APPENDIX A

Biological Assessment

INTRODUCTION

The following biological assessment, when considered with the information provided in the accompanying environmental assessment, provides the analysis of the Proposed Action Alternative necessary to comply with Section 7 of the Endangered Species Act of 1973, as amended. In brief, the Bureau of Reclamation (Upper Colorado Region; Reclamation), the National Park Service (Glen Canyon National Recreation Area and Grand Canyon National Park), and the U.S. Geological Survey (Grand Canyon Monitoring and Research Center; GCMRC) propose to conduct experimental releases from Glen Canyon Dam (test flows) in combination with mechanical removal of non-native fish in 2003-2004.

The action agencies seek to provide long-term benefits to native fish and sediment-based resources, and to allow for collection of data for use in determining future dam operations. The Proposed Action Alternative has been developed under the auspices of the Glen Canyon Dam Adaptive Management Program through a cooperative effort among the Grand Canyon Monitoring and Research Center, the Adaptive Management Work Group, the Technical Work Group, the Science Advisors, and participating scientists. The proposed combination of experimental dam releases and non-native fish removal has been developed using knowledge gained in nearly 20 years of research and monitoring of resources in this reach of the Colorado River, first under the Glen Canyon Environmental Studies and now as part of the Adaptive Management Program. It is important to note that the Proposed Action is expected to produce an overall positive benefit to the ecosystem downstream of Glen Canyon Dam, including the endangered species, despite short-term minor impacts to some resources.

Six species identified as threatened or endangered are addressed in this biological assessment: Kanab ambersnail (*Oxyloma haydeni kanabensis*), humpback chub (*Gila cypha*), razorback sucker (*Xyrauchen texanus*), southwestern willow flycatcher (*Empidonax trailii extimus*), bald eagle (*Haliaeetus leucocephalus*), and California condor (*Gymnogyps californianus*). Critical habitat also is considered for humpback chub, razorback sucker, and southwestern willow flycatcher. The list of species is based on discussions with the Fish and Wildlife Service (Service) and previous consultations. Impacts of the test flow and mechanical removal 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 (Reclamation 1995). This assessment summarizes the distribution and abundance, life requisites, and potential impacts of the test flow and mechanical removal on these species and their habitats. Summary affect determinations are provided in table A-1.

Table A.1.—Affect determinations for federally listed species and their critical habitats under the Proposed Action.

Federally Listed Species/ Critical Habitat	Proposed Action
Kanab ambersnail	May affect, likely to adversely affect
Humpback chub w/critical habitat	May affect, likely to adversely affect
Razorback sucker w/critical habitat	May affect, not likely to adversely affect
SW willow flycatcher w/critical habitat	May affect, not likely to adversely affect
Bald eagle	May affect, likely to adversely affect
California condor	May affect, not likely to adversely affect

Kanab Ambersnail Species Account

Experimental releases under the Proposed Action include steady releases, fluctuating releases, and high releases at and above power plant capacity. Depending on the scenario, some of the four proposed flows may occur at more than one time of year. Thus, it is appropriate to evaluate the effects of each of the flows, as they differ in hydrology and in time.

Distribution and Abundance

The genus *Oxyloma* has a broad distribution (North America, Europe and South Africa), but the taxonomy, which previously has been based entirely on internal anatomy and shell morphology, is being revisited through molecular genetic techniques. Two species presently are recognized in the southwestern U.S., *O. retusa* in New Mexico and *O. haydeni* in Arizona and Utah. Within *O. haydeni* there are two subspecies, the Niobrara ambersnail (*O. h. haydeni*) and the Kanab ambersnail (*O. h. kanabensis*). Both are found in Arizona and Utah. Harris and Hubricht (1982) identified both subspecies, as *O. haydeni* and *O. kanabensis*, as occurring in Alberta, Canada, however, subsequent genetic analyses of individuals from these populations suggest they are not closely related to the Arizona and Utah populations (Stevens et al. 2000).

Kanab ambersnail (KAS: Succineidae: *Oxyloma haydeni kanabensis* Pilsbry 1948) is a federally endangered wetland snail that was proposed for emergency listing in 1991 (England 1991a, 1991b) and officially listed in 1992 (England 1992). In the southwestern U.S., extant populations of ambersnails morphologically and anatomically congruent with descriptions of KAS are presently known from two springs: one at Three Lakes, near Kanab, Utah, and the other at Vaseys Paradise, a spring and hanging garden 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. Recent genetic studies have revealed departures from identifications of subspecies and populations within the taxon currently recognized as *Oxyloma haydeni*, including the populations of *O. h. kanabensis* at Three Lakes and Vaseys Paradise. Results indicate that the Utah and Arizona populations considered to be the endangered subspecies are not as closely related as was indicated by morphological and anatomical comparisons (Miller et al. 2000, Stevens et al. 2000). The taxon at Vaseys Paradise may well be unique.

KAS was first collected at Vaseys Paradise in 1991 (Blinn et al. 1992) and an interagency team first examined KAS ecology there in 1995 (Stevens et al. 1997). 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 in much of the area (Stevens et al. 1997). 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.

Vaseys Paradise is characterized by a fast-flowing, cool, dolomitic-type spring, with abundant wetland and phreatophytic vegetation, including native crimson monkey-flower and poison ivy, and non-native water-cress. Crimson monkey-flower (*Mimulus cardinalis*) and water-cress (*Nasturtium officinale*) are perennial wetland plants or hydrophytes (Kearney and Peebles 1960). Within the Grand Canyon region, KAS apparently is restricted to Vaseys Paradise. No KAS were observed at more than 150 springs and seeps in tributary canyons to the Colorado River that were surveyed from 1991 to 2000 (Sorensen and Kubly 1997, 1998, Meretsky 2000, Meretsky and North 2000, Webb and Fridell 2000).

Stevens et al. (1997) defined primary habitat at Vaseys Paradise as crimson monkey-flower and non-native water-cress and secondary, or marginal, habitat as patches of other species of riparian vegetation that are little or not used by KAS. Surveys in 1995 revealed rapid changes in vegetation cover over the growing season, with 5.9 percent (%) to 9.3% of the primary habitat occurring below the 33,000 cubic feet per second (cfs) stage, and 11.2% to 16.1 % occurring below the 45,000 cfs stage. Area of primary habitat varied from 850 m^2 to 905 m^2 from March-September 1995. The same vegetation occupied from 7.0 to 12.5% of the area below 45,000 cfs from 1996-1999 following a 45,000 cfs beach/habitat building flow (BHBF) test (GCMRC 1999).

The total estimated Vaseys Paradise KAS population rose from approximately 18,500 snails in March 1995 up to 104,000 snails in September 1995 as reproduction took place in mid-summer (Stevens et al. 1997). 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. Subsequent surveys have reported population estimates of between approximately 5,000 and 52,000 individuals (Interagency Kanab Ambersnail Monitoring Team 1997, 1998 GCMRC 1999, Meretsky and Wegner 1999). Sorensen (2001) analyzed sampling and analytical techniques used for these estimates and concluded that overestimation of actual population size has occurred in monitoring reports. He pointed out that these errors make more difficult the assessment of risk to the population.

Introduction of non-native water-cress and construction and operation of Glen Canyon Dam increased the primary KAS habitat area by more than 40% over the pre-dam area and likely also allowed the increase of KAS by a similar amount. Plants forming primary habitat for KAS previously could not grow at levels below about 90,000 cfs because of recurring, scouring floods in the Colorado River at or above that level. The KAS population at Vaseys Paradise survived and recovered from innumerable similar and higher flows during the pre-dam era, and has survived seven flows in excess of 45,000 cfs during the post-dam era (i.e., 1965, 1980, 1983-1986, and 1996). Two full growing seasons were necessary for regrowth of the area of KAS habitat (vegetation) lost during the 1996 BHBF (Interagency Kanab Ambersnail Monitoring Team 1998). The composition of primary habitat remained different for a longer period as monkeyflower lagged behind water-cress in recovering. Population estimates for KAS have very wide confidence intervals and, therefore, are less useful for comparisons of loss and recovery. Meretsky and Wegner (1999) suspected that by spring 1998 the Vaseys Paradise KAS population had recovered to pre-BHBF levels, however, no measurements were made of individuals above the 100,000 cfs stage. On this basis, short-term reduction in primary habitat area by scouring flows does not appear to affect the long-term integrity of the KAS population.

In September 1998, Kanab ambersnails from Vaseys Paradise were translocated to three new sites in Grand Canyon National Park in an attempt to establish new populations fully protected under the Endangered Species Act. Supplementation of all populations occurred in 1999 and in two of the populations in 2000. The most recent report on the new populations indicates that two may not have been successful, but the third, at Upper Elves Chasm, is self-sustaining. Habitat at Upper Elves Chasm is all above the 100,000 cfs stage of the river and is predominately of monkeyflower and maidenhair fern (*Adiantum capillus-veneris*), with lesser amounts of sedges (*Carex aquatilis*), rushes (*Juncus* sp.), cattails (*Typha* sp.), water-cress, helleborine orchids (*Epipactis gigantea*) and grasses. Total habitat for KAS at this location is approximately 25 m^2. Population estimates have increased from approximately 130 in April 1999 to approximately 1900 in August 2001 (Nelson and Sorensen 2002).

Life Requisites

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. 1997, Nelson 2001). In some years with relatively warm winters, more than one reproductive period can occur. Loose, gelatinous egg masses are laid 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. Summer populations are comprised predominantly of maturing individuals. Adult mortality increases in late summer and autumn leaving the overwintering population dominated by subadults. KAS become dormant in winter, secreting a mucoid plug and attaching themselves to vegetative material or rock surfaces.

Nelson (2001) investigated aspects of the life cycle of KAS from Vaseys Paradise in the laboratory. He found variation in the life history between individuals reared in the laboratory on watercress and those raised on monkey-flower. Snail fecundity and growth rates were greater on water-cress than on monkey-flower, but hatching success, survivorship, size at first reproduction, and size at death were very similar.

