RECLAMATION

Managing Water in the West

COLORADO RIVER BACKWATERS ENHANCEMENT Species Profiles Report

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Submitted by:
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INTRODUCTION

Project History

In response to the 1994 designation of critical habitat along the lower Colorado River for endangered fish species, the U.S. Bureau of Reclamation (Reclamation) and other Federal, state, and tribal agencies formed a partnership to develop and implement the Multi-Species Conservation Program (MSCP). The program is intended to protect threatened, endangered, and sensitive species and their habitats along the lower Colorado River while maintaining river regulation and water management requirements. The MSCP was evaluated through an Environmental Impact Statement as required by the National Environmental Policy Act of 1969 (42 USC §4321 et seq.) and was recently finalized.

In 1986, BIO-WEST conducted a study for Reclamation of the backwaters along the Colorado River between Davis Dam and the Southerly International Boundary (SIB) with Mexico[1]. BIO-WEST mapped existing backwaters and developed a model to classify general wildlife and fish habitat values for these backwaters. The mapping and classification system developed during these studies have allowed Reclamation to determine the extent of backwaters, to assess existing backwaters for habitat value, and to determine factors necessary when constructing backwaters for wildlife and fish.

Purpose and Description of the Study

Under the auspices of the MSCP, Reclamation must restore over 8,000 acres of riparian, marsh, and backwater communities to provide habitat for 26 covered species. In order to develop criterion for rating terrestrial and aquatic backwater communities, habitat requirements must be determined to allow for the assessment of habitat quality. The primary purpose of this project was to determine habitat requirements for backwater, marsh, and associated riparian obligate MSCP covered species by conducting an extensive literature review of published and unpublished literature. Upon completion of species profiles, gaps in our knowledge of each species' habitat requirements were identified and research recommendations were made.

Profiles for each of the covered species include the following information: Brief species introduction and current status, distribution, systematics including geographic variation when applicable, migration including timing when applicable, habitat requirements during breeding and non-breeding periods, food habits, and issues related to conservation and management including research recommendations.

[1] Holden, P. B., R. D. Hugie, L. Crist, S. B. Chanson, and W. J. Masslich. 1986. Development of a fish and wildlife classification system for backwaters along the lower Colorado River. Final report submitted to U.S. Bureau of Reclamation, Lower Colorado Region, Boulder City, NV, by BIO-WEST, Inc., Logan, UT. 205 pp.

Each group of species is presented as follows:

1. **Chapter 1: Fish Species** - This chapter presents profiles for the Razorback Sucker, Bonytail Chub, and Flannelmouth Sucker.

2. **Chapter 2: Avian Species** - Species profiles for the Yuma clapper rail, California black rail, least bittern, southwestern willow flycatcher, and western yellow-billed cuckoo.

3. **Chapter 3: Mammalian Species** - Species profile for the Colorado River cotton rat.

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CHAPTER 1: FISH SPECIES

Introduction

The razorback sucker (*Xyrauchen texanus* [Abbott]), flannelmouth sucker (*Catostomus latipinnis* [Baird and Girard]), and bonytail (*Gila elegans*) are endemic fish species of the Colorado River Basin and three of four endemic Colorado River fishes that Minckley (1973) described as rare, if not extinct, in the Lower Colorado River Basin (LCRB). Historically, razorback sucker was used extensively as food by native cultures, and populations were commercially harvested as recently as 1949 (Ellis 1914, Miller 1955, Minckley 1973). Razorback sucker are presently considered endangered by the U.S. Department of the Interior (USDI) (USFWS 1991).

The flannelmouth sucker has been regarded as one of the most abundant and widespread native fishes in the Upper Colorado River Basin (UCRB) (Tyus et al. 1982). However, flannelmouth sucker have been considered rare in the LCRB due to a multitude of habitat alterations (Miller 1952, Miller and Lowe 1964, Holden 1973, Minckley 1973, Holden and Stalnaker 1975, Mueller and Marsh 2002). Although not listed as Federally endangered, populations of flannelmouth sucker are being monitored with special concern (USFWS 1994), especially in the LCRB. In 1976 flannelmouth suckers were introduced below Davis Dam and have persisted to the present time, representing the only known, successful reintroduction of a native fish species to the mainstem of the Lower Colorado River (Mueller and Marsh 2002, Mueller and Wydoski 2004). While being one of the more common native fishes in the Colorado River and its larger tributaries, little is known of the basic biology of the flannelmouth sucker particularly when compared with data available for the endangered razorback sucker.

Bonytail was once regarded as one of the most common fish species in the Colorado Basin, occurring throughout the mainstem river as well as its tributaries. However, bonytail is currently considered by many to be one of the most endangered vertebrates in North America (Sowka and Brunkow 1999, Muller and Marsh 2002) and, as such, information regarding its specific habitat preferences is lacking. Bonytail are also considered endangered by the USDI (USFWS 1980).

As this report is largely habitat oriented, and as much of the information pertaining to native fishes and their habitat use stems from work conducted in the more "natural," riverine setting typical of the UCRB, it is imperative that the physical attributes of the UCRB and LCRB are understood. For example, there are dramatic differences in habitat type and the overall river disposition evident between the UCRB and the LCRB. Typically, reaches in the UCRB tend to consist of higher-gradient, erosional, dynamic sections compared with the more depositional, channelized, and homogenous habitats types that are occasionally interspersed with highly vegetated, perennial, and permanently connected off-channel backwater impoundment structures typical of the LCRB (Bradford and Gurtin 2000). Furthermore, although the river in the upper basin can be constricted to a single, monotypic channel, it also maintains a high degree of secondary channels and braided reaches, and an overall greater degree of habitat complexity, particularly when compared with the channelized reaches of the LCRB. Obviously, this change from a more natural riverine system to a straightened, monotonous, and channelized system has pronounced impacts to the types and densities of various backwater and other slow-water habitat types currently available in the LCRB. For example, typical UCRB literature describes slow-water habitats as backwaters, pools, debris

pools, eddies, edge pools, sand shoals, cobble shoals, slackwaters, embayments, pocket waters, and a multitude of other terms in an attempt to classify the dynamic and complex nature of the more natural river. In addition to occurring in greater variety, certain UCRB habitats also tend to be smaller and ephemeral, often only lasting for a few weeks and changing as river flows change. However, in LCRB literature, habitat descriptions are limited since this is a more human-influenced system. For example, backwaters in the LCRB have been described as off-channel ponds, isolated impoundments, or as refuge lakes that may or may not be highly separated from the main channel, and they are much more permanent than backwaters in the UCRB. While typical and common descriptive riverine terms can still be and typically are applied to the river in the lower basin, it should be recognized that the overall LCRB system is different than the UCRB system. While this can be viewed in both a positive or negative light, the LCRB, and the recent interest in creating habitat conducive to native fish recovery efforts, provides an exciting opportunity (both from a research and management perspective) to apply, develop, and create habitats that may allow for a more natural aquatic ecosystem to exist, particularly given the interrelationships that science has historically drawn between habitat complexity and fish population dynamics (Hayes et al. 1996). While it should not be implied that the UCRB and LCRB were historically identical in terms of their geomorphology, it should be noted that habitat differences are presently evident between the upper and lower basins of the Colorado River.

The three fish species of concern in this report have been included under the auspices of the Lower Colorado River Multi-Species Conservation Plan (MSCP), with the goal of working towards the recovery and persistence of rare fishes in the LCRB. As a result, Reclamation has contracted BIO-WEST to determine the extent of Lower Colorado River backwater habitats and their value to native fishes, and determine factors important to native fishes when constructing backwaters that may be more directly tailored to individual species needs. This section serves as an initial step in investigating all known physical and biological requirements of these unique fish species and is based on an extensive literature review documenting and outlining information known about the native fish species. It will help to identify potential gaps in the current knowledge of the species, which may serve to better focus future research efforts. The following species habitat profiles will be used to develop at a classification system that can be used by Reclamation to determine the ecological value of a backwater relative to each of the three native fishes.

Razorback Sucker

Distribution

Razorback sucker was historically widespread and common throughout the larger rivers of the Colorado River Basin, from Sonora and Baja, California, into Arizona, Colorado, Nevada, New Mexico, and Wyoming (Minckley et al. 1991, Marsh 1996). Gilbert and Scofield (1898) noted particularly high razorback sucker abundance in the LCRB near Yuma, Arizona, however, Bestgen (1990) indicates that razorback sucker may have historically been uncommon in the turbulent canyon reaches of the LCRB, citing research by Tyus (1987) and Lanigan and Tyus (1989) that suggests that razorback sucker in the Green River (the largest known riverine population) were typically found in calm, flatwater river reaches, not turbulent, fast-water canyon reaches. This trend is evident even within basins, as razorback sucker are typically collected in sand-bottomed, low-gradient, flatwater reaches outside of the spawning period. Razorback sucker have persisted in

several of the reservoirs that were constructed in the LCRB; however, these populations were comprised primarily of adult fish that apparently recruited during the first few years of reservoir formation (Bestgen 1990). Residual lacustrine populations of long-lived adults then disappeared 40 to 50 years following reservoir creation and the initial recruitment period following reservoir creation (Minckley 1983, McCarthy and Minckley 1987). The largest reservoir population, estimated at 75,000 in the 1980s, occurred in Lake Mohave, Arizona and Nevada, but it had declined to less than 3,000 by 2001 (Marsh et al. 2003). Today, the Lake Mohave population is largely supported by stocking captive-reared fish (Marsh et al. 2003, Marsh et al. 2005). Catches of razorback sucker were reported often from the early 1940s through early 1980s in the LCRB (Minckley 1983, Marsh and Minckley 1989). More recently, over 12 million razorback sucker have been stocked into the LCRB with limited success in retention and survival (Mueller et al. 2003).

To date, the only substantial natural razorback sucker recruitment (low, yet steady numbers) and documentation of razorback sucker progression through all life stages in the LCRB occurs in Lake Mead, with limited and sporadic captures of naturally occurring fish throughout the remainder of the LCRB (Marsh and Minckley 1989; Holden et al. 1997, 1999, 2000a, 2000b, 2001; Abate et al. 2002; Welker and Holden 2003; Welker and Holden 2004).

