absent on other plant species and substrata in 1995. Introduction of water-cress provided KAS with an alternate host plant, and completion of Glen Canyon Dam increased overall primary habitat area at Vaseys Paradise by approximately 40%.

Demographic analyses based on size class distribution indicated that KAS is essentially an annual species, with much of the population maturing and reproducing in mid-summer (July and August), and most snails over-wintering as small size classes (Stevens et al., 1995). Loose, gelatinous egg masses were observed on the undersides of moist to wet live stems, on the roots of water-cress, and on dead or decadent stems of crimson monkey-flower in August, 1995. No data on egg development or emergence success are available.

KAS at Vaseys Paradise were parasitized by a trematode, tentatively identified as Leucochloridium sp., with 8.3 to 9.5% of the mature snails expressing sporocysts in August, 1995 (Stevens et al., 1995). Potential vertebrate predators of KAS at Vaseys Paradise include rainbow trout (Oncorhynchus mykiss) in the stream mouth, summer breeding Says and black phoebe (Sayornis sayi and S. nigricans), canyon wren (Catherpes mexicanus), and winter resident American dipper (Cinclus mexicanus).

Impacts of the test flow

In a December 21, 1994, Final Biological Opinion (BO) the Service evaluated impacts to KAS from the operation of Glen Canyon Dam according to operating and other criteria of the MLFF contained in the GCD-EIS. They determined implementation of the MLFF would not jeopardize the continued existence of the KAS. This opinion also supported the concept of a beach/habitat building flow of 40,000 to 45,000 cfs, which is part of MLFF. However, the opinion clearly articulates that incidental take of KAS will be exceeded if more than 10% of the occupied habitat in Grand Canyon will be inundated by high flows or a controlled flood. Considering the Service previously determined the MLFF (including the beach/habitat-building flow) would not jeopardize the KAS, this assessment examines the probability of exceeding the incidental take level during the test flow.

Introduction of non-native water-cress and construction and operation of Glen Canyon Dam increased the primary KAS habitat area by more than 40%, and resulted in an increase in the snail population. The KAS population at Vaseys Paradise survived and recovered from innumerable similar and higher flows during the pre-dam era, and has survived six flows in excess of 45,000 cfs during the post-dam era (i.e., 1965, 1980, and 1983-1986). Short-term reduction in primary habitat area by scouring flows does not appear to affect the long-term integrity of the KAS population.

The lowest KAS population observed during 1995 by Stevens, et al., occurred in March, 1995. Thus, natural winter mortality from November, 1995 to March, 1996 may reduce the percentage of take on KAS considerably. This would reduce the potential number of individuals lost during the test flow, and the impact to the Vaseys Paradise KAS population.

Test flows are scheduled to begin on or about 22 March, 1996. Stevens, et al. found 6.3% of KAS examined on March 30, 1995, retained mucosal plugs, indicating that the population was just completing emergence from winter dormancy at that time. It is reasonable to assume similar conditions will exist in March, 1996 if weather patterns are similar, although early warming may affect the emergence of the KAS.

The following analysis is based on the data collected by Stevens, et al. If KAS distribution in March, 1996, is equivalent with March, 1995, only 3.3 % of the KAS population will be affected by the test flow. If primary habitat in March 1996 is equivalent to primary habitat in March, 1995, 14.6% of the habitat will be assumed lost due to scour during the test flow, which exceeds the level of incidental take.

In the Stevens, et al. report, potential loss of habitat and KAS was estimated to be as much as 16.1% of the primary KAS habitat and 11.4 to 16.4% of the estimated population. This estimate was based on the summer, 1995 data. This data is displayed in a table in the report as having been collected in June, 1995. The actual amount of habitat and KAS affected by the test flow would most accurately be compared to the March, 1995 survey data considering that once the test flow begins, vegetation and KAS numbers below the 45,000 cfs stage will not increase. Appendix A indicated that 16.2 M^2 of the primary habitat in March consisted of mixed patches of Carex, Polygonum and other species, which is defined as secondary habitat, which rarely contains KAS. This portion of the habitat should not be defined as occupied habitat in calculating the incidental take level. If these patches of vegetation were eliminated form the calculation of primary habitat, only 12.8% of the primary habitat is lost.