Mortality of KAS from dam releases is caused by scouring and sediment deposition over the inundated habitat, and by drowning of ambersnails carried downstream in high flows. No evidence has been found for survival of any individuals carried downstream from Vaseys Paradise by high flows. There are several known sources of mortality for Vaseys Paradise KAS in addition to that caused by dam releases. KAS at Vaseys Paradise are 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. 1997). Potential vertebrate predators include rainbow trout (*Oncorhynchus mykiss*) in the stream mouth, summer breeding Say's and black phoebe (*Savornis savi* and *S. niaricans*), canyon wren (*Catherpes mexicanus*), winter resident American dipper (*Cinclus mexicanus*), and canyon mice (*Peromyscus crinitus*). A pour-out in the Redwall limestone above Vaseys Paradise drains a portion of the valley above and occasionally produces flash floods that scour and bury KAS habitat and animals. Mortality in laboratory populations occurs predominately in early life stages, whereas in field studies, up to 80% of observed mortality appears to occur during the period of winter dormancy. Extrapolations from laboratory populations indicated that KAS grown on monkey-flower would decline over time, whereas a population grown on water-cress would experience net growth.

Impacts of the Proposed Action Alternative: Experimental Test Flows

8,000 cfs Steady Flows.— KAS is affected by dam releases only when flows rise above the minimum stage at which it occurs. Our most recent information (B. Ralston, personal communication) places the flow for that stage at 17,000 cfs. Thus, 8,000 cfs steady releases will have no effect on KAS.

6,500 to 9,000 cfs Fluctuating Flows.— KAS is affected by dam releases only when flows rise above the minimum stage at which it occurs. Our most recent information (B. Ralston, personal communication) places the flow for that stage at 17,000 cfs. Thus, 6,500 to 9,000 cfs fluctuating releases will have no effect on KAS.

5,000-20,000 cfs Fluctuating Non-Native Fish Suppression Flows.— These releases will occur irrespective of whether sediment triggers are met. If they occur within two years of either a 31,000-33,000 cfs habitat maintenance flow or a 42,000-45,000 cfs high flow, they will have little effect because KAS habitat will have been scoured by the higher flows and not fully regrown. If they occur previous to these flows, or in a no sediment scenario year, there will be a loss of KAS habitat estimated to be less than 10%.

31,000-33,000 cfs Habitat Maintenance Flows.— Under the Proposed Action Alternative, these flows will occur in a year following that of the autumn sediment input scenario, and thus the effect of the July-December habitat maintenance flow will be diminished from what it would have otherwise been if the habitat had not been lost during the previous flow. Based on previous experience this diminishment will disappear in approximately two growing seasons. If KAS habitat has recovered from the preceding autumn sediment input scenario high flow, the habitat maintenance flows, which are to be purposefully combined with a sediment-laden tributary input of from 2,500 cfs to 12,000 cfs, will scour and cover with sediment between 10% and 17% of the KAS primary habitat at Vaseys Paradise.

42,000-45,000 cfs High Flows.— Experimental dam releases of 42,000-45,000 cfs could occur in January, February, or March under the Proposed Action Alternative. With sufficient sediment inputs they could occur in two successive years, a rate that exceeds the once in five year frequency identified in the Glen Canyon Dam EIS (Reclamation 1995). During the normal course of events in any given year, KAS primary habitat is expected to increase somewhat as new growth begins, probably by mid-February. The most proximate estimate for KAS habitat below the 45,000 cfs stage for this evaluation is the April 2002 estimate, which was 117 m^2 (B. Ralston, personal communication), slightly less than the 120 m^2 present in March 1996 prior to the BHBF. Irrespective of which month the experimental spike flow occurs, we expect that it will remove or damage most of the KAS primary habitat and cause mortality of most KAS up to the stage of the flow. The actual number of KAS lost due to the flood will depend greatly on the amount of winter mortality, which can vary dramatically among years dependent on the severity of winter temperatures (Stevens et al. 1997, IKAMT 1998). Based on best available data, the area of primary habitat will not exceed the amount that was present in late March of 1996 when the 45,000 cfs BHBF occurred and thus the amount of incidental take (17%) identified by the Service (2000) will not be exceeded.

In its December 21, 1994, Final Biological Opinion, the Service evaluated impacts to KAS from the operation of Glen Canyon Dam according to operating and other criteria of the preferred alternative contained in the FEIS. The Service determined implementation of the preferred alternative would not jeopardize the continued existence of the Vaseys Paradise KAS population. This opinion also supported the concept of a beach/habitat building flow of 40,000 to 45,000 cfs, which is part of the preferred alternative. At the time of the 1994 biological opinion, the Service thought that 10% of KAS habitat would be lost in a 45,000 cfs flow and set this amount, as vegetation rather than number of snails, to be the expected incidental take. Information obtained in ensuing investigations showed that the incidental take in a 45,000 cfs release could be as much as 17% of KAS habitat (Service 1996), and, pursuant to that finding, the Service adjusted the incidental take to be 17% (Service 2000). The present assessment examines (1) the probability of exceeding the incidental take level during the experimental flows and (2) the potential for higher than maximum power plant releases (beach-habitat building flows) occurring at a frequency exceeding the once in five years assumed for the analysis of impacts used in the FEIS.

Losses of KAS habitat and KAS at Vaseys Paradise are partially offset by the developing population at Upper Elves Chasm. Long-term success of this population cannot be assured, but it has now persisted and grown for four years. The amount of occupied habitat at upper Elves Chasm, estimated at 25 m^2, is approximately 20% of the occupied habitat expected to be temporarily lost at Vasey's Paradise due to the proposed action.

Mechanical Removal of Non-Native Fish

The mechanical removal component of the Proposed Action would not affect the Kanab ambersnail.

Conclusion

Given the above considerations, it is our assessment that the Proposed Action may affect, and is likely to adversely affect, the Vaseys Paradise population of Kanab ambersnail.

Humpback Chub Species Account

Distribution and Abundance

The humpback chub (HBC) is a cyprinid fish species found only in the Colorado River Basin and described by Miller (1946) from specimens collected in Grand Canyon. Earliest evidence of humpback chub in Grand Canyon comes from non-fossilized remains about 4,000 years old that were collected in Stanton's Cave (Miller 1955) at RM 31.5 approximately 46 miles below Glen Canyon Dam. HBC was apparently widely distributed in canyon reaches of the Colorado River, at least down to 24 miles (39 km) below the present site of Hoover Dam (Miller 1955). Present distribution in the Colorado mainstream in Grand Canyon has been reduced about 25% from the pre-dam period and is largely limited to small aggregations of adult fish. Valdez and Ryel (1995) identified nine distinct aggregations in the mainstream Colorado River downstream from Glen Canyon Dam, including: 30-Mile, Little Colorado River (LCR) inflows, Lava/Chuar to Hance Rapids, Bright Angel Creek mouth, Shinumo Creek mouth, Stephens Aisle, Middle Granite Gorge, Havasu Creek mouth and Pumpkin Spring.

The largest aggregation of HBC is located within the LCR. HBC occupy approximately 8 miles of the 12 miles of perennial flowing water in the tributary above its confluence with the Colorado River. Lack of HBC in the upper 4 miles has been attributed to high concentrations of free carbon dioxide from the springs that provide the perennial flows, however Robinson et al. (1996) provided evidence that physical obstructions, i.e. travertine dams, may be precluding occupation of that reach.

HBC was listed as an endangered species in 1967 (32 FR 4001; March 11, 1967) and critical habitat was prescribed in 1994 (Service 1994). Critical habitat includes the

Colorado River in Coconino County, Arizona, from Nautiloid Canyon (RM34) to Granite Park (RM208) and the lower 8 miles of the LCR. Primary constituent elements include water quantity and quality, habitat for spawning, feeding, and rearing, or corridors between these areas, and, in the biological environment, food supply, predation, and competition. Instream alteration, including flow modification, diversion for irrigation, channelization, and fragmentation by reservoirs, and introduction of non-native fish competitors and predators, with their attendant diseases and parasites, have been suggested as being responsible for declining populations of HBC throughout the Colorado River Basin (Colorado River Fishes Recovery Team 1990, Valdez and Carothers 1998).

Five HBC populations remain in canyon-bound reaches of the upper Colorado River basin: Black Rocks and Westwater Canyon in the Colorado River, Cataract Canyon and Desolation/Gray canyons in the Green River, and the Yampa River population. The Grand Canyon population is the only successfully reproducing HBC population in the lower Colorado River Basin (Kaeding and Zimmerman 1983, Valdez and Ryel 1995).

The first population estimate for HBC in Grand Canyon was made by Kaeding and Zimmerman (1982), who gave a rough estimate of 7,000-8,000 individuals larger than 200 mm in the LCR and a 19 mile (32 km) reach of the mainstream in the vicinity of the confluence. Valdez and Ryel (1995) estimated that 3000 to 3500 adult (>200 mm total length) HBC occupied the mainstream Colorado River in 1991-1993, most of which were concentrated within 4.2 miles of the mouth of the LCR. Douglas and Marsh (1996) estimated the LCR population size in 1992 for HBC greater than 150 mm total length at approximately 4,500 individuals, using a closed population model. Since a portion of the HBC population moves back and forth between the LCR and mainstream, some individuals may have been counted twice. Thus, the total population was less than the sum of these estimates. Coggins and Walters (2001) have identified a decline in LCR humpback chub numbers beginning in 1993. Their most recent estimate, for 2001, indicates the population of HBC >150 mm total length in the LCR has declined to about 2100 individuals.

Population estimates have not been made for the mainstream aggregations since 1993, but their numbers are thought to be sustained largely by influx of individuals leaving the LCR population (Valdez and Ryel 1995). The response of HBC to the year 2000 experimental native fish flows can not be determined using the stock assessment approach, because fish hatched in 2000 are not yet recruited to the population. Potential causes of the decline in HBC include: 1) Colorado and Little Colorado River hydrology, 2) infestation of juvenile HBC by Asian tapeworm, 3) predation by or competition with warm-water native cyprinids and catastomids and non-native cyprinids and ictalurids within the LCR, 4) predation by or competition with cold-water non-native salmonids within the Colorado River and (5) perennially cold hypolimnial releases from Lake Powell through Glen Canyon Dam (Service 1994a, Clarkson and Childs 2000, Robinson and Childs 2001, GCMRC 2002b).