Historical Habitat Modifications

Numerous researchers have identified that the major factor contributing to the decline of razorback sucker and other large-river fishes has been the construction of mainstem dams and the resultant cool tailwaters and reservoir habitats that replaced a once warm, dynamic, riverine environment Holden and Stalnaker 1975, Joseph et al. 1977, Wick et al. 1982, Minckley et al. 1991). This change in the physical environment presumably allowed for an increase in competition and predation from non-native fishes, which are successfully established in the Colorado River and its reservoirs and have also contributed to native fish population declines (Minckley et al. 1991). For further detailed information including examples, ramifications, and research needs pertaining to the effects of habitat modifications on native Colorado River fishes, please see U.S. Fish and Wildlife Service (USFWS 1998, 2002), Minckley et al. (1991), Tyus (1990), and Tyus and Karp (1990).

Systematics and Morphomethrics

Xyrauchen is one of three monotypic genera of the family Catostomidae. According to Bestgen (1990) and the USFWS (1998), Abbott (1861) originally described the razorback sucker as *Catostomus texanus*. Subsequent classifications were made by Kirsch (1889), Jordan (1891), Hubbs and Miller (1953), LaRivers (1962), and Minckley (1973). Meristic and morphological descriptions given by Abbot (1861), Ellis (1914), Hubbs and Miller (1953), Minckley (1973, 1983), Moyle (1976), Snyder and Muth (1990), and McAda and Wydoski (1980), as cited in Bestgen (1990) follow below:

> The razorback sucker is distinguishable from all other catostomids by its unique, abruptly rising, bony, dorsal keel rising posterior from the head. Body shape is elongate, robust, and somewhat laterally compressed. The caudle peduncle tends to be short and deep. An enlargement of the interneural bones forms the distinctive razor-like keel, providing basis for the common name, razorback sucker. The

> moderate sized mouth has a clefted lower lip, and lateral margins of the lips are continuous and rounded. Razorback sucker have elongated heads with a flattened dorsal surface and well developed fontanelle. Primary dorsal fin rays are usually 14-15, primary anal fin rays 7, vertebrae 45-47, scales in the lateral series range from 68-87, with gill rakers containing 44-50 on the first arch. Body coloration is dark brown to olivacious on the upper dorso-lateral surfaces and ranges from yellow to white on the lower ventro-lateral surfaces. Adults can reach up to 1,000 millimeters (mm) total length (TL) and weigh 5-6 kg, but are more typically found within the 400-700 mm TL range, weighing less than 3 kg. During spawning, razorback suckers are sexually dimorphic, with breeding males showing bright yellow and orange laterally and ventrally, dark dorsal surfaces, and tuberculation present, especially on the anal and caudle fins.

Furthermore, Eastman (1980) described razorback morphology, based on skeletal measurements as being heavily ossified, thickened and likely adapted to the strong river currents historically occupied by this species. Larval stages are best described in Snyder et al. (2004).

Hybridization

As reviewed by Bestgen (1990), hybridization between razorback sucker and other native Colorado River catostomid species has historically been documented to occur. Most often, razorback sucker have been shown to hybridize with flannelmouth sucker, but they may also hybridize with Sonora suckers (*Catostomus insignis*), and other native catostomids (Hubbs et al. 1943, Hubbs and Miller 1953, Holden 1973, Holden and Stalnaker 1975, McAda and Wydoski 1980, Minckley 1983, Bozek et al. 1984, Tyus and Karp 1990, Douglas and Marsh 1998). Buth et al. (1987) used allozymic data to directly quantify presumed introgression in the range of 0-5% toward flannelmouth sucker and 0-3% toward razorback sucker. Furthermore, in a natural river setting, Ryden (2000) noted adult flannelmouth sucker were captured consistently over the same cobble-bottomed riffles as mature, adult razorback sucker, suggesting concern for possible hybridization in San Juan River populations due to an overlap in physical habitat usage of the adult life stage of both species.

Habitat

Adults

Historically, razorback sucker inhabited virtually all components of riverine habitat; in particular, low-velocity habitats such as backwaters, sloughs, oxbow lakes, and other slackwater habitats within the main channel were important for razorback sucker (Holden 1973, Holden and Stalnaker 1975, Behnke and Benson 1980, Minckley 1983). Seasonally submerged off-river habitats, including bottomlands and other marsh-like, lowland habitats, may have also been important habitat for razorback sucker prior to the construction of mainstream dams and the resultant changes in flow regimes, especially during spring-runoff periods (Tyus and Karp 1989, Bestgen 1990, Osmundson 2001).

More recent authors have documented that habitat selection by adult razorback sucker changes seasonally. Tyus and Karp (1990) document habitat use by adult razorback sucker to consist of flooded areas during spring months. Radiotelemetry efforts by Tyus (1987) identified adult fish

utilizing near-shore runs during the spring, but they subsequently shifted habitat use during the summer to shallow waters associated with submerged mid-channel sandbars, with little use of backwaters. This suggests that the use of backwaters by razorback sucker may be overstated and an artifact of relatively easy capture with electrofishing rather than actual habitat use and preference. Osmundson and Kaeding (1989) reported adult razorback using pools and slow eddies from November through April, shifting to runs and pools from July through October. They also note increased backwater habitat use by adult fish during the months of May and June, the typical UCRB spawning period.

More detailed information of razorback sucker habitat use, needs, and selection is provided by Ryden (2000), based on radio-telemetered razorback sucker occupying the dynamic and relatively natural (by today's standards) San Juan River of the UCRB. During pre-runoff periods (March and April) tagged fish were found to use a variety of low-velocity habitats. Habitat usage included pools, eddies, shoals, and backwaters, with evident seasonal use of fast-water habitat types. Ryden (2000) indicates that the majority of these habitats were located along the inner edge of large bends in the main river channel. Specific habitats selected for during the month of March were primarily considered to be slow or slackwater habitat types, with the most highly selected habitat type being pools. In March the mean water depth at fish contact locations was 2.7 feet (ft), with warmer temperatures at razorback sucker locations than adjacent main-channel habitats (mean = 10.9 degrees C, main channel = 9.8 degrees C). Mean bottom velocity in March was 1.5 feet per second (ft/s), while mean water column velocities averaged 1.7 ft/s. During April razorback sucker primarily selected low-velocity, sand-shoal habitats, as well as other backwater and pool areas. April was reported to be the only month of the year in which sand-shoal or backwaters were the most commonly selected habitat types. Furthermore, in April mean water depth at fish location was 2.3 ft, with razorback sucker seeking warmer temperatures (13.0 degrees C) than the main channel (12 degrees C). Mean bottom velocity was found to be 0.6 ft/s, and average column velocity was 1.0 ft/s. During May habitat selection demonstrated that razorback sucker showed a strong preference for eddy habitats located along the inside of large river bends. Also during May razorback sucker displayed a strong affinity for mid channel cobble riffles and run-riffles, as well as shoreline cobble-shoal, run-type habitats. Fish collected in these areas appeared to be exhibiting spawning behavior coinciding with the ascending limb of the hydrograph (see spawning ecology section below). Mean water depth usage in May was 3.3 ft, and temperatures in habitats utilized by razorback sucker were the same as those recorded for the main channel (14.8 degrees C), with bottom velocities averaging 0.8 ft/s and water column velocities averaging 1.4 ft/s.

During runoff, or descending limb of the hydrograph, and post runoff months (June and July), razorback sucker habitat selection in the San Juan River was dominated by use of inundated vegetation. During high-flow periods radio-telemetered razorback sucker were found utilizing the river's margins and other low-velocity areas. Ryden (2000) suggests that habitat selection in June was likely the result of fish avoiding high, turbulent flows, as well as foraging forays. Water depths utilized in June averaged 3.9 ft, and June was the last of three consecutive months where water temperatures at fish locations were warmer than adjacent main-channel areas (15.0 degrees C versus 14.8 degrees C). Mean bottom velocity at the June contact locations was 1.7 ft/s, while the water column velocities were 2.0 ft/s. Habitat use during July, as flows began to recede, was reported to be very similar to the habitat use described during May, with eddies being the dominant habitat type

utilized. Ryden (2000) reports the mean bottom velocity during July to be 0.7 ft/s, mean column velocity to be 1.6 ft/s, and the average temperature occupied by razorback sucker 21.1 degrees C.

During the post-runoff summer and fall months (August through October) Ryden (2000) found razorback sucker displaying unique habitat selection, compared with the periods mentioned earlier. For example, a strong shift of habitat types, from slow-water to main-channel and fast-water habitats, was noted. No low-velocity habitat types were selected for during the summer-fall base-flow period. For example, during August, razorback sucker were typically found utilizing main-channel runs and shoal-runs. Likewise, depths utilized by razorback sucker tended to increase, with the mean depth at location being 6.2 ft. During September similar habitat use was observed, and in October tagged fish were only observed using main-channel runs with mean water depths of 4.0 ft. These spring-to-summer habitat shifts in the San Juan River are similar to the Green River razorback sucker habitat shifts seen by Tyus and Karp (1989).

Habitat selection during the fall-winter transitional period (November) resulted in fish being located only in mid and main-channel run habitats (Ryden 2000). Mean water depth at fish location was 3.8 ft, and mean temperature at fish location was reported as 5.3 degrees C. Mean bottom velocity at fish location in November was 1.2 ft/s, while the mean column velocity was 1.7 ft/s.

During winter base-flow periods (December through February) only two habitat types were selected. Main-channel runs and edge pools were selected for during early December, when daytime water temperatures surpassed 3.0 degree C. However, later in December, as temperatures began to decline, radio-tagged razorback sucker were observed utilizing edge pools only, and fish became notably more sedentary. Mean temperatures throughout the river were 3.0 degrees C, and velocities averaged 1.3 ft/s on the bottom of the river and 1.5 ft/s higher in the water column. In January razorback sucker were only found utilizing edge pools. They only ventured from these pools when water temperatures rose above 3.0 degrees C and then only for very short time periods. Mean temperatures throughout the river were 1.3 degrees C, with mean bottom velocities of 0.5 ft/s and mean column velocities of 0.6 ft/s (Ryden 2000).

During February tagged razorback sucker once again became fairly active and selected edge pools, main-channel runs, eddies, and shore runs. Water depth at fish location averaged 3.7 ft. Mean velocity at point of contact was 1.0 ft/s on the bottom, as well as mid column. Ryden (2000) reports the mean temperature at point of razorback sucker contact to be identical to that of adjacent main-channel habitats, 4.3 degrees C (Ryden 2000).