Conclusion

Considering that: 1) Stevens, et al concluded the Vaseys Paradise population of KAS appears to be relatively large and self-sustaining; 2) more than 40% of the present primary KAS habitat at Vaseys Paradise lies below the pre-dam 10-year flood stage of 3540 m^3/s, most of which is new, post-dam habitat; 3) KAS have survived numerous flood events; 4) estimated loss KAS habitat contains only 3.3 % of the KAS population; 5) estimated habitat loss is 12.8 to 14.6% of primary habitat; and, 6) vegetation will have the opportunity to re-colonize the scoured area prior to and during the expansion of the population due to reproduction; we request the Service reconsider the allowable incidental take in terms of a percentage of the population of KAS, as opposed to a per cent of KAS habitat. We acknowledge that take of any amount of an endangered species a matter of serious concern. Reclamation will monitor the population and its habitat both pre- and post-test flow, as required by the

1994 BO incidental take statement, to assist in defining the species' response to the event and in refining a take level. We believe the benefits of the test flow to endangered the humpback chub and its mainstem habitat, and to other ecosystem processes should also be given full consideration.

The incidental take statement and the Stevens, et al. report consider relocation of snails to a position within the habitat above the 45,000 cfs level. We do not believe moving of snails is acceptable as a long-term management strategy. The success of such an effort is hard to predict. Negative impacts to the KAS may result from disturbance due to handling and moving and some KAS may be placed into already occupied habitat resulting in increased competition. If snails are only emerging from winter dormancy their ability to re-orient may be impaired. Self sustaining population(s) of KAS would not require frequent human intervention to maintain themselves in the event of habitat disturbance such as a spring flood, and they have survived flood events larger than 45,000 cfs six times since the dam was closed. We favor focusing resources on understanding the ecology of the species and refining management strategies.

HUMPBACK CHUB SPECIES ACCOUNT

Distribution and Abundance

The humpback chub (HBC; Cyprinidae: Gila cypha) is an endemic fish species in the Colorado River basin. The HBC was taxonomically described by Miller (1946), and was listed as an endangered species in 1968. Stream alteration, including flow modification, diversion for irrigation, channelization, and introduction of non-native fish species, have been suggested as responsible for declining populations of HBC throughout the Colorado River basin (Valdez 1995). Five HBC populations remain in canyon-bound reaches of the upper Colorado River basin: Black Rocks (upper Colorado River), Westwater Canyon (upper Colorado River), Cataract Canyon, Desolation/Gray canyons (Green River) and in the Yampa River.

The Grand Canyon population is the only successfully reproducing HBC population in the lower Colorado River basin (Kaeding and Zimmerman 1983; Valdez 1995). Valdez (1995) identified nine distinct aggregations in the mainstream Colorado River downstream from Glen Canyon Dam, including: 30-Mile, Little Colorado River inflows, Lava/Chuar to Hance Rapids, Bright Angel Creek mouth, Shinumo Creek mouth, Stephens Aisle, Middle Granite Gorge, Havasu Creek mouth and Pumpkin Spring. From 3000 to 3500 adult HBC occupy the mainstream Colorado River, and these are largely concentrated within ± 4.2 miles of the mouth of the Little Colorado River (Mile 61; Valdez 1995), the largest sub-population.

Habitat use by HBC varied between age classes and by time of day. Young HBC in the mainstream commonly use return current channels and other backwater habitats (Maddux et al., 1987); however, HBC use of backwater habitats in Grand Canyon has been compromised by fluctuating flows and cold-stenothermic releases which reduce warming and create unstable conditions. In addition, backwater habitat area has been reduced and backwaters have aggraded through siltation under Interim Operations in Grand Canyon (McGuinn-Robbins 1995).

Subadult HBC in the Colorado River mainstream often use irregular shorelines as habitat, and adult HBC often occur in or near eddies (Valdez 1995). Adult radio-tagged HBC demonstrated a consistent pattern of greater near-surface activity during the spawning season and at night, and day-night differences decreased during turbid flows (Valdez 1995).

Life Requisites

The life history and ecology of HBC in Grand Canyon has been intensively studied (Suttkus and Clemmer 1977, Kaeding and Zimmerman 1983, Carothers and Minckley 1981, Maddux et al., 1987, Gorman 1994, Valdez 1995). A key issue is lack of recruitment to the adult population as reflected by low survivorship of young fish (Valdez, 1995). Individual adult HBC demonstrate high microsite fidelity (Valdez 1995), but young HBC may drift for relatively long distances (Tuegel 1995). Mainstream Colorado River HBC in Grand Canyon spawn primarily in the lower nine miles of the Little Colorado River from March through May.