Habitat use by HBC varies across age classes. Individual adult HBC demonstrate high microsite fidelity (Valdez and Ryel 1995), but young HBC may drift for relatively long distances (Tuegel et al. 1995). Young HBC in the mainstream commonly use return current channels and other backwater habitats. Backwaters offer low velocity, relatively warm, protected, food-rich habitats when compared to nearby mainstream habitats (Kennedy 1979, Grabowski and Hiebert 1989, Arizona Game and Fish Department 1996, Hoffnagle 1996). HBC use of these habitats in Grand Canyon has been compromised by fluctuating flows and perennially cold dam releases, which reduce warming and create unstable conditions (Maddux et al. 1987, Hoffnagle 1996). Backwater numbers and area generally have been reduced under ROD operations in Grand Canyon, but this process is interrupted by periods of increase in response to changes in hydrology and sediment inputs (McGuinn-Robbins 1995, Stevens and Hoffnagle 1999).

Gorman (1994) found shifts in habitat use among HBC in the LCR of different size and age. As fish grew, they tended to move from nearshore to offshore waters having greater depth, current velocity, average substrate coarseness, and amount of vertical structure. Childs et al. (1998) revealed habitat partitioning among small (<30 mm total length) native fish, including HBC, in the LCR. Subadult HBC in the Colorado River mainstream often use irregular shorelines as habitat, and adult HBC often occur in or near eddies (Valdez and Ryel 1995, Converse 1996). 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 and Ryel 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 and Ryel 1995, Valdez and Carothers 1998). A key issue is lack of recruitment to the adult population from low survivorship of young fish (Valdez and Ryel 1995). Perennially cold mainstream water temperatures are strongly implicated as being responsible for unsuccessful mainstream reproduction. The minimum water temperature for successful reproduction is 16°C(Hamman 1982, Marsh 1985), well above the commonly observed 10°-12°C summer mainstream temperatures. Mortality of larval and postlarval humpback chub emerging from the warm waters of the LCR has been ascribed to thermal shock and enhanced susceptibility to predation from the more protracted debilitating effects of cold water on swimming ability and growth (Lupher and Clarkson 1994, Clarkson and Childs 2001, Robinson and Childs 2001, Ward et al. 2002).

Sexual maturity of female chubs begins at approximately 250-280 mm total length, which is at about three years of age for HBC in the LCR (Kaeding and Zimmerman 1983). Gonadal development is rapid in the LCR and Colorado River between December and February to April, at which time indices reached highest levels (Kaeding and Zimmerman (1983). Most successful spawning by HBC in Grand Canyon occurs in the lower 8 miles of the LCR from March through May. Adult fish initially stage for spawning runs in large eddies near the confluence of the LCR in February and March. They make spawning runs that average 17 days into the tributary from March through May. Spawning has not been observed, but ripe males have been seen aggregating in areas of complex habitat structure (boulders, travertine masses, and other sources of angular variation), and it is thought that ripe females move to these aggregations to spawn (Gorman and Stone 1999). After spawning, some adult chub return to specific microsites in the mainstream and others remain in the LCR.

As LCR flows decrease, warm, and clear, reproduction increases and larval fish appear (Valdez and Ryel 1995). Young HBC remain in the Little Colorado River, or drift and swim into the mainstream (Robinson et al. 1998) where lack of recruitment is attributed to effects of cold mainstream temperatures and non-native fish predators and competitors (Lupher and Clarkson 1994, Valdez and Ryel 1995, Marsh and Douglas 1997, Clarkson and Childs 2000, Robinson and Childs 2001). During the summer the young HBC that survive in the mainstream occupy low-velocity, vegetated shoreline habitats, including backwaters; 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 of the following year.

Limited spawning and hatching of HBC occurs in mainstream aggregations. Valdez and Ryel (1995) documented limited spawning success near the warm spring at 30-Mile in upper Marble Canyon. Young-of-year HBC in the size range of 10-30 mm have been collected sporadically at considerable distances below the LCR, usually beginning in June (Kubly 1990, Arizona Game and Fish Department 1996, Brouder et al. 1997). The combination of larval to postlarval sizes and the low probability these fish surviving the extreme rapids of the inner gorge in Grand Canyon suggests strongly that their source was below the LCR. Some limited reproduction may occur in other smaller tributaries. Young HBC have been collected in or near Bright Angel Creek, Shinumo Creek, Kanab Creek, and Havasu Creek (Maddux et al. 1987, Kubly 1990, Arizona Game and Fish Department 1996, Brouder et al. 1997).

Dietary analyses reveal HBC to be opportunistic feeders, largely feeding on algae and aquatic and terrestrial invertebrates (Kaeding and Zimmerman 1982, Kubly 1990, Valdez and Ryel 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 comprises 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 clear water conditions often prevail.

Kaeding and Zimmerman (1983) identified 13 species of bacteria, six protozoans, and a fungus from HBC in Grand Canyon. In 1990 the Asian tapeworm (*Bothriocephalus acheilognathi*), an introduced parasite, was first identified from HBC in the LCR (Clarkson et al. 1997). This tapeworm is particularly worrisome, because it infects HBC at a high rate and has been reported to be pathogenic and potentially fatal in a variety of other fish (Hoffman and Schubert 1984, Hoffnagle et al. 2000).

Impacts of the Proposed Action Alternative: Experimental Test Flows

8,000 cfs Steady Flows.—Steady 8,000 cfs releases can occur for two-week periods during the months of September-December, and for up to 10 days following experimental spike releases during the months of January-March. In the first case, they are interspersed with equivalent periods of fluctuating 6,500-9,000 cfs releases. In the latter, they are preceded by experimental spike releases of 42,000-45,000 cfs and followed by non-native fish control releases fluctuating between 5,000 and 20,000 cfs.

Larval and young-of-year HBC that drift or swim out of the tributary into the mainstream and make it to nearshore rearing habitats during the months of June-October would experience more days of stable flow conditions under these releases than under No Action ROD fluctuations. By remaining in these habitats young fish would enjoy warmer water temperatures and a greater abundance and diversity of food resources. Preliminary results from the year 2000 experimental flow research (Fritzinger et al. 2000) indicate that feared large increases of non-native fish in these rearing habitats during steady 8,000 cfs releases were not realized, and that mainstream reproduction was suspected by three native fish—flannelmouth sucker, bluehead sucker, and HBC (Trammell et al. 2001). Larger HBC in offshore eddies might experience some diminishment in organic matter drift during this period, but it is not established how much this species feeds on drift in the current as opposed to benthic matter off of bottom substrates. This flow would not be less than the minimum allowable under ROD operations, and therefore wetted area available for primary and secondary productivity would not be diminished.

The interruption of steady releases at two-week intervals by fluctuating releases differs from the year 2000 experience, and limits the extent to which results from this experiment can be extrapolated. Expectations for changes in rearing habitats during the 6,500-9,000 cfs autumn fluctuating releases are provided below.

Steady 8,000 cfs releases during the period of January-March would have little effect on young HBC, who by this time in their lives have moved into deeper water habitats of eddies adjoining their earlier rearing habitats. Adult HBC in pre-spawning condition are congregated near the mouth of the LCR for at least the early part of this period, but they assumedly are keying more on flows emerging from the tributary than on those in the mainstream. Some diminishment of drifting organic matter could occur relative to No Action ROD fluctuations, however this effect may well be reduced by reduction in the standing crop of particulate matter during the preceding experimental spike release.

6,500-9,000 cfs Fluctuating Flows.—Under the Proposed Action Alternative, these releases can occur for two-week periods from September-December in the autumn sediment input scenario. In this scenario, they would alternate with two week periods of steady 8,000 cfs releases. Anticipated stage changes downstream of Glen Canyon Dam from these fluctuations are from 0.7-1.8 feet depending on location below Glen Canyon Dam (table 3.1). Exact changes in rearing habitats brought about by these fluctuations can not be determined in advance. Past reports of environmental conditions in backwaters reveal that they have warmer water temperatures than the mainstream during summer and autumn months, and that warming of backwaters is diminished by fluctuating flows (Arizona Game and Fish Department 1996, Hoffnagle 1996, Brouder et al. 1997). Backwater habitats also contain much higher densities of planktonic and benthic food organisms than other nearshore habitats accessible to young fish (Kubly 1990, Brouder et al. 1997). Fluctuating flows dewater portions of backwaters at low releases, and in the extreme can temporarily dry them or isolate them from the mainstream. When releases increase, warmer backwaters are infused with cold mainstream water released from the depths of Lake Powell. These cyclical changes disrupt the stability of the rearing environment for young native warm water fish (Kennedy 1979, Ward 1976). The amount of daily change in backwater environments that occurs at fluctuations of 6,500-9,000 cfs will vary dependent on the geometry of the return channel, with those having lesser slopes being more affected.

5,000-20,000 cfs Fluctuating Non-Native Fish Suppression Flows.—Non-native fish suppression flows would occur from January-March under the Proposed Action. Stage changes at stream gages for this release fluctuation are approximately 3.6 ft at Lees Ferry, 5.7 ft at the LCR, and 8.2 ft at Grand Canyon. These fluctuations would exceed those under the No Action Alternative. Major physical changes in environments from these flows are anticipated along shorelines and in shoreward habitats from regular dewatering. Effects on humpback chub will be reduced, however, because most individuals in the mainstream, even if the progeny of the preceding year, will already have moved to deeper habitats further off shore by late autumn. Some local displacement may occur in shallower eddies, potentially resulting in increased energy expenditure for the few fish that would be affected. Survivorship of young-of-year through the winter apparently is very low, however, irrespective of hydrology (Valdez and Ryel 1995). Low survivorship may well be as much a consequence of cold water temperature as of hydrology, although it is clear that these physical parameters are tightly linked, particularly in rearing habitats. Long-term benefits are expected from reduced trout predation as a consequence of these flows.