Comparing the specific findings of Ryden (2000) with researchers in other UCRB locations, similar trends of razorback sucker habitat use are evident. For example, water velocity selection by adult razorback sucker is also typified by seasonal shifts in preferences. Tyus (1987) noted that during the summer, razorback sucker typically were found utilizing velocities averaging 0.5 m/s, while in the winter months adult fish were typically found in currents moving 0.03-0.33 m/s. These findings corroborate hypotheses and findings of Lanigan and Tyus (1989) and Minckley et al. (1991) that few adult razorback sucker utilize swift, whitewater habitats (e.g., Marble and Grand Canyons of the LCRB), although other efforts have documented movement of radio-telemetered fish through these locations (Tyus and Karp 1990). Furthermore, it becomes apparent that razorback sucker in a natural river setting do not appear to utilize solely backwater habitat types, although it appears that

these habitats are important during specific times of the year. Lastly, adult razorback sucker have been reported to select shallower depths during the summer months (0.9-1.65 m) while typically utilizing deeper depths during the winter months (1.65-2.16 m) (Osmundson and Kaeding 1989).

In contrast, hatchery-raised, sonic-tagged razorback sucker in the LCRB were found to use backwater habitat types more frequently, in relationship to their availability, compared to other, main-channel habitat types in the LCRB throughout every season of the year (Bradford et al. 1998, Bradford and Gurtin 2000). However, in contrast, efforts by Lee (2005) indicate that further telemetric efforts in the LCRB show that adult fish prefer main-channel habitats, as virtually all contacts made with radio-telemetered fish were in the main-channel areas typically associated with eddies and other slow-moving, near-shore, sand depositional habitats, not backwaters. This is very recent information, and the reasons for the difference in habitat usage are still being studied. More specific to findings presented in Bradford et al. (1998) and Bradford and Gurtin (2000), Slaughter et al. (2002) reports that adult razorback sucker prefer large, irregularly shaped backwaters with a mean depth greater than 1.5 meters (m). Backwater size and depth were found to be more important in determining LCRB razorback sucker habitat usage than were water quality factors such as turbidity, pH, or temperature (Slaughter et al 2002). Mueller (1989) observed spawning razorback sucker in LCRB riverine habitat. This habitat was a main-channel, backwater interface at the mouth of a dry wash, and substrates consisted of scoured sands and gravels. Habitat depths were between 3.9-6.6 ft, and velocities were reported between 0.0-1.2 ft/s.

Discrepancies in annual habitat-use findings within and between the UCRB and the LCRB have been attributed to a general lack of contacts with fish, particularly in the LCRB, but more likely are thought to reflect dramatic differences in habitat availability between the UCRB and the LCRB. For example, as previously stated, UCRB reaches tend to consist of higher-gradient, erosional, dynamic sections compared with the more depositional, channelized, homogenous habitats types that are occasionally interspersed with highly vegetated, perennial, and permanently connected off-channel backwater impoundment structures typical of the LCRB (Bradford and Gurtin 2000). Lastly, based on observed habitat use in the UCRB (a more natural riverine environment), it can be speculated that the habitat preferences reported for razorback sucker in the LCRB (i.e., mainly backwater habitat use) may simply be a reflection of habitat availability in this highly altered system. It appears as though razorback sucker, although displaying extensive use of backwaters in the LCRB, may be actually (or simply) using the best available habitat, not by preference, but potentially by necessity.

One of the current habitat types presently occupied by populations of razorback sucker are lentic situations imposed by various impoundments in the LCRB. In these lentic situations, adult razorback suckers have also been documented to display interesting and rather extensive habitat use. The majority of such information suggests that lentic-dwelling razorback sucker use a wide variety of habitats, including vegetated areas, littoral shoreline habitats, and substrates ranging from silt and sand to gravel and cobble. Adult razorback sucker have been documented via sonic surveillance to typically occupy depths less than 30 m (averaging between 3.1-16.8 m) and are generally located within 50 m from the shore during winter months (less than 30 m from shore during peak spawning activity). However, during summer months, adults were located at deeper depths, often surpassing 30 m, in an effort to hold body temperatures between 18-22 degrees C, a behavior thought to maximize bioenergetics (Marsh and Minckley 1989; Holden et al. 1997, 1999, 2000a, 2000b, 2001; Mueller et al. 2000; Abate et al. 2002; Welker et al. 2003; Welker and Holden 2004). Thermal

preferendum for adult razorback sucker was estimated to lie within the range of 22-25 degrees C based on laboratory observations (Bulkley and Pimentel 1983). Information on the pH preferences specific to razorback sucker were not found, but warm-water fish species, in general, survive well within a pH range of 6.5-9.0 (Boyd 1979, Piper et al. 1982). Furthermore, the majority of backwaters investigated by Slaughter et al. (2002) ranged between a pH of 8-9. Boyd (1979) and Piper et al. (1982) also suggest that fish growth may become hindered when dissolved oxygen concentrations drop below 6.0 milligrams per liter (mg/L). However this may not be the case for razorback sucker, as early life stages have been reported utilizing backwater habitats with dissolved oxygen levels approaching 2.0 mg/L at times, with critical dissolved oxygen levels being dependant upon water temperatures (Modde 1996, Modde et al. 2001).

Spawning

The spawning season for razorback sucker has been reported to begin as early as November in some LCRB reservoirs and documented to continue through June in some populations of the UCRB. In upper basin riverine habitats, ripe razorback sucker have been collected from mid April to mid June, typically over a very limited time frame (4-5 weeks) (Tyus 1987; Osmundson and Kaeding 1989; Tyus and Karp 1989, 1990; Bestgen 1990). However, in lentic lower basin habitats the majority of spawning generally is carried out between January and April, months when water temperatures are typically within the range of 10-15 degrees C (Bestgen 1990). Male razorback sucker remain ripe for a period of 2-28 days, while females apparently are ripe for less time (2-15 days) in the Green River (Tyus and Karp 1990) but appear to have extended periods of sexual activity in lower basin reservoirs (Holden et al 2001). Although spawning razorback sucker have been collected over a variety of substrates, the majority of spawning individuals tend to be captured over clean gravel- and cobble-sized substrates (Douglas 1952, Tyus 1987, Bozek et al. 1990, Tyus and Karp 1990, Minckley et al. 1991). In UCRB rivers spawning occurs during the ascending limb of the hydrograph (et al. 2005), which apparently an important adaptive feature for larvae as discussed below.

In the Green River, when spring flows have elevated to allow access to bottomland and backwater habitats, adult razorback sucker have been documented moving in to these slightly warmer than main-channel environments (typically 2-4 degrees C warmer). This behavior has been termed "staging" since it occurs just before and during spawning, and presumably allows for additional heat units to be obtained, a strategy that is thought to stimulate gamete production and minimize the costly act of spawning bioenergetically (Tyus and Karp 1990, USFWS 1998, Holden 1999, Ryden 2000). Razorback sucker have also been documented to use warmer backwater habitat types post spawn, apparently to recover and feed (Modde and Irving 1998).

Reservoir-spawning razorback sucker have been documented to successfully spawn in various LCRB impoundments. Spawning populations have been located in Lake Mead (Jonez and Sumner 1954; Holden et al. 1997, 1999, 2000a, 2000b, 2001; Abate et al. 2002; Welker et al. 2003; Welker and Holden 2004), Lake Mohave (Bozek et al. 1984, Marsh and Langhorst 1988, Mueller 1989, Bozek et al. 1990), Lake Havasu (Douglas 1952, Minckley 1983), Senator Wash Reservoir (Medel-Ulmer 1980), and likely other locations. Spawning activities are most frequently associated with relatively shallow, flat to gently sloping shoreline areas over relatively clean gravel and cobble (Bestgen 1990). Spawning activity has been documented in depths up to 20 meters in Lake Mead, but typically occurs in less than 2 meters of water (Minckley et al. 1991; Holden et al. 1997, 1999).

Spawning fish have been documented to congregate near river inflow areas that tend to be somewhat more turbid than the majority of the available spawning areas (Jonez and Sumner 1954; Holden et al. 1997, 1999). Most of this spawning in the LCRB results in larvae but little or no recruitment, apparently due to the lack of nursery habitat for young that allows them to escape predation. Recent studies in Lake Mead have shown that spawning of reservoir-recruited fish presently occurs only in a few sites with abundant nearby vegetation and turbidity that serves as cover for the larvae. Apparently, increased turbidity and vegetation serve as cover to promote razorback sucker survival during the highly predator susceptible early life stages (Holden et al. 1997, 1999; Johnson and Hines 1999; Holden et al. 2000a, 2000b, 2001; Mueller et al. 2000; Abate et al. 2002; Welker et al. 2003; Welker and Holden 2004). This suggests that although razorback sucker will spawn in a variety of areas, only areas that promote recruitment will result in long-term population survival.

The majority of information on reproduction in lotic systems comes from UCRB research. McAda and Wydoski (1980) collected razorback sucker in spawning condition from gravel bars in water typically 1 m deep. They report substrate used by spawning razorback sucker to consist largely of cobble located in water velocities of approximately 1 m/s. Researchers in the Green and Yampa rivers of the UCRB have depicted that spawning occurs on main-channel gravel and cobble bars (McAda 1977, McAda and Wydoski 1980, Tyus 1987, Tyus and Karp 1990, Modde and Irving 1998). Bliesner and Lamarra (2005) measured substrate size and depth to embeddedness at a suspected razorback sucker spawning site on the San Juan River and compared the information with another nearby riffle. They found that the suspected spawning site had smaller substrate (average of 3.5 centimeter [cm]), deeper depth to embeddedness, and fewer fine materials than the control riffle. This suggests that razorback sucker may have narrower preferences for spawning habitat when given the opportunity to select a site than has been generally known.

Modde and Irving (1998) used radiotelemetry data to document the spawning activity of individually tagged fish at different spawning locations during their 1993-1995 study, suggesting that razorback sucker in the Green River represent a single reproductive population. Tyus and Karp (1990) used radiotelemetry to document the importance of flooded lowlands and other slackwater habitats as resting-feeding areas for razorback suckers during the breeding season. Spawning in riverine sections is associated with increasing spring flows and associated increases in turbidity (Tyus 1987, Tyus and Karp 1990, Modde et al. 2005). Razorback sucker also display an apparently strong spawning site fidelity both in lentic and lotic habitats (Mueller 1989; Tyus and Karp 1990; Holden et al. 2000a, 2000b, 2001; Abate et al. 2002; Welker et al. 2003; Welker and Holden 2004; Modde et al. 2005).