Adult fish initially stage for spawning runs in large eddies in February and March, and make spawning runs that average 17 days into the LCR from March through May, as LCR flows decrease, warm and clear (Valdez 1995). Spawning runs of up to 25 miles have been reported for this species. After spawning, many adult chub return to specific microsites in the mainstream. Young HBC remain in the Little Colorado River, or move into the mainstream where mortality due to thermal stress (Lupher and Clarkson 1993) and predators (Valdez 1995) is perceived to be extremely high. During the summer the young HBC that survive in the mainstream occupy low-velocity, vegetated shoreline

habitats; however, low survivorship over the year virtually eliminates the young of the year HBC in the mainstream. Therefore, few if any HBC spawned during the previous year are present in the mainstream in March.

Limited breeding of HBC occurs among other sub-populations in the Colorado River. Valdez (1995) documented limited spawning success at 30-Mile Spring in upper Marble Canyon, and rare young HBC have been documented at Kanab Creek; however, these sightings are rare compared with the reproductive success of those HBC that spawn in the Little Colorado River.

Dietary analyses reveal HBC to be opportunistic feeders, selectively feeding on aquatic and terrestrial invertebrates (Valdez 1995). HBC diet changes over the course of the year in response to food availability and turbidity-related decreases in benthic standing biomass over distance downstream from Glen Canyon Dam (Blinn et al., 1992). Non-native Gammarus lacustris occasionally comprise a large proportion of HBC diet, and Gammarus selectively feeds on epiphytes (i.e., diatoms) associated with Cladophora glomerata, the dominant alga in the upper reaches where clearwater conditions often prevail.

Impacts of the test flow

The test flow is not likely to adversely affect the HBC. The timing of the high flow event has been specifically planned to limit impacts to 1995-spawned HBC. Furthermore, the high flow will occur prior to significant spawning-related movement of adult chub into the Little Colorado River, and will take place at a time of year when warming of backwaters is not significant. Some loss of young chub may occur; however, 1995 was not a remarkably strong recruitment year (Tuegel 1995). Therefore, the impacts of the high flow are unlikely to be measurable. High flow events of the magnitude planned here occurred virtually every year in pre-dam time, and adult HBC appear to be well-adapted to survival of such events.

The Reasonable and Prudent alternative of the 1994 BO includes habitat/beach building flows, however, the Service determined some HBC would be taken during a high flow event. Their discussion of incidental take considers testing and studies to determine impacts of flows on young humpback. One goal of the test flow is redistribution of channel bottom sediment to the channel margins to establish and maintain habitats for young life stages of HBC in the mainstem. This hypothesis will be examined through the test flow.

The test flow may benefit HBC directly and indirectly. The high flow may serve as a spawning cue for movement into the mouth of the Little Colorado River. High flows are expected to result in additional drift, which may provide additional food resources for staging HBC. Although some scour of the benthos is anticipated by the event, rejuvenation of return current channels and other mainstream backwater habitats should provide additional recruitment habitat for young HBC in the summer of 1996. Backwater formation will occur through the test flow prior to dispersal of young HBC from the Little Colorado River into the mainstream in the late spring and summer, 1996.

In conclusion, the test flow may affect HBC, but is not likely to adversely affect HBC populations.

RAZORBACK SUCKER SPECIES ACCOUNT

Distribution and Abundance

Razorback sucker (RBS; Catostomidae: Xyrauchen texanus) is a widely distributed, endemic, warm water Colorado River fish. RBS formerly occurred throughout the Colorado River, but has declined since 1930 with the regulation of the Colorado River (Dill 1944, Minckley 1991). The decline of RBS has been attributed to thermal regime changes, altered spawning habitat and introduction of non-native fish species, which have cumulatively resulted in wide-scale recruitment failure (Bestgen 1990, Minckley 1991). This species was listed as an endangered species by the U.S. Fish and Wildlife Service in 1991 (U.S. Fish and Wildlife Service 1991).