31,000-33,000 cfs Habitat Maintenance Flows.—Under the Proposed Action, these flows can occur from July-December. For such a flow to occur, a minimum sediment trigger of 500,000 metric tons must be reached, but this trigger could occur more than once in a season. Small native fish, including humpback chub, are present in nearshore rearing habitats from approximately May-October, then as nearshore water temperatures cool and the fish reach juvenile to subadult sizes they move to deeper eddies (Maddux et al. 1987, Converse 1996). Prior to this transition in habitat use, small humpback chub and other native fish could be displaced from rearing habitats by flows of this magnitude. Similar effects were observed after the early September habitat maintenance flow in year 2000 (M. Trammel, SWCA, personal communication). Since few young HBC appear to survive in the mainstream under normal ROD operations, i.e., the No Action Alternative, little additional mortality is expected from these flows.

Anticipated effects on mainstream critical habitat from these flows are that during the flows, rearing habitats formed in soft sediments will be disturbed. Some backwaters will be deepened by erosional processes and others will be partially filled by depositional processes. A desired effect of these flows is the accumulation of fine sediments in eddies and on beaches. It is likely that organic matter, both from the river and of terrestrial origin, will be buried in areas where sediment deposition is predominant. This organic matter may provide important nutrients in local environments through decomposition and nutrient mobilization similar to that seen following the 1996 BHBF (Parnell et al. 1999). Sediment-laden water at high flows also will cover and scour algae, macrophytes, and invertebrates that provide food for native and non-native fish. This effect will be offset at least partially by entrainment of both river-produced and terrestrially-produced organic matter food materials. The duration of the event will be short, and long-term effects on these habitats are expected to be positive.

42,000-45,000 cfs High Flows.—These flows could occur during January-March, a time of year by which surviving young-of-year HBC have moved to deeper eddies. Subadults and adults are expected to be little affected by these larger flows, although they do occur at a time of the year prior to the rise in the predam hydrograph. Little is known about the extent to which HBC rely on changes in flow as a reproductive cue. Valdez and Ryel (1995) held that neither water quantity or quality serve as cues for gonadal development or staging behavior in HBC; rather they hypothesized that climatic factors, such as photoperiod, were important. Humpback chub typically begin to spawn on the receding hydrograph as water temperatures start to rise (Tyus and Karp 1989, Kaeding and Zimmerman 1983, Valdez and Ryel 1995, Kaeding et al. 1990), but the LCR population also spawns in years with little appreciable runoff.

Effects on critical habitat from these flows should be similar to the BHBF of 1996. One difference between the two events will be the amount of fine sediment in the system to be mobilized, but we perceive no significant negative impact on HBC from this change. There will be mobilization and redistribution of fine sediment that forms native fish rearing habitat, but this effect, which is to rejuvenate these habitats, should have a beneficial effect. The anticipated outcome of the high winter flows following a period of sediment conservation is that they will succeed in attaining this desired outcome to a greater extent than the spring 1996 flows.

The Reasonable and Prudent alternative of the 1994 biological opinion (Service 1994) includes habitat/beach building flows; however, the Service determined some HBC would be taken during such an event. The discussion of incidental take in the biological opinion 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 mainstream. This hypothesis will be examined through the test flow.

In summary, we anticipate incidental take of HBC and short-term effects on critical habitat as a result of these test flows. The long-term effects on HBC from reduced numbers of deleterious non-native fish and rejuvenated rearing habitats are expected to be positive.

Mechanical Removal of Non-Native Fish

The flow treatments discussed above, as related to non-native fish reduction, center around the hypothesis of improving future HBC recruitment by reducing the number of adult rainbow trout and brown trout residing in the Colorado River downstream of Glen Canyon Dam. Conceptually, this is to be accomplished primarily by reducing rainbow and brown trout recruitment by increasing the early life mortality rate of these fishes with highly fluctuating flows during their winter and spring spawning and rearing seasons. The other experimental treatment calls for the reduction of adult rainbow and brown trout abundance in the Colorado River mainstream near the confluence of the Little Colorado River (LCR) via electrofishing and mechanical removal. The mechanical removal of salmonids and its potential effects on the endangered HBC are described in this section.

GCMRC proposes to use electrofishing as a collection method for this effort. It is important to state here that electrofishing itself is not intended to be lethal to fish but only to disable a fish's normal swimming ability to allow its capture and use for other purposes. GCMRC and its predecessor organizations have developed electrofishing techniques designed specifically to optimize capture and minimize incidental mortality as a result of sampling. They also have a specific set of fish handling protocols. All GCMRC staff and cooperating investigators adhere to these procedures

Two types of electrofishing boats will be utilized. The first is the standard electrofishing boat developed and used during the GCES Phase II research and monitoring effort. The boat is an Achilles SU-16, outfitted with the Coffelt CPS electrofishing system. The second type of electrofishing boat is a rigid hull aluminum sport boat constructed by Osprey. This boat has been recently permitted for use in Grand Canyon National Park and has proven to be a safe and more efficient working platform than the Achilles SU-16. Like the Achilles SU-16, the Osprey boat will be outfitted with the Coffelt CPS electrofishing system.

The Coffelt CPS system generates a pulse train of three 240 Hz, 1.6-ms pulses every 1/15 second and is quite effective at reducing electrofishing induced injuries related to the use of this equipment (Sharber and Carothers 1988, Valdez et al. 1993, Snyder 1992, Cowdell and Valdez 1994, Sharber and Black 1999). Cowdell and Valdez (1994) found hemorrhaging along the spine of two out of 40 adult roundtail chub using the CPS system, but no vertebral damage was observed in lateral xrays. Ruppert and Muth (1997) exposed juvenile humpback chub and bonytail to electrofishing and found no significant difference in results for the two species. They found no mortalities, external injuries, or vertebral injuries in any of the fish. Spinal hemorrhages were found in 13% (46 of 360) of shocked bonytails. Only one treatment could be applied to the few HBC available for that test. In that test, using CPS, 6 of 30 treatment HBC and 1 of 30 control HBC had spinal hemorrhages. Hemorrhages were described as class 2 with wounds on the spine being the width of two or less vertebrae and ranged from one to eight per fish. In all experiments, fish subjected to enough electricity to induce strong muscle contractions produced more injuries than those subjected to an amount causing paralysis. Early life stages of HBC have not been subjected to electrofishing experiments. Embryos and larvae of razorback sucker have been studied by Muth and Ruppert (1996).

They found reduced survival in embryos and diminished growth in larvae subjected to electrical current.

The effort will be conducted in habitat used by humpback chub and an unknown number of humpback chub will be collected. Precise numbers of chub captured cannot be determined *a priori*. However, GCMRC has considerable data from previous electrofishing efforts in the LCR reach that allow estimation of the likely range of capture rates. The following table A-2 presents projected humpback chub electrofishing capture data from the LCR reach of the Colorado River based on data from an approximately 10-year period for river miles 61-65. This area does not correspond precisely to the intended 9.5 mile mechanical removal area for non-native salmonids, which is somewhat larger. We believe HBC are less abundant in the additional reach included for mechanical removal procedures and HBC numbers have declined in the last 10 years. Therefore, the range of capture rates presented here is biased high and may be up to two times the actual catch rates for HBC.

Table A-2.—Projected HBC captures for each trip from the LCR reach of the Colorado River.[1]

		CPUE/10 hrs[2]		CATCH (number)	
	Effort (trip hrs)	HBC <200mm	HBC ≥200mm	HBC <200mm	HBC ≥200mm
Mean	320	11.94	0.45	382	15
Median	320	5.16	0.27	165	9
Minimum	320	0.00	0.00	0	0
Maximum	320	89.15	5.61	2853	180

[1] Projections based on electrofishing data from an approximately 10 year period for a 5-mile reach of the Colorado River around the confluence with the LCR (River Miles 61-65).

[2]CPUE refers to the number of individuals collected in a unit of time, herein 10 hours.

Captures of adult (HBC ≥200mm) are estimated to range from a mean derived value of 15 to a high of 180 fish per trip. Juvenile and sub-adult capture rates would be higher, ranging from perhaps 165 to 2853 fish. It is not possible to estimate precisely what rate of incidental mortality might be incurred in these fish. We have little evidence of direct mortality resulting from electrofishing sampling of HBC. It is GCMRC's supposition that median values from historical data represent the best projected numbers for likely capture rate. The following table A-3 provides a summary of the planned removal trips.

All HBC captured will be handled according to procedures specified in the GCMRC fish handling protocol and marked as appropriate for research purposes, which will yield additional information regarding this population. GCMRC estimates that one-quarter of 1 percent of juvenile fish may suffer mortality from electrofishing and that less than 1 percent of adults will suffer mortality. Using median capture values this would represent a take of less than one-tenth adult fish per trip and less than one-half a juvenile fish per trip. Reasonable limits on lethal incidental take for all trips combined during 2003 and 2004 might be 10 adult and 50 juvenile fish.

Table A-3.—Summary of sampling schedule for rainbow and brown trout mechanical removal trips, 2003-2004.

Trip Type	Trip Date	FY-Year	Trip Length	Electrofishing Passes/Trip
Electrofishing Depletion	15 – 30 Jan	2003	15 - day	5
Electrofishing Depletion	15 – 30 Feb	2003	15 - day	5
Electrofishing Depletion	15 – 30 March	2003	15 - day	5
Electrofishing Depletion	1- 15 Jul	2003	15 - day	5
Electrofishing Depletion	1- 15 Aug	2003	15 - day	5
Electrofishing Depletion	1- 15 Sept	2003	15 - day	5
Electrofishing Depletion	3 trips Jan-Mar	2004	15 - day	5
Electrofishing Depletion	3 trips Jul-Sep	2004	15 - day	5

Based on literature findings, effects from electrofishing on individuals will vary by degree of exposure and size. The principal intended consequence of the proposed activity is to benefit HBC. Nevertheless, some incidental take of HBC may occur as a consequence the mechanical removal of non-native fish by electrofishing.