In the LCRB Mueller (1989) provides insight to riverine razorback sucker spawning below Hoover Dam and in the lower Colorado River below Davis Dam. He indicates similar observations as those described above, with razorback sucker typically spawning in water depths between 1.2-2.0 m and velocities ranging from 0.00-0.37 m/s. Most recently in the LCRB, spawning activities of 126 razorback sucker were visually observed upstream from Needles Bridge, in water approximately 1 m deep over large cobble substrates (Wydoski 2005). This is particularly interesting, as Ryden (2000) and other researchers from the UCRB (e.g., Modde et al. 2005) have associated spawning with large riffle habitats comprised of relatively clean cobble substrates, and the area described by Wydoski (2005) is likely one of the few such habitat types in the LCRB. This suggests that

razorback sucker populations (and likely other native fish populations) are selecting the best of the limited habitat available for use during the various times of the year and life history stages.

If the multitude of UCRB habitats described by Ryden (2000) and others were available in the LCRB, would we see different habitat use by riverine populations of razorback sucker? In addition, is it plausible that the spawning razorback population described by Mueller (1989) is actually utilizing the only/best in-channel riffle habitat currently available? Are there options for increasing the complexity of the lower river to promote the entire suite of razorback sucker and other native fish life-history stages through various in-channel and off-channel habitat manipulations? It seems at least plausible that populations of native fishes in the LCRB would benefit from research and manipulative efforts directed towards these ends with the overall goal of restoring habitat that is/was historically, actually selected for, rather than habitat that is for one reason or another "better" than other habitat types currently available.

Larvae

In lentic (reservoir) settings razorback sucker larvae have been collected over a variety of habitat types, but they typically are collected over or near areas frequented by adult spawning aggregates. As a result, the majority of larval fish are captured over gravel and cobble, at near-shore locations, typically at depths of 0.0-4.9 m (Sigler and Miller 1963; Minckley 1983; Bozek et al. 1984; Marsh and Langhorst 1988; Holden et al. 1997, 1999, 2000a, 2000b, 2001; Abate et al. 2002; Welker et al. 2003; Welker and Holden 2004).

In the Green River larval razorback sucker apparently have an affinity for backwater and flooded bottomland habitats (Tyus 1987, Muth et al. 1998). Historically, high spring flows flooded low-lying areas along the river and redistributed recently emerged and drifting larval razorback sucker into these food-rich backwaters and other seasonally flooded bottomlands, providing unique nursery habitats for razorback sucker (Tyus and Karp 1989, 1990; Modde 1996; Modde et al. 1996; Modde et al. 2005). Laboratory experimentation has documented the importance of backwater habitats for larval razorback sucker by evaluating nocturnal drift tendencies of young razorback sucker exposed to various degrees of flow. Drift tended to increase with an increase in flows, a scenario that would lead to downstream transport, eventually resulting in larval fish being deposited into relatively calm, low-flow environments, conditions present in backwater habitats (Tyus et al. 2000). However, construction of mainstream dams has reduced spring flows and eliminated important nursery areas. Nursery habitats have been either cut-off or do not refill due to insufficient flow in the river. Recently, a "reset" hypothesis has been suggested, which requires the flooded bottomlands to be dried every year or so and then re-flooded, thereby reducing numbers of potential predators in areas that maintain water between years (Modde 2005). Predation in nursery habitats appears to be the major limiting factor for razorback sucker in both the UCRB and LCRB today (Tyus and Karp 1989, Osmundson and Kaeding 1990, Minckley et al. 1991, Mueller 1995, Tyus and Saunders 1996, Modde et al. 2005).

Wild-spawned razorback sucker larvae have been collected in the San Juan River of New Mexico and Utah annually since 1998 (Brandenburg et al. 2005). The larvae are collected with seines in small backwaters, embayments, and other low-velocity habitats along shorelines. The San Juan River floodplain does not have large, flooded bottomlands like the Green River system, suggesting that razorback sucker larvae can survive in the face of non-native predators without large nursery

habitats. As noted below, some larvae in the San Juan River are escaping predation and have been found well into the juvenile stage. These findings are unique in that no other riverine or reservoir system with a host of predators has shown the ability to recruit razorback sucker except for Lake Mead. It also suggests that larval habitat may not be as specific (i.e., flooded bottomlands) as is being studied in the Green River system.

Juveniles

Habitat important to the juvenile life stages of razorback sucker remains relatively understudied, as catches of this size class remains minimal, presumably due to the predatory and competitory impacts of non-native species (Tyus 1987, Bestgen 1990, USFWS 1998). The majority of juvenile, riverine catches come from the UCRB (Taba et al. 1965, Gutermuth et al. 1994, Modde 1996), with almost non-existent data on juvenile habitat use from the LCRB. Brandenburg et al. (2005) recently captured wild-spawned juvenile razorback sucker in the San Juan River. They captured 125 juvenile from 30-125 mm TL in 2002, and 10 in 2003. Golden and Holden (2005) captured six wild juveniles in the San Juan River in 2004 ranging in size from 54 to 94 mm standard length (SL). All of the juveniles in the San Juan River were found using seines in shoreline habitats including backwaters, embayments, and other lower-velocity habitats. In addition, Jackson (2005) collected six other wild-spawned juveniles from 120-280 mm TL using electrofishing in the lower San Juan River in 2003 and 2004. Habitats for these fish were not recorded, but they were likely also collected from shoreline habitats.

Mueller and Marsh (1998) tracked movements of 55 hatchery-reared subadult (juvenile) razorback sucker that were released into Lake Mohave and Lake Powell. Their telemetry data demonstrated that juvenile razorback sucker utilized backwaters, vegetated areas, and rocky cavities (thought to provide important cover and food resources). This description of habitat use is corroborated by the recapture of two experimentally stocked juvenile razorback sucker in the San Juan River (Holden 1999). These fish were found occupying slackwater and backwater pools 1-3 ft deep and 1-3 degrees C warmer than adjacent main-channel habitats. Razorback sucker studies in the LCRB on Lake Mead (Holden et al. 1997, 1999, 2000a, 2000b, 2001; Abate et al. 2002; Welker et al. 2003; Welker and Holden 2004) have documented capture of wild, sexually immature, juvenile razorback suckers. This is one of the only known locations documented to produce this rather obscure life stage with periodic consistency. Juvenile fish were mainly collected near spawning areas with adult fish, although none showed signs of sexual maturity.

Studies by Modde (1996) and Modde et al. (2001) in the Green River demonstrate that juveniles and subadult razorback sucker may favor floodplain depressions when available (depth = 1-2 m, dissolved oxygen remaining greater than 2.0 mg/L, usually above 5.0 mg/L, and maximum surface temperatures 26.6 degrees C) over main-channel habitats based on habitat variables such as zooplankton density, water temperature, depth, and vegetation abundance. These researchers also indicate that growth and survival in floodplain depressions are more likely than in main-channel habitats despite heavy impacts of non-native fish predation and competition typically associated with backwater habitats. Furthermore, they suggest that draining wetlands before spring may be important for non-native fish control, while still allowing razorback sucker growth and development to occur during the remainder of the year. Additionally, Modde (2005) outlines strategies and indicates that a combination of correct flow regimes (to allow for larval razorback sucker deposition into floodplains), coupled with annual reset draining of backwaters (to remove residual non-native

fishes), increases young razorback sucker growth and survival throughout the first year of life by allowing razorback sucker to exist at sizes similar to their non-native competitors and predators. Modde (1997) documented similar growth and survival rates for young-of-year razorback sucker and carp (Cyprinus carpio) in a managed wetland. Mueller et al. (2003) demonstrates that flow acclimation of stocked razorback sucker may be another important way to bolster year-class strength of natural populations, and Marsh et al. (2005) indicate that a size increase of repatriated razorback sucker to lengths greater than 350 mm TL doubles post-stocking survival.

Reproduction

Fecundity for razorback sucker expressed in terms of number of ova per unit standard length was derived by Minckley (1983). Estimates by Minckley (1983) ranged from 1,600 ova/cm SL to 2,000 ova/cm SL. These results were based on 15 fish estimated at having anywhere from 27,614-144,000 total number of ova, accounting for 9.2-11.5% of an individual female razorback sucker's body weight. Male gonadal information was not supplied. Bozek et al. (1984) indicated that during the spawning season in Lake Mohave, male:female ratios of razorback suckers ranged from 1.2-3.6:1. They also report that approximately 80% of male and less than 65% of female razorback suckers were ripe during peak spawning activity. Current recapture data compiled by Albrecht and Holden (2005) from nine consecutive years of accumulated data indicate that adult female razorback suckers tagged with a passive integrated transponder (PIT) tags in Lake Mead were captured consistently at greater than 1-year intervals, while the majority of tagged, male fish were captured on an annual basis, thereby supporting historical ideas that female razorback suckers may exhibit non-annual spawning. Age at maturity for razorback sucker ranges from a minimum estimate of 2 years for male razorback sucker and 3 years for female razorback sucker to a maximum of 6 years for some populations, or sizes typically greater than 350 mm (Bestgen 1990).

Research efforts by Bozek et al. (1990) show that successful incubation of razorback sucker eggs in Lake Mohave occurs between 9.5-15.0 degrees C, and in the laboratory successful embryo hatching occurs at 10-20 degrees C. Hatching (eggs at a controlled 15 degrees C) was reported to occur in 5.2-5.5 days (Minckley and Gustafson 1982). Egg mortality has been attributed to fluctuating water levels, scouring by currents and/or wave action, suffocation due to silt deposition, and non-native egg predation (Minckley 1983, Bozek et al. 1984). Fertilized gametes are reported by Minckley and Gustafson (1982) as adhesive for a 3-4 hour duration post fertilization, with cleavage being completed within 24 hours, gastrulation occurring at 34 hours, and blood circulation becoming established at 117 hours. Furthermore, all fins were reported to be fully formed and ossified at 64 days (27 mm TL) (Minckley and Gustafson 1982). Papoulias and Minckley (1990) found yolk absorption to occur approximately 8 days post hatching (Minckley and Gustafson report 13 days at 15 degrees C) and that the critical period during which exogenous feeding must occur in order to avoid mortality lies between 8-19 days after hatching. Papoulias and Minckley (1990) also found that the majority of larval mortality likely occurs within 20-30 days and is a result of starvation or receiving food too late after hatching, indicating that zooplankton levels are an important driver of larval razorback sucker survival (see diet section). Larval razorback sucker are photosensitive and display diel patterns in drift periodicity (Carter et al. 1986, Burke 1995).