The largest RBS population in the Lower Colorado River Basin exists in Lake Mohave, where it was estimated to be approximately 60,000 fish in 1989 (Marsh and Minckley 1989). Other, smaller lower basin RBS populations occur in Lake Mead, downstream from Hoover Dam, and in Senator Wash Reservoir. In the Upper Colorado River Basin, RBS occur regularly in the upper Green and lower Yampa rivers, and individual RBS have been collected at rare intervals in the Colorado River near Grand Junction, Colorado, and in the major tributary arms of Lake Powell. RBS are long-lived (20 to 50 yr), but most wild-caught RBS are old individuals, and recruitment failure may lead to the rapid demise of this species (McCarthy and Minckley 1987, Minckley 1991). Experimental releases in the Upper Basin, and attempts to propagate RBS in Lower Basin reservoirs are encouraging, but the mainstream Colorado River populations continue to decline.

RBS are extremely rare in Grand Canyon. Recent observations are those of Carothers and Minckley (1981) who reported four RBS from the Paria River in 1978-1979; Maddux et al. (1988) reported one blind female RBS at Upper Bass (Colorado River Mile 107.5) in 1984; and Minckley (1991) reported records of 5 additional RBS captured in the lower Little Colorado River from 1989-1990. Putative hybrids between flannelmouth sucker (Catostomus latipinnis) and RBS have been reported from the Little Colorado River (Suttkus and Clemmer 1979, Carothers and Minckley 1981).

Life Requisites

RBS are generally associated with calm river reaches, particularly man-made lakes (Tyus 1987); however, river spawning typically occurs in riffle habitats over gravel and cobble substrata (Mueller 1989). Larval RBS drift downstream from the spawning habitat, and concentrate in warm, low-velocity areas (e.g. flooded bottoms). These areas also support post-larval RBS, and channel and mid-stream river habitats floored by fine-grained alluvium are important to subsequent RBS life stages (Minckley 1983, Tyus and Karp 1989, Minckley 1991). Springtime concentrations of adult RBS have been noted in side-channels, off-channel impoundments, and in tributaries (Bestgen 1990, Minckley 1991). The optimal thermal range for RBS is 72 to 77°F (Bulkley and Pimentel 1983); however, RBS occur in widely varying temperatures. RBS habitats in the Upper Colorado River Basin are ice-covered during winter, while the temperatures of mainstream habitats in the Lower Colorado River exceed 90°F (Dill 1944).

RBS diet varies by age class and habitat type, but few data are available on the diet of larval and juvenile RBS (Bestgen 1990). Larval RBS are known to feed on phytoplankton and zooplankton, and (in fluvial habitats) on chironomid larvae. Adult RBS in lentic habitats feed on benthic and planktonic algae and macroinvertebrates, while adult RBS in rivers feed primarily on benthic algae and invertebrates.

RBS spawn earlier in the season than do most other native, warm water Colorado River fish (Minckley 1973, 1991). Lake Mohave RBS spawn from November into May, with the peak of spawning activity between January and March when water temperatures were stable (50 to 54°F) or rising from 50 to 59°F (Bozek et al., 1984). In riverine situations in the Upper Basin, RBS begin spawning on the rising limb of the spring (April-May) hydrograph, and spawn for an extended period through the spring runoff. Although it occurred throughout the day, spawning activity is most intense at dusk.

RBS are susceptible to parasitic bacteria, protozoa and copepods. Minckley (1983) and others have reported a high incidence of blindness in one or both eyes; however the reasons for this condition are not clear (Bestgen 1990).

Impacts of the test flow

The test flow will have no effect on the remaining RBS population in the Colorado River in Grand Canyon. Remaining RBS are mature or senile fish, which survived comparable or higher mainstream flows in 1965, 1973, 1980, and 1983-1986, and possibly during the pre-dam era. The older fish are capable of finding suitable refugia, and the lack of recruitment of this species indicates that no young razorback sucker are in the system or at risk during the test flow. Because RBS spawn somewhat earlier than HBC, the early date of the test flow may stimulate some additional RBS spawning activity.