Other Related Actions

The GCMRC, in its role as provider of technical and scientific information to the Glen Canyon Adaptive Management Program, participates in and/or coordinates monitoring activities intended to provide information regarding the status and trends of fish species in the Colorado River and its tributaries in Grand Canyon. Cooperators in these activities are the Arizona Game & Fish Department, the U. S. Fish & Wildlife Service, and the consulting firm SWCA, Inc. Each of these parties holds research permits pertaining to its activities. Although personnel from all of the organizations participate in sampling trips in a reciprocal manner, each organization has a lead responsibility for certain monitoring activities under contract with GCMRC. The planned sampling trips are detailed in table A-4. Some adjustment of these sampling dates may be necessary to avoid overlapping with the mechanical removal analyzed in this biological assessment. Adjustments will be made as needed to ensure that sampling activities occur at least 2-4 weeks earlier or later than electrofishing mechanical removal efforts, since the additional research being conducted through GCMRC also will involved capture and handling of HBC. The sampling schedule for this work is based on six Colorado River mainstem trips and five trips in the Little Colorado River during 2003, and four Colorado River mainstream trips and six trips in the Little Colorado River during 2004.

Table A-4. Related research and monitoring field trips for sampling of endangered humpback and other fishes in Grand Canyon during 2003-2004.

2003-Trip No.	**Location**	**Date**	**Description**	**Agency**
2003-1	Mainstream CR	January	juvenile humpback chub mortality monitoring	USFWS
2003-2	Mainstream CR	February	trout and carp monitoring	AGFD
2003-3	Mainstream CR	April	trout and carp monitoring	AGFD
2003-4	LCR	April	humpback chub mark-recapture	USFWS
2003-5	LCR	April-May	humpback chub lower 1200 m monitoring	AGFD
2003-6	LCR	May	humpback chub mark-recapture	USFWS
2003-7	LCR	September.	humpback chub mark-recapture	USFWS
2003-8	LCR	October	humpback chub mark-recapture	USFWS
2003-9	Mainstream CR	July	native fish monitoring	SWCA
2003-10	Mainstream CR	September	native fish monitoring	SWCA
2003-11	Mainstream CR	November	juvenile humpback chub mortality monitoring	USFWS

2004/Trip No.	**Location**	**Date**	**Description**	**Agency**
2004-1	Mainstream CR	14-Feb – 4 Mar	trout and carp monitoring	AGFD
2004-2	Mainstream CR	4 – 21 Apr	trout and carp monitoring	AGFD
2004-3	LCR	8 - 19 April	humpback chub mark-recapture	USFWS
2004-4	LCR	18 April – 24 May	humpback chub lower 1200 m monitoring	AGFD
2004-5	LCR	13 – 24 May	humpback chub mark-recapture	USFWS
2004-6	LCR	24 Jun – 3 Jul	catfish and carp gear evaluation	AGFD
2004-7	Mainstream CR	17 July –2 Aug	native fish monitoring	SWCA
2004-8	Mainstream CR	11-27 Sep	native fish monitoring	SWCA
2004-9	LCR	16 - 27 Sept.	humpback chub mark-recapture	USFWS
2004-10	LCR	21 Oct. - 1 Nov	humpback chub mark-recapture	USFWS

Conclusion

It is our conclusion that the Proposed Action Alternative may affect, and is likely to adversely affect, humpback chub and its critical habitat. Both humpback chub and the biological component of its critical habitat are expected to benefit in the future from removal of these non-native fish.

Razorback Sucker Species Account

Distribution and Abundance

Razorback sucker (RBS; Catostomidae) is part of the highly endemic native fish fauna of the Colorado River Basin. RBS formerly occurred throughout the Colorado River, from Wyoming to northwestern Mexico, (Minckley et al. 1986), but its distribution has declined dramatically since 1930 with the increasing fragmentation and regulation of the Colorado River (Dill 1944, Minckley 1991). The decline of RBS has been attributed to water temperature changes, altered spawning habitat, fragmentation of river systems by reservoirs, and introduction of non-native fish species, which have cumulatively resulted in wide-scale recruitment failure (Bestgen 1990, Minckley 1991). The species is listed as endangered with critical habitat under the Endangered Species Act (Service 1991, 1994b). Critical habitat includes the Colorado River and its 100-year flood plain from the confluence with the Paria River to Hoover Dam, including the full pool elevation of Lake Mead. Primary constituent elements include water quantity and quality, habitat for spawning, feeding, and rearing, or corridors between these areas, and in the biological environment, food supply, predation, and competition.

The largest RBS population in the Lower Colorado River Basin exists in Lake Mohave. That population was estimated to be approximately 60,000 fish in 1989 (Marsh and Minckley 1989), but it has declined considerably since that time (Marsh 1994). A second RBS population of approximately 500 individuals occurs in Lake Mead. In the Upper Colorado River Basin, RBS occurs regularly in the upper Green and lower Yampa rivers (Tyus 1987). RBS have been collected at rare intervals in the Colorado River near Grand Junction, Colorado, and in the major tributary arms of Lake Powell. Most wild-caught RBS are old individuals (RBS live from 20 to 50 years), 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 Colorado River Basin reservoirs are encouraging, but the predam remnant mainstream populations continue to decline.

Based on available literature, RBS is very rare in Grand Canyon and genes from the taxon are present largely in hybrids with flannelmouth sucker (Douglas and Marsh 1998). Some fish biologists speculate that this species was never more than a transient member of the native fish fauna there (Minckley 1991, Douglas and Marsh 1998). Recent observations are those of Carothers and Minckley (1981) who reported four RBS from the Paria River in 1978-1979. Maddux et al. (1987) reported one blind female RBS at

Upper Bass Camp (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). Douglas and Marsh (1998) confirmed the presence of such hybrids and estimated their numbers between 8 and 136.

The population of RBS in Lake Mead has been studied since 1996 (Holden et al. 2000). During the first four years, 115 individuals were collected, not counting larvae. In August 1999, an adult RBS was found in upper Lake Mead at the western side of the mouth of Grand Wash. This discovery was followed in 2000 by collection of larval RBS in the far eastern part of Lake Mead. Holden et al. (2000) concluded that "spawning occurred in the lake, either near the Colorado River inflow area or in the actual Colorado River before it enters the lake." Douglas and Douglas (2000) reported a larval RBS identified by the Colorado State University Larval Fish Laboratory from collections made at the mouth of Havasu Creek in Grand Canyon. They admitted the possibility that this could have been of a hybrid between RBS and flannelmouth sucker, but noted that all known hybrids occur considerably higher in the system, in Marble Canyon and the LCR.

Life Requisites

RBS are generally associated with calm river reaches, and since damming of the Colorado River, with reservoirs (Tyus 1987). Riverine spawning typically occurs in shallow water over gravelly substrates, often in areas of inflowing streams where gravel sorting has occurred (Minckley 1983, Mueller 1989). 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 are in a range from 10-15°C (50 to 59°F) (Bozek et al. 1984). Marsh (1985) demonstrated in the laboratory that the highest successful hatching percentage for RBS occurs at 20°C, and that the hatch declines considerably at 15°C with complete mortality at 10°C. In riverine situations in the Upper Basin, RBS begins spawning on the rising limb of the spring (April-May) hydrograph and continues for an extended period through the spring runoff. Although spawning occurs throughout the day, it is most intense at dusk.

Larval RBS drift downstream from the spawning habitat, using quiet shorelines, and concentrate in warm, low-velocity areas (e.g. flooded bottoms). These areas also support post-larval RBS. Mainchannel and mid-stream river habitats floored by fine-grained alluvium are important to subsequent RBS life stages (Minckley 1983, 1991, Tyus and Karp 1989). 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 22-25°C (72-77°F) (Bulkley and Pimentel 1983); however, RBS occurs 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 32°C(90°F) in summer (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. Papoulias and Minckley (1990) determined densities of plankton leading to starvation in larval RBS in the laboratory. Adult RBS in lentic habitats engage in both planktivorous and benthic feeding on a variety of zooplankton, phytoplankton, filamentous algae, and detritus (Marsh 1987), while adult RBS in rivers feed primarily on benthic algae and invertebrates (Banks 1964).

Growth among individuals in the same cohort is highly variable (Minckley et al. 1991), and this variation may represent divergent strategies in this long-lived fish for dealing with the highly unpredictable environment of desert rivers in southwestern U.S. Growth is rapid for approximately the first six years, but then it slows dramatically (McCarthy and Minckley 1987).

A variety of bacteria, protozoans, cestodes, trematodes, nematodes, and copepods is known to infect RBS, but there seems to be little overall impact on health or mortality (Minckley et al. 1991). Many captured individuals are aged and afflicted with tumors, blindness or other maladies (Minckley 1983, Bestgen 1990).

Predation on RBS by non-native fish, primarily ictalurids, is well documented (Marsh and Brooks 1989). Larval RBS are highly susceptible to predation by non-native fish and invertebrates (Marsh and Langhorst 1988, Horn et al. 1994), but there is evidence that their susceptibility can be lessened by turbidity from suspended sediments (Johnson and Hines 1999). Recent efforts at conservation include rearing fish in grow out ponds to a size where predation is reduced and then placing them in reservoirs.

Impacts of the Proposed Action Alternative: Experimental Flows

8,000 cfs Steady Flows.—Steady releases that follow spawning by any remaining RBS in Grand Canyon would provide one of the required elements for successful RBS rearing, but even with steady releases, larval fish in the mainstream would face challenges of limited rearing habitat. Flooded bottomlands that are characteristic rearing habitat for this fish are largely lacking in Grand Canyon. Also, no tributaries in the affected reach appear to have suitable conditions for successful spawning and rearing by RBS.

6,500-9,000 cfs Fluctuating Flows.—Under the Proposed Action Alternative, these releases could occur between September and January, alternating in two-week long releases with steady 8,000 cfs releases. Any backwaters inhabited by larval RBS would be subject to daily inflow of cold mainstream water under the fluctuating releases. Larval fish displaced from backwaters would likely enter the drift and be transported downstream through major rapids. Individuals that survived the physical challenges of transport also would be subjected to predation by non-native fishes.

5,000-20,000 cfs Fluctuating Non-Native Fish Suppression Flows.—Under the Proposed Action Alternative, these releases are implemented during January–March and are independent of sediment inputs. They follow either ROD operations or the 8,000 cfs releases that have followed a 42,000-45,000 cfs peak release. There would likely be little effect on RBS as the releases would precede spawning and any subsequent occupation of backwater habitats by juvenile RBS.