Diet

Razorback sucker diet composition is highly dependant upon life stage, habitat, and food availability. Upon hatching, razorback sucker larvae have terminal mouths and shortened gut lengths (less than 1 body length) which in combination, appears to facilitate and necessitate selection of a wide variety of food types. Exogenous feeding occurs at approximately 10 mm TL (approximately 8-19 days), after which larvae from lentic systems feed mainly on phytoplankton and small zooplankton, while riverine inhabiting larvae are assumed to feed largely on chironomids and other benthic insects (Minckley and Gustafeson 1982, Marsh and Langhorst 1988, Bestgen 1990, Papoulias and Minckley 1990, USFWS 1998). Papoulias and Minckley (1992) reared larval razorback sucker in three different ponds containing different densities of food resources to demonstrate that increased growth was positively related to invertebrate densities, suggesting the importance of larval food switching from algal and detrital food items to a diet enriched with invertebrates. Papoulias and Minckley (1990) show that larval razorback mortality is minimized when food levels are within the range of 50-1,000 organisms/L.

Later during growth (age and size information unknown, but at some point during the juvenile life stage), razorback sucker undergo an ontogenetic shift in mouth morphology, with the mouth becoming more inferior and allowing for more efficient access to benthic food sources. Thereafter, razorback sucker likely consume a variety of benthic-associated food items (USFWS 1998).

As adults, razorback sucker populations display unique diet compositions, depending upon whether the individual exists in a lacustrine or riverine setting (Bestgen 1990, USFWS 1998). Riverine fish consume a mixture of benthic invertebrates, algae, detritus, and inorganic materials, with little evidence of zooplankton consumption (Jonez and Sumner 1954, Banks 1964, Vanicek 1967). Lacustrine-inhabiting adult razorback sucker consumption is dominated by Cladoceran zooplankton, with some degree of algal and detrital material present in gut contents as well (Minckley 1973, Marsh 1987). While it is possible that razorback sucker may exhibit varying degrees of pelagic zooplanktivory, it is equally likely that the abundance of zooplankton noted in lentic-dwelling razorback sucker diets is simply the result of omnivorous benthic feeding. For example, Wurtsbaugh and Hawkins (1990) report large densities of zooplankton in samples collected from the profundal, hypolimnetic, water-substrate interface in Bear Lake, Utah-Idaho, particularly during daylight hours.

Age and Growth

Published growth estimates for razorback sucker vary, and available information is highly dependant upon life stage, habitat type, and overall ecological setting (Bestgen 1990, USFWS 1998). Information on growth is lacking for the early life stages of wild razorback sucker. The majority of growth information for larval and juvenile razorback sucker has been based largely on hatchery-produced fish (Brooks 1985, Marsh 1985, Marsh and Brooks 1989, Minckley et al. 1991, Mueller 1995). Razorback sucker that upon hatching are 7-9 mm can reach lengths of over 23 mm within 2 months (Papoulias and Minckley 1990). Subsequently, during the initial 6 years of life, young razorback sucker appear to grow rapidly (e.g., growth of 55-307 mm in 6 months for young razorback sucker stocked into ponds) (Osmundson and Kaeding 1989), after which growth becomes minimal (2 mm/year or less) as older age-classes are reached (McCarthy and Minckley 1987, Minckley et al. 1991). Studies of age and growth on the Lake Mohave razorback sucker population

have shown that older adult fish show very little (approximately 2 mm per year or less), if any, growth (McCarthy and Minckley 1987). Modde et al. (1996) studied the largest extant riverine population of razorback sucker in the Green River, Utah, where he also found very slow growth in adults (1.66 mm per year). The highest growth rates described have been those reported for the Lake Mead razorback population, with growth rates of adult fish approaching 10-20 mm per year (outside of isolated refugia and hatchery ponds) (Holden 1999, Holden et al. 1997, 2000a, 2000b, 2001; Abate et al. 2002; Welker et al. 2003; Welker and Holden 2004). This population has been shown to be much younger overall than either the Lake Mohave or Green River populations, which likely accounts for the higher growth rates.

Past attempts to age razorback sucker using scales and other morphological structures were unsuccessful (McCarthy and Minckley 1987). The lack of clear annular marks, or irregular annuli that do not correspond to annuli found on other structures from the same fish, made aging razorback sucker reliably from scales problematic. This inability to accurately age individual fish using scales has also been a problem for other researchers working on wild razorback sucker populations in the Colorado River (McAda and Wydoski 1980, McCarthy and Minckley 1987) and on populations of white sucker (*Catostomus commersoni*) (Beamish 1973, Quinn and Ross 1982).

McCarthy and Minckley (1987) found pectoral fin rays to be a valid structure for use in aging young razorback sucker and used otoliths to determine that the razorback sucker population in Lake Mohave was 24-44 years of age in the 1980s. Beamish and Harvey (1969) used the first four pectoral fin rays to age white sucker and found this method reliable. Quinn and Ross (1982) reported that pectoral fin rays were accurate in determining ages in younger (age 7 and under) populations of white sucker but that caution should be used in aging older and slower-growing fish.

During the early years of BIO-WEST razorback sucker studies on Lake Mead, two razorback sucker carcasses recovered from the lake were aged using both otoliths and pectoral fin rays to evaluate and develop a non-lethal technique for reliably aging razorback sucker populations in hopes of developing hypotheses pertaining to patterns of recruitment. While striving towards the development of a non-lethal aging technique, a dead, 381 mm TL razorback sucker of unknown sex was recovered from Echo Bay. Subsequently, another carcass was recovered from Las Vegas Bay (a 588 TL male). By using the combined carcasses, BIO-WEST was able to validate that in both fish, ages estimated from pectoral fin rays agreed with those obtained from sectioned otoliths. Both fish proved to be relatively young (ages 5 and 8) (Holden et al. 1999). Use of fin rays as a structure for aging has been further validated by aging multiple, known-age fish originating from Floyd Lamb State Park. Furthermore, attempts at aging razorback sucker from Lake Mead (over 80 individual razorback suckers evaluated to date) demonstrates that ages ranged from 5-35+ years of age, with the majority of data being collected from wild, naturally recruited fish (Albrecht and Holden 2005).

Flannelmouth Sucker

Distribution

The flannelmouth sucker was historically the most abundant large fish species in the UCRB (Vanicek et al. 1970, Holden 1973, Holden and Stalnaker 1975, McAda 1977). However, the flannelmouth sucker was found to be declining in the LCRB, and for a period it was largely

restricted to the Colorado River above Lake Mead and a short segment of the Salt River (Minckley 1973). Flannelmouth sucker apparently are not able to persist in large impoundments, unlike the razorback sucker. Jonez and Sumner (1954) found razorback sucker to be relatively common in Lakes Mead and Mohave, while flannelmouth sucker were only encountered rarely. This trend appears to continue, with annual catches of razorback sucker being the norm on Lake Mead, while only one flannelmouth sucker was captured during 9 years of intense BIO-WEST field efforts associated with razorback sucker studies (unpublished data) and only two flannelmouth sucker have been captured in Lake Havasu (Mueller and Marsh 2002). Interestingly, flannelmouth sucker may never have been vastly abundant in the LCRB. For example, Dill (1944) did not indicate the presence of flannelmouth sucker in the LCRB during his sampling efforts at the turn of the century. Miller (1961) reported few captures of the species the LCRB, but he stated that the species was found in the Salt and San Pedro rivers in the early 1900s. Furthermore, flannelmouth sucker was thought to be extirpated from the LCRB by the 1970s (Mueller and Wydoski 2004).

However, in 1976 the Arizona Game and Fish Department stocked 611 adult flannelmouth sucker originating from the Paria River into the mainstem of the LCRB, near Bullhead City, Arizona, for purposes of black fly abatement (Minckley 1979). Post introduction flannelmouth sucker survival was uncertain, as Minckley (1979) reported that none of the fish were observed post stocking. However, Mueller and Wydoski (2004) report that by the mid 1990s, young flannelmouth sucker were found by the Arizona Game and Fish Department and the Nevada Department of Wildlife. In 1998 Mueller and Wydoski initiated a study that documented natural recruitment in the highly altered 80 kilometers (km) reach of the Lower Colorado River downstream from Davis Dam. This introduction represents the first successful reintroduction of a native species in the highly altered LCRB (Mueller and Wydoski 2004). The successful reintroduction of flannelmouth sucker is of particular interest, because similar trends in flannelmouth sucker abundance and distribution were noted following disturbances from the construction of mainstem dams in the UCRB. For example, Wiltzius (1976) indicated that flannelmouth suckers were being replaced by introduced longnose suckers, a phenomenon likely attributable to the cold water releases from mainstream dams and competitive interactions with non-native species. However, flannelmouth sucker remain relatively abundant in the UCRB in less-altered sections of river (McAda 1977, McAda and Wydoski 1985, Tyus et al. 1982) and the most abundant large-bodied fish species in some river sections (Ryden 2005). As discussed below, the flannelmouth sucker is the “generalist” of the river, especially with regard to nursery habitat requirements, compared with the other rare species (razorback sucker, bonytail, pikeminnow and other *Gila* species). The interesting suite of characteristics that have allowed for successful flannelmouth sucker re-establishment should be scrutinized in hopes of providing important insights applicable to establishing self-sustaining populations of this and other endemic Colorado River species.

Historical Habitat Modifications

Numerous researchers have identified that the major factor contributing to the decline of flannelmouth sucker and other large-river fishes has been the construction of mainstem dams and the resultant cool tailwaters and reservoir habitats that replaced a once warm, riverine environment (Holden and Stalnaker 1975, Joseph et al. 1977, Wick et al. 1982, Minckley et al. 1991). Competition and predation from non-native fishes that are successfully established in the Colorado River and its reservoirs have also contributed to their decline (Minckley and Deacon 1991).