BALD EAGLE SPECIES ACCOUNT

Distribution and Abundance

The bald eagle (Accipitridae: Haliaeetus leucocephalus) has suffered population declines from habitat loss, mortality from shooting and poisoning, and reduced reproductive success from ingestion of contaminants (U.S. Fish and Wildlife Service 1983), and it was recognized as a threatened and declining species in 1967. This species occurs throughout North America from Alaska to northern Mexico, and commonly breeds in the northern portion of its range (Stahlmaster 1987). Although bald eagles face numerous threats throughout the 48 states, they have recovered from dramatic population declines over the past several decades. Consequently, the U.S. Fish and Wildlife Service downlisted the bald eagle from endangered to threatened status (U.S. Fish and Wildlife Service 1995).

A wintering bald eagle concentration was first observed in Grand Canyon in the early 1980's and has increased dramatically since 1985 (Brown et al., 1989, Brown and Stevens 1991, Brown and Stevens 1992). The wintering bald eagle population has been monitored since 1988, and it occurs throughout the upper half of the Grand Canyon (in Marble Canyon) and on both Lakes Powell and Mead. Density of the Grand Canyon bald eagles during the winter peak (in late February and early March) ranged from 13 to 24 birds between Glen Canyon Dam and the Little Colorado River confluence from 1993 to 1995 (Sogge et al., 1995). A concentration of wintering bald eagles occurs in late February at the mouth of Nankoweap Creek, where bald eagles forage on spawning rainbow trout (Brown et al., 1989, Brown 1993). Bald eagle density there ranged from 6 in 1987 to 26 in 1990, and 18 bald eagles occurred at Nankoweap Creek in 1995 (Sogge et al., 1995). Eagle density was correlated with trout density in the lower 0.5 km of Nankoweap Creek, and trout density was correlated with tributary stream water temperature. Territorial behavior, but no breeding activity, has been detected in Grand Canyon.

Life Requisites

Bald eagles are opportunistic feeders, preying on fish, waterfowl, rabbit and road-killed game (Stahlmaster 1987). Wintering bald eagles frequent rivers, reservoirs and lakes, including western reservoirs (Detrich 1987), and their distribution is dependent on prey availability, perch suitability, weather and human disturbance intensity (Ohmart and Sell 1980).

At Nankoweap Creek in Grand Canyon, wintering bald eagles preferentially capture rainbow trout in the shallow creek, rather than in the mainstream where foraging success is low (Brown 1993, Sogge et al., 1995). Bald eagles at Nankoweap Creek prefer roosting and feeding areas that are relatively free of vegetation. The eagle population there consists of all age classes, with considerable piracy and other interactions between individuals (Brown and Leibfried 1992). The ease and relative safety of foraging in Nankoweap Creek affords wintering bald eagles at Nankoweap Creek the opportunity to accumulate energy reserves needed for their long, northward migration flights and initiation of nesting.

Impacts of the test flow

Wintering and migrant bald eagles have largely left the Grand Canyon region by late March. The few remaining eagles forage opportunistically and may continue to catch trout in the mainstream. The rainbow trout will have concluded their spawning run in Nankoweap Creek by the time of the test flow, so remaining bald eagles will no longer have access to that food source.

The test flow will have no effect on the wintering bald eagle population in the Colorado River in Grand Canyon.

PEREGRINE FALCON SPECIES ACCOUNT

Distribution and Abundance

The peregrine falcon (Falconidae: _Falco peregrinus_) is a federally listed endangered raptor, which has declined dramatically as a result of the biological concentration of pesticide residues in prey species, and resulting eggshell thinning (U.S. Fish and Wildlife Service 1984) and the increase in food base in Grand Canyon. The population in the Rocky Mountain/southwest region declined from 180 pairs prior to 1975 to 55 pairs in 1983, largely as a result of DDT/DDE thinning of eggshells (U.S. Fish and Wildlife 1984).

The Grand Canyon peregrine population was low in the mid-1970's (Ellis and Monson 1989), but increased dramatically in the 1980's, following recovery efforts by the U.S. Fish and Wildlife Service (1984). At present, the Grand Canyon supports the largest breeding population of peregrine falcons in the coterminous United States (Brown 1991a). Surveys for nesting peregrine falcons in 1988 and 1989 revealed 28 and 58 pairs, in 15% and 24% of the park, respectively. Habitat-based estimation of the potential number of peregrine falcons in Grand Canyon suggested that as many as 96 pair existed in Grand Canyon in 1989.