31,000 cfs-33,000 cfs Habitat Maintenance Flows.—Under the Proposed Action Alternative, these flows could occur anytime between July and January and would last for two days. Any backwaters inhabited by larval RBS in late summer or early autumn would likely be converted to eddies as stage increased. They and other nearshore habitats would be subjected to large increases in current velocity. Larval fish displaced from backwaters would likely enter the drift and be transported downstream through major rapids. Individuals that survived the physical challenges of transport also would be subjected to predation by non-native fishes.

42,000-45,000 cfs High Flows.—Almost all RBS remaining in Grand Canyon are likely mature or senile fish, which survived comparable or higher mainstream flows in 1965, 1980, 1983-1986, and 1996. Based on the decline in the Lake Mohave population, very few RBS hatched prior to emplacement of Glen Canyon Dam are likely to be alive in 2002. If older fish do exist in Grand Canyon, they are likely capable of finding suitable refugia from high flows and unlikely to use nearshore areas affected by fluctuating flows.

If there are reproductively active RBS in Grand Canyon, an experimental high flow in January-March might serve as an environmental cue for spawning. This high flow would also be experienced by RBS in upper Lake Mead. However, historically, RBS in Grand Canyon most likely received their cue to begin spawning on the rising limb of the spring (April-May) hydrograph. Also, the short duration of the flow (60 hours) would contrast greatly with the long-term winter snowmelt peak of the pre-dam era, and it may not evoke any change in RBS behavior or physiology.

RBS feed on a wide variety of planktonic and benthic food sources. Dam releases under the proposed action may result in positive effects to the RBS food base. Results of the 1996 beach/habitat-building flow showed two major responses in eddy recirculation zones: (1) deposition and infilling in many small recirculation zones, and (2) extensive scouring of low elevation deposits in large recirculation zones (Hazel et al. 1999). Deposited fine sediments buried organic matter produced in the river and subsequent decomposition of this organic matter produced enriched nutrients that stimulated primary production (Parnell et al. 1999). Similar effects from the proposed action releases could increase the size and depth of larger backwater rearing habitats and increase their productivity in months following the high release.

Mechanical Removal of Non-Native Fish

The potential effect of mechanical removal of non-native fish on RBS is largely dependent on the probability that individuals will be in the zone of electrofishing effect. Minckley (1991) reported five RBS collected from the mouth of the LCR in 1989-1990. We know of no meristic or genetic analyses conducted on these specimens to ascertain whether they were pure RBS or RBS-flannelmouth sucker hybrids. Valdez and Ryel (1995) collected 2,197 adult flannelmouth suckers, five adult hybrids, and no RBS from the mainstream during 1990-1993. Douglas and Marsh (1998) reported 41 putative RBS/flannelmouth sucker hybrids from 2619 unique individuals collected in the LCR from 1991-1995. Twelve of 41 individuals were examined genetically by restricted endonuclease analysis of mtDNA and nine of these were assayed electrophoretically. Of the nine, eight were of hybrid origin and the remaining individual was a flannelmouth sucker. In their discussion, Douglas and Marsh (1998) discussed difficulties of field personnel in consistently identifying these individuals as flannelmouth, razorback, or hybrids.

Adult RBS use a wide variety of habitats, including eddy complexes and runs that would be sampled during the mechanical removal effort. If present, individuals could be subjected to electrical shock at a rate similar to that experienced by HBC. Unlike HBC, however, RBS do not exhibit high site fidelity and thus the potential for multiple exposures to electricity should be reduced in RBS. Based on published reports cited above as evidence of the rarity of RBS in Grand Canyon, it appears very unlikely that any pure RBS will be in the vicinity of the LCR during the period of mechanical removal in 2003-2004. In addition, reduction of the salmonid population in the reach of the Colorado River above and below the confluence with the LCR could reduce predation on RBS or hybrids of RBS that occur in Marble and Grand canyons.

Conclusion

Based on the information above, we conclude that the Proposed Action Alternative may affect, but is not likely to adversely affect, RBS or its critical habitat.

Southwestern Willow Flycatcher Species Account

Distribution and Abundance

The southwestern willow flycatcher (Tyrannidae: Empidonax traillii extimus) was added to the federal endangered species list on February 27, 1995 (Service 1995). A draft recovery plan was published in April of 2001 but has yet to be finalized. In 1997, critical habitat was designated along the Colorado River from River Mile 39 to River Mile 71.5 below Glen Canyon Dam (Service 1997). The boundaries of critical habitat include the main river channel and associated side channels, backwaters, and pools and marshes throughout the March to September breeding season, as well as areas within 100 meters of the edges of the surface water.

The southwestern willow flycatcher (SWWF) 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, preferring habitat near open water, marshes, or backwaters (Gorski 1969, Sogge 1995). The historic breeding range includes Arizona, New Mexico, western Texas, southern California, and southern portions of Nevada, Utah, and perhaps southwestern Colorado (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 (Aldrich 1953, King 1955, Phillips 1948, Sogge 1995).

Although never common, SWWF 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. 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 tamarisk or saltcedar (Tamarix ramosissima). Habitat fragmentation and modification has benefited some southwestern avian species, especially cowbirds (Molothrus sp.), which parasitize SWWF nests, thus contributing to the precipitous population declines of SWWF (Brown 1994, Johnson and Sogge 1995, Sogge et al. 1995a). SWWF habitat loss in Central and South America also has undoubtedly contributed to recent SWWF population declines.

The SWWF has been extirpated from much of its former range (Hunter et al. 1987) and has experienced a sharp reduction in abundance since 1950. The SWWF is more rare than most other currently listed avian species (Unitt 1987). An estimated 915 territories exist in the United States (Service 2001). It has been given endangered species status by the Game and Fish Departments in Arizona, New Mexico, and California.

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-species 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.

Distribution of the Grand Canyon population fluctuates between Colorado River Miles 47 and 54, and at River Mile 71 (Sogge et al.1995a, Tibbitts and Johnson 1999, Tibbitts and Johnson 2000, Unitt 1987). Nesting SWWF were common in Glen Canyon in the 1950s (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 (Brown 1991a). 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. In 1999 and again in 2000 and 2001, a single nesting pair was detected and monitored (Tibbitts and Johnson 1999; Johnson and Abeita 2000). This site of this nesting has been occupied annually since 1993 (table A-5). The 1999 and 2000 nest was located within several meters of the locations of the nests located in this patch in 1993-1998.

The year 2001 marked the fourth consecutive year in which surveys located a single breeding pair and no unpaired adult willow flycatchers in the Grand Canyon. These last four years represent the lowest population levels since surveys began in 1982. The continued presence of the SWWF in the Grand Canyon appears

to be tenuous (Tibbitts and Johnson 1999, Johnson and Albeita 2000). The number of resident adults available to breed has steadily decreased since a high point of 8 and 9 (in 1993 and 1994, respectively) to a single pair in 1998 through 2001.

SWWF return to wintering grounds in August and September (Brown 1991b). Willow flycatchers have strong winter site fidelity (Koronkiewicz and Sogge 2000). Recent survey and ecology work (Kornonkiewicz and Whitfield 1999, Lynn and Whitfield 2000, Koronkiewicz and Sogge 2000) suggests that wintering flycatchers are not habitat generalists and that suitable and/or high quality wintering habitat is very rare on a landscape level.

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 (Tibbitts et al. 1994). A clutch of three or four eggs is laid from late May through July; in Grand Canyon two or three eggs (usually three) are the norm (Sogge 1995). Breeding generally extends into mid-July but may continue into August.

After a 12-14 day incubation, nestlings spend 12 or 13 days in the nest before fledging (Brown 1988, Tibbitts et al.1994). 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 provide new foraging habitat for brown-headed cowbirds (*Molothrus ater*) and populations of brown-headed cowbirds continue to expand (Hanka 1985, Harris 1991). Brood parasitism remains one of the greatest threats to SWWF and probably many other neotropical migrants (Bohning-Gaese et al.1993, Sogge et al.1995a). Over half the nests in Brown's study (1988) contained brown-headed cowbird eggs. Cowbirds may remove flycatcher eggs, but their eggs also hatch earlier and the larger cowbird nestlings are more competitive in the nest. Brown-headed cowbirds occur extensively around mule corrals on the rim of the canyon and travel down to the Colorado River.

The SWWF breeds and forages in dense, multistoried riparian vegetation near surface water or moist soil along low gradient streams (Whitmore 1977, Sferra et al. 1995, Sogge 1995). Nesting in the Grand Canyon typically occurs in non-native tamarisk approximately 4-7 meters tall (13-23 feet), with a dense volume of foliage 0-4 meters from the ground (Tibbitts et al. 1994). While tamarisk is ubiquitous along the river corridor, the few sites occupied by SWWF are somewhat distinct. In these locations, the tamarisk thickets tend to extend relatively far back from the riverbank, in the range of approximately 30 to 50 meters, and are comprised of dense stands of large, old tamarisk. This contrasts with most of the river corridor, where tamarisk thickets exists as relatively narrow strips close by the water's edge. From above, occupied thickets tend to be broad oval or crescent-shaped areas and have a much greater ratio of interior volume to edge when compared to the thin, linear strip of tamarisk that are common throughout the corridor. Occupied sites also tend to have relatively quiet water, and/or eddies adjacent to them, and notable growths of emergent aquatic vegetation (*Equisetum sp*., *Scirpus* sp.) at the edge of the habitat patch (Tibbitts and Johnson 1999, Johnson and Albeita 2000).

Table A-5.—History of occupied willow flycatcher sites from 1992-2000[1] in Grand Canyon National Park, Arizona.