Furthermore, given declines of suitable riverine habitat, increased predation and competition in current suitable and limited habitats may occur (Mueller and Marsh 2002). For further detailed information including examples, ramifications, and research needs pertaining of the effects of habitat modifications on native Colorado River fishes, please see Tyus et al. (1982), Minckley and Deacon (1991), Chart and Bergersen (1992), and Mueller and Marsh (2002).

Systematics and Morphomethrics

The flannelmouth sucker belongs to the family Catostomidae. Likely the most conspicuous feature of this group is the distinct, fleshy lips ventrally located on the snout. Mouthparts of most catostomids are enlarged, protrusible, and plicate or covered with papillae, presumably adapted for benthic feeding strategies (Miller and Evans 1965). The genus Catostomus contains a large number of species that includes several species from the Rocky Mountains, most of which are native and isolated to tributaries of the Western United States (Eddy and Underhill 1969). Flannelmouth sucker are distinguished from other species of the genus Catostomus by the thickened lower lip that is elongated, compared with other species, and completely divided by the median groove. Furthermore, dorsal ray counts are typically 10-11 and lateral line scales are generally more than 80. Scales located immediately above the lateral line tend to be bordered in dark pigment (Eddy and Underhill 1969, Mueller and Marsh 2002). Holden (1973) reports flannelmouth sucker as typically dark brownish-green dorsally, yellowish or orange laterally, and white ventrally. However, in more turbid reaches, flannelmouth sucker captured by Holden (1973) tended to be lighter shades of tan dorsally and silvery to white on the lateral and ventral surfaces. Colorations become more prominent during the spawning season, with tubercles becoming highly evident on male fish (Cross 1975, Muller and Marsh 2002). Adult flannelmouth sucker reach lengths over 18 in (Eddy and Underhill 1969), with some of the largest specimens being captured in the LCRB where lengths can exceed 26 in (Muller and Marsh 2002). Larval stages are best described in Snyder et al. (2004).

Hybridization

As reviewed by Bestgen (1990) and described above for razorback sucker, hybridization between flannelmouth sucker and other native Colorado River catostomid species has historically been documented to occur. Flannelmouth sucker have been shown to hybridize with populations of razorback sucker, but it is foreseeable that populations of flannelmouth sucker would hybridize with Sonora sucker (*Catostomus insignis*) and other native catostomids, given that flannelmouth sucker presently exist in greater abundances in the Colorado River and its tributaries, as compared with other native fishes (Hubbs et al. 1943, Hubbs and Miller 1953, Holden 1973, Holden and Stalnaker 1975, McAda and Wydoski 1980, Minckley 1983, Bozek et al. 1984, Tyus and Karp 1990, Douglas and Marsh 1998). Buth et al. (1987) used allozymic data to directly quantify presumed introgression in the range of their samples as being 0-5% toward flannelmouth sucker and 0-3% toward razorback sucker. Hybridization between flannelmouth sucker and razorback sucker is problematic, given that adults of both species have been documented to simultaneously utilize cobble-bottomed, main-channel, riffles for spawning activities in natural riverine settings (see also razorback sucker profile). A larger concern today is hybridization between flannelmouth sucker and introduced sucker species. Hybridization has been documented between flannelmouth sucker and white sucker in most UCRB streams (Holden 1973, Ryden 2005).

Habitat

Adults

Historically, flannelmouth sucker inhabited virtually all components of riverine habitat ranging from fast current, to riffle, eddy, and stagnant backwater areas. Flannelmouth sucker has been known to be the predominate species comprising native fish catches (Holden 1973, Holden and Stalnaker 1975, McAda 1977, Holden 1999). Minckley (1973) described the flannelmouth sucker as being an inhabitant of the larger, swifter streams and rivers of the Colorado River Basin. Cross (1975) found flannelmouth sucker to occur 64% of the time in runs, 31% in pools, and only occasionally in riffle habitat types in the Virgin River. Furthermore, Cross (1975) documents that while flannelmouth sucker were collected over all types of substrate, ranging from mud-bottomed flats to boulders, the majority of collections were made when sampling sand and rubble-cobble substrates (60 % and 35 %, respectively). Flannelmouth sucker also appear to have affinity for overhead cover, with more than 50% of the collections associated with boulders, overhanging trees, or undercut banks in the relatively small Virgin River (Cross 1975). Flannelmouth sucker were typically found in deeper water (mean 59 cm, +/- 49 cm, in the Virgin River) and water velocities at capture were variable (0-1.0 m/s, mean value of 0.44 m/s) (Cross 1975). More applicable to mainstem Colorado River habitats, Gaufin et al. (1960) found adult flannelmouth sucker in the Green River to be most abundant in slower, deeper sections, similar to those described by Cross (1975). McAda and Wydoski (1980) report that flannelmouth sucker were most often captured at the lower portions of a glide or pool and in the vicinity of a mud-silt bottom. Mueller and Wydowski (2004) report flannelmouth sucker captures typically in water ranging to 2 m deep in the lower Colorado River. Some of the information noted above conflicts, illustrating that flannelmouth sucker utilizes and thrives in a wide variety of habitats. More rigorous studies of fairly long river reaches have shown that flannelmouth sucker populations are larger in reaches with large amounts of cobble-gravel substrates and smaller in areas of predominately sand substrate (Ryden 2005), just the opposite of razorback sucker.

Chart and Bergersen (1992) documented the migrational impacts on flannelmouth sucker as a result of a mainstream impoundment on the White River, both pre- and post-dam construction on the White River, Colorado. Recapture data suggest that the dam blocked the return of adult flannelmouth sucker to their home ranges after dam closure, indicating the likelihood of large, seasonal, migrational movements historically. Furthermore, Vanicek (1970) found flannnelmouth sucker and bluehead sucker (*Catostomus discobolus*) to be the only native species to persist below Flaming Gorge Dam. This finding is likely attributable to the cobble-bottomed, riffle habitats typically created below large impoundments, a known habitat important to flannelmouth sucker (Holden 1999). Radio telemetry work by Beyers et al. (2001) suggests that habitat use by adult flannelmouth sucker does not change with time of day, unlike other native fishes which tend to display unique diel habitat use patterns (see bonytail species profile). Furthermore, radio-telemetered flannelmouth sucker were found in water ranging from 0.5 m to >3.0 m, with the most contacts made in depths of 1.5 m. Cross (1975) mentions that flannelmouth sucker were collected the majority of time in “unmodified” physical habitat (80% of collections) with “occasional” collections in habitats of “poor” water quality. Finally, McAda (1977) suggests that the flannelmouth sucker is not as specific in its habitat selection, compared with other large-bodied native fishes (e.g., razorback sucker, bonytail, humpback chub [*Gila cypha*], and Colorado pikeminnow [*Ptychocheilus lucius*]). However, unlike razorback sucker that continue to persist in

reservoir situations, flannelmouth sucker apparently either do not survive well in reservoirs or avoid large, lentic habitats and, therefore, exhibit at least some degree of an obligatory riverine life history strategy (Mueller and Marsh 2002).

Flannelmouth sucker have been commonly captured in water temperatures ranging from 10 to 35 degrees C (mean of 24 degrees C and conductivities varied from 150 to 2,700 mmhos) (Cross 1975). Deacon et al. (1987) found the final thermal preferendum of flannelmouth sucker to be 25.9 degrees C (+/- 0.5 degrees C) using laboratory techniques, but they suggest that the upper temperature threshold determining habitat usage is highly dependant upon acclimation temperature and that flannelmouth sucker in particular, have one of the highest temperature tolerances displayed by native fishes of the Colorado River. Ward et al. (2002) tested the effects of temperature, fish length, and exercise on the swimming performance of young-of-year flannelmouth sucker. Results suggest that fatigue velocities increased with fish size and water temperature, suggesting that warmer in-river temperatures may be important for flannelmouth sucker recruitment and survival. This hints at the relatively high thermal preferences of flannelmouth sucker and the potential impacts that cold water releases may have on populations of this species. Carter and Hubert (1995) found the upper elevational threshold for flannelmouth sucker inhabitance to be 2,192 m based on studies conducted in southwestern Wyoming.

Spawning

Flannelmouth sucker initiate spawning during May and June in the UCRB, when water temperatures are between 6-12 degrees C (McAda 1977, Weiss et al. 1998). Ripe male fish have been captured through July; however, ripe females are rarely collected after the first few weeks in June (Holden 1973, McAda 1977). In contrast, Mueller and Marsh (2002) report spawning to occur in April and May in the LCRB. Historically, flannelmouth sucker, as well as other native catostomids, were reported to make impressive spawning migrations (Simon 1946, La Rivers 1962). More recently, relatively little migrational movement has been observed, presumably due to the impacts associated with mainstem impoundments (Holden 1973, McAda 1977). McAda (1977) captured ripe male and female razorback sucker over cobble and gravel bars in the Yampa and Colorado rivers during spawning season. This is corroborated by research on the San Juan River, as adults have been found spawning over in-channel, cobble-bottomed riffles (Holden 1999). Most recently in the LCRB, spawning activities of aggregations of >200 flannelmouth sucker were visually observed and sampled below Davis Dam to Lake Havasu. Spawning fish were found congregating in water approximately 1-2 m deep, over large cobble and gravel substrates, and in relatively swift currents (0.5-1.0 m/s) (Mueller and Wydoski 2004). To the best of our knowledge, flannelmouth sucker do not spawn in reservoir or other still-water situations, and reproduction is largely limited to riverine settings, particularly main-channel riffles.

Larvae

As mentioned for razorback suckers, dispersal of larval flannelmouth sucker is also important and heavily influenced by flow regimes (Robinson et al. 1998). However, unlike razorback sucker larvae, young flannelmouth sucker apparently do not exhibit diel drift periodicity. They do appear to be associated with near-shore (defined as shoreline habitats with velocities <0.2 m/s), slackwater-rearing habitat types, indicating at least some degree of an active affinity for drift habits (Robinson et al. 1998). Larval flannelmouth suckers are found in shoreline backwaters, embayments, and other low-velocity habitats in the San Juan River using larval seines (Brandenburg et al. 2005).