Life Requisites

Peregrine falcons feed on more than 20 species of birds and several small mammals (Porter and White 1973). Hunting areas included marshes or narrow tongues of streamside vegetation, and peregrine falcons may forage up to 17 mi from nest sites. Peregrine falcon diet at nest sites in national parks in southern Utah included small and medium-sized birds, especially including white-throated swifts, large shorebirds and Clark's nutcracker (Burnham 1987).

In Grand Canyon, peregrine falcons feed on waterfowl, swifts, swallows and bats (Brown 1991a, Stevens personal communication), which feed on invertebrate species (especially Diptera) that emerge out of the Colorado River (Blinn et al., 1992). Therefore, dam operations that influence aquatic macro-invertebrate populations exert indirect impacts on peregrine falcons.

Peregrine falcons breed up to 3,130 m elevation, typically on ledges on steep cliff faces (U.S. Fish and Wildlife Service 1984). The mean distance between nest sites along the South Rim of Grand Canyon varied from 3.5 to 5.0 linear miles, with minimum distances of 1.8 linear miles (Brown 1991a). The breeding season in Grand Canyon extends from February to July.

The primary reason for the national decline of the peregrine falcon population has been eggshell thinning from DDE and other environmental contaminants, which are biologically concentrated through the food chain. DDE sources to peregrine falcons are derived from their prey, many of which are migratory insectivores. Burnham (1987) reported that swifts, shorebirds, and other migratory insectivores contained 5.8 ppm DDE (wet weight), while mean DDE levels in granivorous migrants, such as grosbeaks and mourning doves, was only 0.14 ppm DDE. Peregrine eggshells from southern Utah parks from 1985 to 1987 were 21% thinner than those from the pre-DDE era, indicating poor viability of eggs (Burnham 1987). Brown (1991b) reported that peregrine falcon eggshells from the Grand Canyon in 1988 were 11.4% to 12.7% thinner than pre-DDE controls.

In addition to pesticide concentrations, competition with other raptors has been considered as a possible cause of peregrine falcon population declines; however, Porter and White (1973) examined peregrine and prairie falcon interactions and concluded that competition was not important.

Impacts of the test flow

Most wintering waterfowl on which peregrine falcons feed will have migrated from Grand Canyon by late March; however, mallard and late migrating gadwall and American widgeon are still likely to be common (Stevens, personal communication). Springtime food sources (swifts, swallows and bats) should be present in large numbers at that time of year (Stevens, personal communication), and are only indirectly influenced by dam operations. Therefore, peregrine falcons will not lack food resources during the proposed high release. The test flow will have no effect on peregrine falcons in the Colorado River downstream from Glen Canyon Dam.

SOUTHWESTERN WILLOW FLYCATCHER SPECIES ACCOUNT

Distribution and Abundance

The southwestern willow flycatcher (SWWF; Tyrannidae: Empidonax trailii extimus) is a Neotropical migrant with a broad breeding range, extending from Nova Scotia to British Columbia and south to Baja California. The SWWF is an obligate riparian insectivore (Hunter et al., 1987), preferring habitat near open water (Gorski 1969; Sogge 1995). The historic breeding range of the SWWF includes Arizona, New Mexico, southern California, and southern portions of Nevada, Utah, and perhaps southwestern Colorado, and extends east into western Texas (U.S. Fish and Wildlife Service 1993). It winters from Mexico to Panama, with historical accounts from Colombia (Phillips 1948). The SWWF is distinguished from other subspecies by distribution, morphology and color, nesting ecology, and possibly by song dialect (Phillips 1948, Aldrich 1953, King 1955, Sogge 1995).

Although never common, E. t. extimus population declines have been noted for nearly 50 years, corresponding with loss and modification of riparian habitats (Phillips 1948). Southwestern riparian ecosystems support a rich avian fauna (Johnson and Haight 1987) and habitat changes have resulted in reduction or extirpation of many avian species (Hunter et al., 1987). Modification and fragmentation of these systems through development and livestock grazing have precipitated devastating changes to SWWF populations. Destruction of native willow/cottonwood vegetation has provided opportunity for invasion by non-native plant species, notably saltcedar. Habitat fragmentation and modification has benefitted some southwestern avian species, especially cowbirds (Molothrus spp.), which parasitize SWWF nests, contributing to the precipitous population declines of SWWF (Brown 1994, Johnson and Sogge 1995, Sogge et al. 1995). SWWF habitat loss in Central and South America has also undoubtedly contributed to recent SWWF population declines.