Site	1992	1993	1994	1995	1996	1997	1998	1999	2000
RM 46.5 R	Vacant	2 single. Banded.	Vacant	Vacant	Vacant	Vacant	Vacant	Vacant	Vacant
RM 50.5 L	Vacant	Polygynous and 2; fledged 1 BHCO	2 pairs; failed	Pair (fledged 1 WIFL) Single	Pair (fledged 1 WIFL) Single	Pair (fledged 1 BHCO) Single	Pair w/ 3 nestlings, fledge unlikely	Pair w/ wifl nestlings, outcome unknown	Pair w/1 bhco, 2 wifl nestlings, fledged 1 wifl
RM 51.4 L	Single ?	Vacant	2 pairs; failed	Single	Single	Single	Vacant	Vacant	Vacant
RM 65.3 L	Vacant	Not surveyed	Single	Single	Vacant	Vacant	Vacant	Vacant	Vacant
RM 71.1 L	2 pairs; 3 young fledging	Pair (failed) Single	Vacant	Vacant	Vacant (Single on 1 June visit)	Vacant	Vacant	Vacant	Vacant
RM 72.0 R	Vacant	Vacant	Vacant	Vacant	Vacant	Vacant	Vacant	Vacant	Single
RM 195.5 R	Vacant	Vacant	Vacant	Vacant	Vacant	Vacant	Vacant	Vacant	Single
Total Adults[2]	5	8	9	5	4	4	2	2	4
Adult Pairs	2	2.5	4	1	1	1	1	1	1
Young Fledged	3	0	0	1	1	0	0	1?	1

[1] Table from Johnson and Albeita, 2000. Sources for data are: Sogge and Tibbitts 1992, Sogge et al. 1993, Sogge and Tibbitts 1994, Sogge et al. 1995a, Petterson and Sogge 1996, Sogge 1998 (for 1997 data), Tibbitts and Johnson 1999 (for 1998 data), Johnson and Albeita, 2000.

[2] Resident adults detected on more than one visit (likely migrants excluded).

Proximity to standing water, exposed sand bars, or nearby fluvial marshes appears to be an important component of SWWF habitat and may be related to food supplies. The SWWF is primarily an insectivore that feeds on a variety of winged, and, to a lesser extent, non-flying insects (Droust et al. 1997). It typically hovers and gleans insects from foliage, or catches flies from conspicuous perches. SWWF also forage on sandbars, near backwaters, and at the waters edge in the Grand Canyon (Tibbitts et al.1994).

Impacts of the Proposed Action Alternative

Experimental Test Flows

8,000 cfs Steady Flows.—The 8,000 cfs steady releases are anticipated to have no direct effects to SWWF as this species completes nesting by fall and fledglings would be almost fully independent by the time these releases would occur. Analyses of the effects of the steady 8,000 cfs experiment conducted during summer of 2000 are not yet available to assist in predicting what effects steady releases at 8,000 cfs would have on the riparian community and thus on SWWF habitat. Based on observations of the effects of similar steady flows in the past, it is likely that there would be no affect on SWWF or SWWF critical habitat from this component of the proposed action alternative.

6,500 to 9,000 cfs Fluctuating Flows.—Effects to SWWF from the 6,500 to 9,000 cfs fluctuating releases would be similar to the 8,000 cfs steady releases test. Like the 8,000 cfs releases, these fluctuating releases would occur in the fall and would be too low to reach nests or nestlings; therefore no affects to SWWF or SWWF critical habitat are anticipated.

5,000-20,000 cfs Fluctuating Non-Native Fish Suppression Flows .—Tamarisk nest stands are extremely resistant to desiccation and would not be negatively affected by the low flows or rapid ramp rates in the daily fluctuations. High flows of 20,000 cfs are well below the level necessary to directly remove nests or affect fledglings and nestlings.

31,000-33,000 cfs Habitat Maintenance Flow.—Southwestern willow flycatchers would be nesting and fledging young during the time period of this test flow scenario. Even with input from the Paria River contributing up to an additional 12,000 cfs, this component of the test flows would still fall below the 45,000 cfs stage level that would be likely to flood SWWF nests or remove current SWWF nest trees.

42,000 - 45,000 cfs High Flows.—The 42,000–45,000 cfs high flow tests would occur in January through March, well before the arrival of SWWF and the establishment of territories. In Grand Canyon, SWWF generally nest in tamarisk trees. Nest trees typically lie above the 45,000 cfs stage. Tamarisk nest trees would be unlikely to sustain direct damage from the flooding event. Stevens and Waring (1988) demonstrated that tamarisk is exceptionally tolerant of flooding in the Grand Canyon, persisting through many weeks of inundation. The tamarisk trees in which the SWWF presently nest survived the >92,600 cfs flows of 1983 and therefore would not likely be scoured by these lower flows.

The wetlands and low-lying areas near SWWF nesting habitats would likely be temporarily altered by the test flow (Stevens and Ayers 1992, Stevens et al.1996). Monitoring and assessment of effects to four SWWF habitat sites following the 1996 test flow (a one-week release of 45,000 cfs) found that marshes associated with SWWF habitat were reduced in area (cover) by 13.2% to 81% by the test flow (Stevens et al.1996). Some sites recovered rapidly, surpassing pre-flood area by 6%. Other marsh areas had only regained 3% of the original area six months after the test flow.

All sites lost litter in the inundation zone. The proportion of bare ground increased significantly at half of the sites (RM 50.5L and RM 51.5L). The 50.5L site is the only site in the Grand Canyon that has been continually occupied in the last five years. This site has also had several years of successful reproduction. We assume, therefore, that the loss of litter has not interfered with successful reproduction at 50.5L. Lack of detailed site descriptions and measurements for all sites prevent applying this assumption to other SWWF sites.

The 1996 test flood also removed lower branches and scoured or buried understory vegetation and ground cover from the lower elevation areas of the SWWF habitat sites. Significant losses of understory vegetation occurred at one site, 50.5L. Again, successful reproduction since the 1996 test flood has occurred only at the 50.5L site; it can therefore be assumed that reduction in understory vegetation has not negatively impacted SWWF nesting success at 50.5L.

Long-term effects of the 42,000–45,000 cfs test flow on SWWF habitat are expected to be beneficial. Flood flows would likely rejuvenate riverside and wetland habitat, resetting the successional vegetation and creating new seedling establishment sites and expansion areas for clonal species, i.e., willow. Impacts to food resources would be minimal because SWWF forage mostly on adult, terrestrial (non-aquatic) flying invertebrates that are unlikely to be affected by the test flow or would recover promptly after the event. Diet studies by Stevens (1985) reported that riparian, plant-dwelling invertebrate populations increased rapidly following a flow comparable to the test flow in 1980 (Reclamation 1990).

Mechanical Removal of Non-Native Fish

There would be no affect on SWWF or SWWF critical habitat from mechanical removal of non-native fish. If any nesting or rearing SWWF or occupied SWWF nest trees are identified in the electrofishing reach, they would be avoided during mechanical removal activities.

Conclusion

We conclude that the Proposed Action may affect, but it is not likely to adversely affect, SWWF and SWWF critical habitat.

Bald Eagle Species Account

Distribution and Abundance

Throughout its range, 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 (Service 1983). As a result, the bald eagle was federally listed as endangered on March 11, 1967 (Service 1967). Although bald eagles face numerous threats throughout the 48 states, they have recovered from dramatic population declines over the past several decades. Consequently, on July 12, 1995, the bald eagle was dowlisted to threatened status (Service 1995). On July 6, 1999, further improvement in the bald eagle population made it possible for the Service to propose delisting of the species (64 Federal Register 36453-36464). This action is still in progress.

The bald eagle occurs throughout North America from Alaska to northern Mexico, and commonly breeds in the northern portion of its range (Stahlmaster 1987). The Service (1999) estimated that the breeding population exceeded 5,748 occupied breeding areas in 1998 and that the bald eagle population has essentially doubled every 7 to 8 years during the past 30 years.

A wintering bald eagle concentration was first observed in Grand Canyon in the early 1980s and has increased dramatically after 1985 (Brown et al. 1989, Brown and Stevens 1991, Brown and Stevens 1992). The wintering bald eagle population was monitored until 1995. It occurs throughout the upper half of the Grand Canyon (in Marble Canyon) and on both Lake Powell and Lake 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. 1995b). In some years, 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. 1995b). 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). Changes in environmental conditions affect foraging strategies and success of wintering bald eagles (Knight and Skagen 1988). Fluctuating flows affect eagle foraging location and strategies. On the Colorado River, most foraging occurs less than 5 meters from shore, often in isolated pools. Brown and Stevens (1992) found that fluctuating dam releases appear to influence foraging behavior of bald eagles and birds tend to shift foraging locations during changes in flow. In Grand Canyon, when conditions allow, wintering bald eagles preferentially capture rainbow trout in Nankoweap Creek rather than in the mainstream where foraging success is lower (Brown 1993, Sogge et al. 1995b). A lower mainstream success rate may be related to water depth as well as velocity and turbidity.

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 (Sogge et al. 1995b). Bald eagles there prefer roosting and feeding areas that are relatively free of vegetation. The eagle population 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 the opportunity to accumulate energy reserves needed for their long, northward migration flights and initiation of nesting.

Impacts of the Proposed Action Alternative: Experimental Test Flows

8,000 cfs Steady Flows.—Bald eagles would not be present in the Grand Canyon during the time of the 8,000 cfs steady release scenario**.**

6,500-9,000 cfs Fluctuating Flows.—Effects of 6,500 cfs to 9,000 cfs fluctuating release would be expected to be similar to those of the 8,000 cfs steady flows in fall.

5,000-20,000 cfs Fluctuating Non-Native Fish Suppression Flows.—Fluctuating releases offer additional foraging opportunities for bald eagle through exposure of isolated pools and stranded trout on shorelines. Sogge et al. (1995b) found a statistically significant inverse correlation between trout numbers in Nankoweap Creek and previous days dam releases; more trout were detected in Nankoweap Creek as river flows decreased. Yet, no significant correlation was detected between the number of eagles present at Nankoweap Creek and the daily minimum and maximum flow releases. No correlation was found between Colorado River flows (minimum and maximum flows) and eagle abundance in Grand Canyon. Minimum flows during Sogge's study were not as low as in this component of the Proposed Action (8,343 cfs vs. 5,000 cfs).

During studies in 1991-1995, trout populations in the creek varied greatly with some years having very low populations (unrelated to Colorado River flows). Bald eagle were not attracted to Nankoweap Creek during these times and foraged instead in the Colorado River and other tributaries.