Juveniles

Juvenile flannelmouth sucker move out of backwaters and embayments two or three months after hatching. This is generally indicated by their sudden disappearance from seine collections, which concentrate on these low-velocity habitats. Juveniles tend to move into habitats with more velocity such as runs and edges of riffles (Holden 1999). This is very similar to habitat shifts that occur with roundtail (*Gila robusta*) and humpback chub (Holden 1977). This habitat shift appears to coincide with a decline in numbers, but this may be an artifact of poor sampling of these habitats. Lastly, Gido et al. (1997) suggest that secondary channels are important to young flannelmouth sucker in the San Juan River, but they do not compare use of those habitats with use of similar habitats in the main channel. Therefore, whether the species uses secondary channels or the habitats found in secondary channels is unclear.

Reproduction

Male and female flannelmouth sucker mature between 4-6 years of age in the UCRB or at lengths ranging from 391-421 mm (McAda 1977, McAda and Wydoski 1985). Fecundity of the flannelmouth sucker is highly dependant upon size, and is therefore highly variable within age groups (see age and growth section for reasoning). Female flannelmouth sucker produce anywhere from 3,000 to 30,000 eggs, depending upon age and size. Furthermore, mature individuals can reproduce for more than 20 years, demonstrating remarkable reproductive abilities that help establish and maintain flannelmouth sucker populations in areas where other native species have declined and continue to decline (Mueller and Wydoski 2004). Even within year classes, numbers of individual eggs produced can differ by 20,000 eggs in some cases (McAda 1977). Sex ratios of male:female fish have been reported as 3:1 (McKinney et al. 1999) and 1:1-2:1 depending upon capture location and time of year (Weiss et al. 1998). Egg diameters range from 2.5-3.8 mm, and eggs are typically deposited in distinct areas. After depositing eggs females typically move on to new spawning areas, while males remain and await the arrival of another female (Weiss et al. 1998).

Diet

Flannelmouth sucker diet composition has been described as being omnivorous, with a wide variety of consumption ranging from algae to detritus, including mud and invertebrates (Ellis 1914, Sigler and Miller 1963, Taba et al. 1965, Minckley 1973, Cross 1975). More specifically, Cross (1975) described specimens from the Virgin River as consuming aquatic insect larvae (*Diptera*, *Ephemeroptera*, *Tricoptera*, and even amphipods), with portions of inorganic and organic material including quantities of filamentous algae. These findings are corroborated by efforts in the UCRB, as larval and early juvenile flannelmouth sucker were discovered to consume chironomids in greatest abundance, with evidence of organic and inorganic matter, diatoms, and unidentified plant seeds identified in the stomach contents.

Bartschi (1964) (as described by McAda [1977]), documented seasonal and size-specific shifts in flannelmouth sucker food habits. For example, flannelmouth sucker <80 mm fed exclusively on copepods, while fish >200 mm did not consume copepods. As flannelmouth sucker increases in size, it appears that *Ephemeroptera* instars become an even more important food item, while other aquatic invertebrates (e.g., *Diptera*) are eaten by all size classes of flannelmouth sucker, particularly in the later spring and summer months. Furthermore, ephemeroptera nymphs increase in importance

during late summer into fall. Overall, the food habits of the flannelmouth sucker are highly dependant upon the availability of food items, with more common items becoming more predominant in diet composition at any given time. This strategy is typical of an omnivorous, opportunistic riverine obligate species (McAda 1977).

Age and Growth

Published growth estimates for flannelmouth sucker vary, and available information is highly dependant upon age and habitat occupied (for details see age and growth tables compiled by McAda [1977]). McAda (1977) used scales as a method for age determination and for back calculation of growth of UCRB flannelmouth sucker. Annuli formation was found to occur in June and July, with scales forming on young flannelmouth sucker as small as 25 mm. Scale development was complete throughout the surface of young fish by 40 mm. Total lengths of older fish were found to be variable, and considerable overlap in size was noted between age groups. Overall, age I fish were found to be between 57-80 mm, age II between 128-167 mm, age III between 245-286 mm, age IV between 351-370 mm, age V between 412-432 mm, age VI between 440-475 mm, age VII between 446-493, and age VIII between 456-514 mm (McAda 1977). McKinney et al. (1999) report growth for flannelmouth sucker collected below Lee's Ferry, with fish displaying nearly isometric growth. Adults typically grew 5.5 mm/year (standard deviation 1.9 mm) and subadults 45.9 mm/year (standard deviation 16.8) based on recaptured, tagged individuals. More recently, Mueller and Wydoski (2004) found that larger aging structures (such as entire fin rays and vertebrae) produced more discernable age information compared with otoliths and smaller fin ray sections. Length distribution data compiled by Mueller and Wydoski (2004) suggests that recruitment of flannelmouth sucker to the spawning cohort documented in the LCRB, ranges between 9.4-31.3 % per year, suggesting the capability of flannelmouth sucker to maintain a population of spawning-aged fish despite extensive riverine habitat modifications.

Bonytail

Distribution

Bonytail was historically widespread and common throughout tributaries of the Colorado River and other larger rivers, with historical captures documented from Mexico to Wyoming (Behnke and Benson 1980, Minckley and Deacon 1991, Mueller and Marsh 2002). The first recorded capture of bonytail from the UCRB was by Jordan (1891) with one specimen collected from the Green River. Subsequent historical collections, albeit limited largely to anecdotal and historical fishing creel interviews, in conjunction with limited scientific collection information combine to demonstrate the once-expansive range of bonytail (USFWS 2002). However, during the 1950s bonytail populations began a rather large, yet poorly documented decline in abundance following numerous biotic and abiotic habitat modifications (see below, and as described in the razorback sucker and flannelmouth sucker species profiles). Holden (1991) described the effects of a large-scale rotenone treatment in the upper Green River, while simultaneously providing insight to the rather large population of bonytail present until 1962, at which time a large piscicide treatment occurred in the UCRB. Bonytail numbers were drastically reduced following the closure of Flaming Gorge Dam in 1963, with very few and sporadic captures of bonytail occurring in the UCRB since that time (Vanicek and Kramer 1969, Holden and Stalnaker 1975, Tyus et al. 1982, Valdez 1990).

Bonytail captures in the LCRB follow similar trends. The USFWS (2002) documents an early capture of 16 individuals from the LCRB by R.R. Miller (from the Grand Canyon). Jonez and Sumner (1954) documented a large aggregation of an estimated 500 adults spawning over a gravely shelf in Lake Mohave. During the period between 1976-1988, 34 bonytail were captured in Lake Mohave, and some of these fish were incorporated in the establishment of a broodstock, the progeny of which are presently stocked into Lakes Mohave and Havasu (Minckley et al. 1989, Minckley et al. 1991, USFWS 2002) and a number of UCRB rivers. Very few documentations of wild bonytail capture have been recorded in recent years and, therefore, little is known about the specific habitat requirements of this unique species.

Historical Habitat Modifications

Numerous researchers have identified that the major factor contributing to the decline of bonytail and other large-river fishes has been the construction of mainstem dams and the resultant cool tailwaters and reservoir habitats that replaced a once warm, riverine environment (Holden and Stalnaker 1975, Minckley et al. 1991, Mueller and Marsh 2002, USFWS 2002). Competition and predation from non-native fishes that are successfully established in the Colorado River and its reservoirs have also contributed to their decline (Minckley and Deacon 1991, USFWS 2002). For further detailed information including examples, ramifications, and research needs pertaining of the effects of habitat modifications on native Colorado River fishes, please see Tyus (1982), Minckley and Deacon (1991), Mueller and Marsh (2002), and USFWS (2002).

Systematics and Morphomethrics

The following species description is based on information supplied by USFWS (2002):

> Bonytail were first collected from the Zuni River, New Mexico, in 1853, by Baird and Girard during their early expeditions to the Colorado River Basin (Sitgreaves 1853, Girard 1856). Gila elegans is commonly known as the bonytail, a name that has been shared by numerous, and other native chubs of the Colorado River. Bonytail are a streamlined fish, typified by its small head, slender body, and thin, pencil-like caudle peduncle. The head is compressed and the snout overhangs the mouth. Bonytail also have a small, smooth hump (smaller than that of the humpback chub) located directly posterior to the head of adult fish. Bozeck et al. (1984) indicates that bonytail may reach lengths greater than 550 mm, and may weigh over 1100 g. Coloration is typically grey dorsally, fading to white ventrally, with yellowish pigmentation near the base of the pectoral and pelvic fins. Adult spawning fish (males and females) display tuberculation on the head and fins. Dorsal and anal fin rays are typically 10 (Holden 1968, Holden and Stalnaker 1970, Rinne 1976) with caudle peduncle length divided by head length equaling 1.0 (or head length divided by caudle peduncle depth usually being 5.0 or more) (Minckley 1973). Bonytails are mostly scaled throughout the body surface, with 75-88 scales along the lateral line. Scales are not as deeply embedded as those of the humpback chub and the pharyngeal teeth formula is 2,5-4,2. As described by Holden (1968), young bonytail are easily confused with roundtail chubs and humpback chubs, particularly at smaller size classes and in areas of known coexistence. As adults, bonytail are often

mistaken due to what appears to be a high level of morphological plasticity among the endemic species of the Colorado River Gila complex, and due to understudied levels of introgressive hybridization of the various species of Colorado River Gila (Dowling and DeMarais 1993, Douglas et al. 1998). The unique morphology of the bonytail has been hypothesized to be adapted to historical, torrential flows, thought to have been typical of the Colorado River (Miller 1946, Beckman 1963).

Hybridization

As reviewed by USFWS (2002), hybridization between bonytail and other native Colorado River Gila species appears to have been common. For example, within the Gila complex, inter- and intraspecific morphological variation is apparently extensive where bonytail, roundtail, and humpback chub occur sympatrically. The result of this apparently high degree of hybridization is a relatively high level of phenotypic plasticity, with multiple authors reporting multiple morphologic intergrades present in samples collected throughout the Colorado River (Holden 1968, Holden and Stalnaker 1970, Smith et al. 1979, Douglas et al. 1989, Kaeding et al. 1990, Douglas et al. 1998). Such genetic intermixing was likely common historically and plausibly served to promote phenotypic plasticity and adaptability of the various species to their environment (Dowling and DeMarais 1993). Furthermore, Miller (1946) suggests evidence of species intergrades prior to anthropogenic influences. Recent mitochondrial- and allozyme-based DNA research efforts suggest that bonytail are a uniquely adapted extension of the roundtail chub complex (Dowling and Demarais 1993). The extent of current and on-going hybridization and its impacts on wild bonytail populations are unknown due to the absence of recent captures, but hybridization and its effects may become important as populations become established through hatchery introductions and overall species recovery, particularly as increasing populations of Gila become potentially and increasingly intermixed due to compressed habitat availability (USFWS 2002).