The SWWF has been extirpated from much of its former range (Hunter et al., 1987) and experienced such a sharp reduction in abundance since 1950 that it was initially proposed (U.S. Fish and Wildlife Service 1992) and listed, with critical habitat, under the Endangered Species Act, on July 23, 1993 (U.S. Fish and Wildlife Service 1993). The SWWF is more rare than most other currently listed avian species (Unitt 1987). An estimated 300-500 breeding pairs remain in the United States, including 115 pairs in California and approximately 100 pairs in New Mexico (U.S. Fish and Wildlife Service 1993). Limited information exists for Colorado, Utah, Nevada, and Texas. It has been given endangered species status by the Game and Fish Departments in Arizona, New Mexico and California.

Arizona has experienced the sharpest decline in SWWF numbers. The SWWF formerly bred throughout the state at high and low elevations. For example, a 1931 breeding record exists from the south rim of the Grand Canyon (Brown et al., 1984). The former range of SWWF included the lower Colorado River, from which it has been extirpated. By 1987, the State population was estimated at less than 25 pairs (Unitt 1987), but much habitat was not surveyed. At least 52 territories or active nests were reported during extensive surveys in 1993 in Arizona (Muiznieks et al. 1994), and at least 62 active nests were located during a more thorough inventory in 1994 (Sferra et al., 1995).

The Grand Canyon population occurs between Colorado River Miles 47 and 54, and at River Mile 71 (Unitt 1987, Sogge et al., 1995). In its recent proposal the U.S. Fish and Wildlife Service (1993) designated the Colorado River from River Mile 39 to River Mile 71.5 as critical habitat. The boundary of this area includes the main river channel and associated side channels, backwaters, pools and marshes throughout the May-September breeding season, as well as areas within 100 meters of the edges of the surface water.

Nesting SWWF were common in Glen Canyon in the 1950's (Behle and Higgins 1959). This area was inundated by Lake Powell and no singing male SWWF were detected in a 1991 survey below Glen Canyon dam, although weather may have been a factor (Brown 1991a). Further down river, in the same survey, two pairs and nests were located at River Mile 50.7 and at River Mile 71.1 (Cardenas Marsh). In an earlier six-year study, Brown (1988) noted a brief population increase in the Grand Canyon from two in 1982, to a maximum of 11 (two in Cardenas Marsh), with a subsequent decline to seven in 1987. Only two pairs were noted in 1991 (Brown 1991b). Surveys in 1992 detected seven SWWF, three unpaired males and two breeding pairs in Cardenas Marsh (Sogge et al., 1995a). A total of five SWWF were detected in Grand Canyon in 1995, three territorial but non-breeding males and one breeding pair that fledged a single young (Sogge et al., 1995a). Male SWWF established territories between Colorado River miles 50.5 and 65.3, and the breeding pair nested at mile 50.5.

SWWF arrive in the Grand Canyon area in mid-May, but may be confused with another subspecies, the more common E.t. brewsteri, which migrates through to more northern breeding grounds (Aldrich 1951; Unitt 1987). E.t. brewsteri sings during migration, making sub-specific distinctions difficult until mid-June (Brown 1991b). Males arrive earlier than females and set up territories.

The characteristic territorial call is a "fitz-bew," most frequently heard in the morning before 10 AM (Tibbitts et al., 1994). The four subspecies may be differentiated by characteristics of this call.

SWWF return to wintering grounds in August and September (Brown 1991b), but neither migration routes nor wintering areas are well known. Birds call and perhaps defend foraging territories in Central America during winter, and winter movement may be tied to water availability (Gorski 1969). Threats to SWWF on the wintering grounds are undocumented, but habitat losses in Latin America may be a major factor in the decline of this species.

Life Requisites

SWWF are highly territorial. Nest building begins in May after breeding territories are established. The nest is placed in a fork or horizontal branch 1-5 meters above ground (Tibbetts et al. 1994). A clutch of three or four eggs is laid from late May through July (Unitt 1987), but in Grand Canyon two or three eggs (usually three) are usually laid (Sogge 1995). Breeding extends through July and singing ceases at the end of the breeding season.