Fluctuating releases of this scenario are designed to reduce trout populations by interfering with and disrupting spawning and through reduction in recruitment of young fish. A realistic estimate is that there would be a 20% reduction in young-of-year trout. This reduction in trout population would likely have no affect on bald eagles in the short-term as bald eagles usually take adult fish. There may be long-term effects, however, resulting in a reduced number of available prey fish as this age class reaches catchable size. Coupled with multi-year mechanical removal of non-native fish, long-term effects would likely reduce available bald eagle prey. We conclude that the 5,000-20,000 cfs fluctuating non-native fish suppression flows may affect bald eagles, but we anticipate this affect would not occur until after the two-year period of this component of the Proposed Action.

31,000-33,000 cfs Habitat Maintenance Flow.—Bald eagles would not be present during most of the time period under which this test flow would occur (July - December). Bald eagles begin arriving in Grand Canyon in late November. High flows would likely increase turbidity, but Colorado River water would already have been made turbid from the tributary inflow. Given the short time span of this test flow and its low likelihood of occurring before bald eagles enter Grand Canyon, it is unlikely that bald eagles would be affected by these flows.

42,000–45,000 cfs High Flows.—Wintering bald eagles would be present in Grand Canyon during this scheduled test flow. Eagle abundance tends to peak in February and then rapidly declines with birds usually gone by the end of March. Foraging conditions in river, shore, and isolated pool habitats are highly variable and are directly influenced by river flows. Low river flows result in eagles capturing and scavenging proportionally more prey from isolated pools and adjacent shore habitat. As river flows increase, these habitats are inundated, reducing or eliminating prey availability. Intermediate and high river flows result in a shift to greater use of creek habitat, e.g. Nankoweap Creek. Increased turbidity and velocity of flood flows likely also play a role in foraging success and location, tending to lower the success rate as these factors increase (Knight and Skagen 1988). The opportunistic nature of bald eagle foraging suggests that eagles may be able to compensate for a loss of prey from isolated pools by foraging in Nankoweap (or other creeks with trout populations). This form of compensation is only possible if spawning trout are present in the creek(s). Eagles in the river corridor that were not near such creeks would possibly experience a temporary reduction in foraging opportunities or reduced foraging success during the 42,000–45,000 cfs two-day high flow. As flows drop to 8,000 cfs for aerial photography purposes, trout may become more available if they are stranded in isolated pools.

Turbidity in the Colorado River would increase during this high flow scenario. But, due to the short time span of this test flow, increased turbidity would not affect bald eagle foraging beyond the two days of this test flow. Bald eagles would likely increase tributary foraging.

Mechanical Removal of Trout

The goal of the mechanical removal component of the proposed action alternative is to reduce the trout population around the Little Colorado River. GCMRC estimates a removal of 3000–9000 rainbow trout with the first mechanical removal trip (S. Gloss, GCMRC, written communication). A total of six trips, covering 10 miles above and below the Little Colorado River are planned for the first year of the proposed action. It is unknown if trout from adjoining reaches would move into the newly available habitat, thereby depleting adjoining reaches, nor is the length of time known before the reach is repopulated to pre-treatment levels.

Part of the Little Colorado River reach is considered wintering bald eagle foraging habitat. The removal would affect approximately 6 miles of 77.5 miles of bald eagle habitat (dam to 1 mile below LCR). At this time, only a crude estimate of the effects to bald eagle can be made. If the assumption is made that the 6 miles would be substantially depleted of trout, then it can be reasoned that 8% of bald eagle foraging habitat would be affected or largely removed, at least temporarily, from foraging opportunities.

Short-term effects of mechanical removal will differ from those of dam releases. Many of the trout removed by mechanical removal will be in the size range of individuals chosen by bald eagles for food, whereas fluctuating dam releases will have their greatest effects on eggs, embryos, and fry not used by these birds. Also, the effects of mechanical removal will likely be concentrated in a reach of the Colorado River where bald eagles are known to forage, whereas the effects of dam releases will be widespread along the river. Only in the longer term, beyond the two years proposed for this test, is it likely that the combined effects of dam releases and mechanical removal would be evidenced to the bald eagle population. This longer term effect will need to be measured through monitoring as part of the adaptive management program.

Conclusion

We conclude that the Proposed Action may affect, and is likely to adversely affect, the bald eagle in Grand Canyon.

California Condor Species Account

Distribution and Abundance

The California condor (*Gymnogyps californianus*) was listed as endangered on March 11, 1967, in a final rule published by the Service (32 FR 4001). On October 6, 1996, the Service announced its intent to reintroduce California condors into northern Arizona/southern Utah and to designate these birds as a nonessential experimental population (equivalent to a "threatened" status)[1] under the Endangered Species Act (Service 1996). There is no critical habitat designation associated with the experimental population.

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[1] The conditions under which a population can be designated as experimental are: the population must be geographically disjunct from any other wild populations of the same species, and the Service determines that the release will further the conservation and recovery of the species (USFWS 1996). Before an experimental population can be released, section 10(j) of the Endangered Species Act (the Act) requires that a determination be made by the Service whether the population is either "essential" or "nonessential" to the continued existence of the species. An experimental population determined to be essential is treated as a threatened species. An experimental population determined to be nonessential is treated as a species proposed for listing as threatened. The exception is a nonessential population located within the National Park System or National Wildlife Refuge System lands will be treated as a threatened species for purposes of section 7(a)(2) of the Act. A designation of nonessential experimental limits the application of section 7(a)(2) of the Act. For the purposes of section 7, the nonessential experimental population is treated as a proposed species except on National Wildlife Refuge System and National Park System lands.

The reintroduction area consists of remote Federal and Native American Reservation lands with limited private lands. The designated experimental population area of the California condor includes portions of three states - Arizona, Nevada, and Utah. As part of the management strategy for this population the Service will relocate any condor within the experimental population area, including the National Park System, to avoid conflicts with ongoing or proposed activities (Service 1996).

On October 29, 1996, six California condors were released at Vermillion Cliffs in northern Arizona. Since then, there have been additional releases and the current (spring 2002) experimental population stands at 32 (California Condor Reintroduction Program 2002).

Life Requisites

California condors are among the largest flying birds in the world. Adults weigh approximately 10 kilograms (22 pounds) and have a wingspan up to 9 ½ feet (2.9 meters) (Kofer 1953, Wilbur 1978). California condors nest in various types of rock formations including protected crevices, overhung ledges, and potholes; nest sites are usually remote and at elevations far above the valley floor. Condors reach sexual maturity and attain adult plumage and coloration by 5-6 years of age. Breeding is likely between 6-8 years of age. If the nesting cycle is successful, pairs produce one egg, usually every other year. Average incubation period for a condor egg is about 56 days. Parental care is lengthy and fledglings may not become fully independent until the following spring (Kofer 1953, Snyder and Snyder 1989, Service 1996).

California condors are carrion-eaters. They are opportunistic scavengers, preferring carcasses of large mammals (Kofer, Wilbur 1976) but will feed on rodents and, more rarely, fish. Ungulates, including the carcasses of domestic livestock, are expected to be the primary sources of food for condors released at the Vermilion Cliffs (Service 1996). Mule deer (*Odocoileus hemionus*) , desert bighorn sheep (*Ovis canadensis nelsoni*), and pronghorn (*Antilocapra americana*) are residents of the region. These ungulates become available to condors as natural mortalities, hunter kills and road kills. Most California condor foraging occurs in open terrain. California condors apparently do not locate food by smell. Typical foraging behavior includes long-distance reconnaissance flights, lengthy circling flights over a carcass, and lengthy waits at a roost or on the ground near a carcass (Service 1996). Condors will also cue into the activity of ravens and other scavengers to locate food sources.

Depending upon weather conditions and the hunger of the bird, a California condor may spend most of its time perched at a roost. Roosting provides opportunity for preening and other maintenance activities, rest, and possibly facilitates certain social functions (Service 1996). California condors often use traditional roosting sites near important foraging grounds. Cliffs and tall conifers, including dead snags, are generally used as roost sites in nesting areas. Although most roost sites are near nesting or foraging areas, scattered roost sites are located throughout the range.

The beaches of the Colorado River through the Grand Canyon are frequently used by the Arizona/Utah experimental population of California condors (Sohie Osborn, Peregrine Fund, personal communication). Activities include drinking, bathing, preening, playing, and possibly feeding on the occasional fish carcass. Condor monitors are noting an increase in interaction between rafters and condors in 2002 as rafting parties seek out unused beaches for lunch stops, exploration, and close observance of condors. There have also been several instances of the immature condors approaching campsites, possible keying into ravens who are experienced camp raiders.

The decline in California condor numbers has been attributed to illegal collection of eggs and birds, poisoning from predator control, lead poisoning, effects of DDT and other organochlorines, and the increase in roads and houses in open country needed by condors for foraging (Kiff et al.1979, Service 1996). Their slow rate of reproduction and high number of years spent reaching breeding maturity make the condor population as a whole more vulnerable to these threats.

Impacts of the Proposed Action Alternative: Experimental Test Flows

There would likely be no adverse effect to condors from the various flow scenarios described in the action alternative. Condors do not routinely forage along the river corridor nor do they appear to rely on any particular vegetation component associated with beach use. Nesting occurs far above the river corridor. California condors do use the Colorado River and beaches for bathing, drinking, resting, and playing. Habitat maintenance flows and the short-term high flow are designed to increase and/or restore beaches of the Colorado River through Grand Canyon. It is assumed that the results of these flows will be beneficial to the California condor by increasing the amount of beach habitat available to condors.

In summary, the Proposed Action Alternative may have short-term and possibly long-term affects on California condors through providing them with an increased area of beach habitat. There are no known negative effects or cumulative concerns associated with the California condor and this action.

Mechanical Removal of Non-Native Fish

The mechanical removal of non-native fish would occur during night time when California condors are not foraging and very likely not near the river. Disposal of trout would be accomplished in such a way that no carrion is created that would create attractive nuisance for condors.

Conclusion

It is our conclusion that the Proposed Action Alternative may affect, but is not likely to adversely affect, California condors.