After a 12-14 day incubation, nestlings spend 12 or 13 days in the nest before fledging (Brown 1988; Tibbetts et al., 1994). The breeding season (eggs or young in nest) along the Colorado River extends from early June to mid-July, but may extend into August. One clutch is typical, however re-nesting has been known to occur if the initial nest is destroyed or parasitized (Brown 1988).

Riparian modification, destruction and fragmentation provided new foraging habitat for brown-headed cowbirds (Molothrus ater) and populations of brown-headed cowbirds continue to expand (Hanka 1985, Harris 1991). Brood parasitism is currently the greatest threat to SWWF and probably many other Neotropical migrants as well (Bohning-Gaese et al., 1993; Sogge et al., 1995). Over half the nests in Brown's study (1988) contained brown-headed cowbird eggs. Cowbirds may remove prey eggs, their eggs hatch earlier, and the larger nestlings are more competitive in the nest. Cowbirds fledged from Sierra Nevada SWWF nests while SWWF nestlings died shortly after hatching (Flett and Sanders 1987). Brown-headed cowbirds occur extensively around mule corrals on the rim of the canyon and travel down to the Colorado River.

SWWF may remove cowbird eggs or more commonly abandon the nest if the parasite's eggs are deposited. The second nesting attempt is energetically expensive, requiring a new nest to be built (Sogge 1995), although Brown (1988) noted that a SWWF pair covered a cowbird egg with fresh nesting material and laid a new clutch. The second nest, already at a temporal disadvantage, is often parasitized as well. Cowbird parasitism could be largely responsible for the absence of SWWF in otherwise suitable habitat in the Grand Canyon (Unitt 1987). Bronzed cowbirds (Molothrus aenus) have recently been reported colonizing the Grand Canyon and represent another threat (Sogge 1995).

The SWWF in Grand Canyon is considered a habitat generalist, occupying sites of average vegetation height and density (Brown and Trossett 1989). The SWWF breeds and forages in dense, multistoried riparian vegetation near surface water or moist soil (Whitmore 1977, Sferra et al., 1995), along low gradient streams (Sogge 1995). Nesting in the Grand Canyon typically occurs in non-native Tamarix approximately 4-7 m tall (13-23 feet), with a dense volume of foliage 0-4 m from the ground (Tibbetts et al., 1994). SWWF commonly and preferentially nest in saltcedar in the Grand Canyon (Brown 1988), and nested in saltcedar in Glen Canyon before completion of the Glen Canyon Dam (Behle and Higgins 1959). Although habitat is not limiting in Grand Canyon (Brown and Trossett 1989), required patch size is not known.

Proximity to water is necessary and is correlated with food supplies. Little is known of SWWF food preferences but it is probably a generalist feeder. It typically hovers and gleans insects from foliage, or flycatches from conspicuous perches (Stevens personal communication). SWWF also forage on sandbars, backwaters, and at the waters edge in the Grand Canyon (Tibbetts et al., 1994).

Impacts of the test flow

The test flow will have no substantial direct effect on SWWF along the Colorado River downstream from Glen Canyon Dam because the birds do not establish territories until May. In Grand Canyon, SWWF generally nest in saltcedar trees. Nest trees typically lie above the 45,000 cfs stage. Saltcedar stands in which SWWF nest are unlikely to sustain direct damage from the flooding event. Stevens and Waring (1988) demonstrated that saltcedar is exceptionally tolerant of flooding in the Grand Canyon, persisting through many weeks of inundation. The saltcedar trees in which the SWWF presently nest survived the >92,600 cfs flows of 1983 (Stevens personal communication), and are therefore unlikely to be scoured by the much smaller flow planned in the test flow.

The wetlands and low-lying areas near SWWF nesting habitats are likely to be temporarily altered by the test flow (Stevens and Ayers 1992); however, the test flow is expected to rejuvenate riverside and wetland habitat. Impacts on SWWF food resources will be minimal because SWWF forage on adult, terrestrial (non-aquatic) flying invertebrates that are unlikely to be affected by the test flow, or will recover promptly after the event. Stevens (1985) reported that riparian plant-dwelling invertebrate populations increased rapidly following a flow comparable to the test flow in 1980 (U.S. Bureau of Reclamation 1990).

In conclusion, the test flow may affect the SWWF, but is not likely to adversely affect the species.

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