Habitat

Adults

As stated previously, information pertaining to bonytail habitat preferences is very limited due to the extirpation of this species prior to extensive sampling of the Colorado River and its fishery. Limited, early fisheries surveys indicate that the bonytail tended to be found in higher-gradient, gravelly riverine sections, with some degree of habitat use similarities as described for the flannelmouth sucker. For example, bonytail is widely characterized as being adapted to the swifter sections of the Colorado River, with affinity for areas of high flow and rocky habitat. Available information suggests that adult bonytail used fast-water sections, as well as eddies and pool habitats. Vanicek (1967) noted habitat selection of bonytail to coincide with habitats occupied by another native chub, the roundtail chub. Vanicek (1967) found these species in pools and eddies, typically near “fast-flowing” riverine areas but also in slower sections. Holden (1991), citing Flaming Gorge preimpoundment surveys, noted that bonytail were apparently fairly common in the Flaming Gorge area of the upper Green River, a canyon-bound, relatively fast water section of river. Valdez (1990) reported bonytail habitat use as being similar to that of humpback chub, with collections being made in shoreline eddy habitats, boulders and cobble, and near swift-water sections (in Cataract and Desolation Canyons).

Telemetry studies by Mueller et al. (2003) revealed that adult bonytail prefer interstitial spaces associated with shoreline riprap during daylight hours in Cibola High Levee Pond, whereas open-water areas are more commonly utilized during the nighttime hours. Intensive telemetric surveillance suggests a high degree of site-specific habitat fidelity, with individually marked bonytail consistently returning to the same cavities formed within the riprap type shoreline. These areas may simulate the boulder fields of many of the UCRB canyon areas where bonytail were once common.

Interestingly, a study conducted by Pimentel and Bulkley (1983) suggests that bonytail, when given the opportunity, tend to select water with high levels of total dissolved solids (TDS). Bonytail are able to persist in water with TDS of 4,700 mg/L, the highest tolerance reported for any species of Colorado River Gila, suggesting an ability to persist despite anthropogenic water quality and habitat degradation.

Spawning

Bonytail have been documented to spawn over gravel substrates, near shore, and were found in water up to 30 ft deep in reservoir situations (Jonez and Sumner 1954). They are hypothesized to use gravel-cobble habitats in lotic environments. Most recently in the LCRB, documentation of successful, natural reproduction in Cibola High Levee Pond suggests that bonytail select shoreline-associated, riprap materials (large-diameter gravel, cobble, and boulder substrates) in water 2-3 m deep for spawning activities (Mueller et al. 2003). Spawning individuals in Lake Mohave display similar diel habitat shifts: adults use in deeper habitats during the day and later form congregations along shoreline habitats (Mueller and Marsh 2002).

Larvae and Juveniles

Relatively little is known about habitat needs for young bonytail. Similar to other native fishes, backwaters and other slackwater habitat types are thought to serve as important nursery areas for young bonytail (USFWS 2002). Larval roundtail and humpback chub tend to use low-velocity backwaters, embayments, and other small, low-velocity habitats along shorelines, moving to water with more current as they become larger (50-75 mm) (Holden 1977, Valdez 1990, Valdez and Ryel 1997). Whether bonytail exhibit the same habitat shift is not known, but it is very likely that the primary reason for the loss of bonytail throughout the basin is related to loss of important nursery habitat. Relatively narrow nursery habitat requirements separate razorback sucker and Colorado pikeminnow from the non-endangered, more common species such as flannelmouth sucker and roundtail chub. Therefore, it seems likely that in a riverine situation bonytail may have a nursery habitat requirement that has not as yet been fully explained.

Young bonytail were most commonly associated with areas of dense overhead cover in depths greater than 1 m. They displayed schooling in warm, shallow areas of Cibola High Levee Pond (Mueller et al. 2003). These findings suggest that refugia-type backwaters designed for bonytail should have similar components in terms of riprapped shoreline materials, one of the few specific habitat preferences that has been documented to date.

Reproduction

Vanicek and Kramer (1969) documented the last substantial spawning of a wild, riverine population of bonytail in Dinosaur National Monument. Ripe fish were collected from mid June through early July in water temperatures around 18 degrees C. Bonytail estimated between 5-7 years old were

found ripe (Vanicek 1967), whereas in controlled hatchery environments, Hamman (1985) found bonytail to begin maturing sexually at age 2. Johnston (1999) classified bonytail as being broadcast spawners and suggested that loss of eddy habitat types due to the construction of impoundments may contribute to the apparent reproductive failure of a closely related species, the humpback chub. Marsh (1985) reported bonytail eggs to be adhesive and apparently remain so throughout the incubational period, which is thought to be an adaptive strategy to swift-moving currents of the mainstem Colorado River.

Ripe bonytail have also been collected from lentic, reservoir situations. As stated previously, Jonez and Sumner (1954) reported active spawning of a large (approximately 500 individuals) aggregate of bonytail in Lake Mohave. Eggs were described as being adhesive, and one individual female contained over 10,000 eggs, suggesting a high level of fecundity, a trait that appears to be typical for native, Colorado River endemics (see razorback sucker and flannelmouth sucker species profiles). Even higher levels of fecundity were found in hatchery settings, with individual female egg production averaging over 25,000 eggs per female (Hamman 1982). Spawning bonytail in Cibola High Levee Pond were observed utilizing shoreline riprap materials, typically in mid April and frequently during nighttime hours, in water temperatures ranging from 20.4-21.6 degrees C. They were observed consuming their own gametes, as well as young razorback sucker larvae (Mueller et al. 2003).

Bonytail egg survival appears to be highly influenced by incubation temperature. Hamman (1982) found 90% survival at water temperatures of 20-21 degrees C, 55% survival at 16-17 degrees C, and only 4% survival when temperatures were held between 12-13 degrees C. Incubation periods ranged from 99 hours to nearly 500 hours, depending upon water temperatures. Newly hatched fry averaged 6.8 mm (Hamman 1982). This research is corroborated by Marsh (1985), who found bonytail embryos to have the highest survival rates at temperatures near 20 degrees C and indicated that newly hatched larvae averaged 6.0-6.3 mm in size.

Diet

Bonytail diet is reportedly comprised of a wide variety of aquatic and terrestrial insects, worms, algae, plankton, and plant debris (Mueller and Marsh 2002). This information is corroborated by McDonald and Dotson (1960) and Vanicek (1967) who also found Colorado River chub to feed omnivorously. More detailed and quantitative descriptions of bonytail diet preferences are not available, including shifts in diet composition by life stage, with the exception of information from bonytail stocked into Cibola High Levee Pond. This experimental population also fed omnivorously, with adult bonytail consuming algae, vegetative material, small fish, and crayfish (*Procambarus* and *Orcopectes* sp.). Young bonytail were documented to feed near the surface of the pond, with gut analysis demonstrating that smaller size classes typically fed on zooplankton and invertebrates (Mueller et al. 2003). Again, more detailed knowledge is unavailable, likely due to the overall rarity of the species.

Age and Growth

Little detailed information exists pertaining to naturally recruited bonytail age and growth patterns. Following information reviewed by the USFWS (2002), the only substantial findings regarding

bonytail age and growth are those reported by Vanicek (1967). This author aged 67 bonytail using scales and found the largest to be 7 years old at a length of 338 mm and weighing 422 grams (g). Ulmer (1983) used otoliths to determine that two Lake Mohave bonytail were 32 and 39 years old. This finding was corroborated by Rinne (1986) who estimated four Lake Mohave fish to be between 34-49 years old. Available data suggest that bonytail are typically captured between 338-535 mm TL (USFWS 2002). In any case, bonytail are long lived, demonstrating yet another trait that has been speculated by many researchers to be an adaptation to an extremely harsh, unpredictable environment (Mueller and Marsh 2002).

Other Aspects of Native Fishes in General

Disease

The effect of disease vectors on the various life stages of native fishes remains relatively unknown and understudied. However, information pertaining to diseases present in catostomids native to the Colorado River drainage likely provides insight to potential vectors of disease that may impact highly jeopardized populations. For example, Minckley (1983) provides information on injuries and diseases report for razorback sucker collected from Lake Mohave. He reported that a large number of individuals were blind in one or both eyes. This problem was attributed to bacterial and protozoan infections, as well as age. Minckley (1983) further describes other common infestations, such as those of the parasitic copepod Lernaea cyprinacea, and less common osteo-deformative problems associated with nematodes, cestodes, and trematodes. Infections were also reported as being most common in females captured during the spawning season. Other researchers have noted similar afflictions, but none has attributed disease as a hindrance to native fish recovery efforts (Flagg 1982). Disease-related information specific to bonytail is limited to the documentation of Lernaea on individuals collected by Vanicek (1967). Furthermore, Mueller et al. (2003) noted an 18% intestinal tapeworm infestation rate in an experimental bonytail population in the LCRB. Other researchers have noted similar afflictions in other native species, but none has attributed disease as a hindrance to native fish recovery efforts (Flagg 1982). Cross (1975) describes infestations of Lernaea sp. as being the most common parasite observed for bluehead sucker in the Virgin River.

Biotic Interactions: Predation and Competition

Historically, predatory impacts to native fishes were likely restricted to predation by other native fishes including: roundtail chub, bonytail, and Colorado pikeminnow. Colorado pikeminnow likely represented the largest historic predatory impacts to young, native Colorado River fishes (Bestgen 1990, Holden 1999). Likewise, it seems plausible for at least some degree of cannibalism to occur, particularly in bonytail populations, given the opportunistic feeding modes of this unique fish. More recently, nearly 70 non-native fish species have been introduced into the Colorado River during the last century (Minckley 1982, Tyus et al. 1982, Minckley and Deacon 1991, USFWS 1998, USFWS 2002). Many of these newly introduced species are large, piscivorous, adaptable fish desired by anglers but thought to be deterrents to native fish propagation. For example, native fish habitat restoration and other recovery efforts have been hampered, particularly when viewed in light of the drastic physical habitat modifications that have occurred during the past century on the Colorado River and coupled with the predatory impacts of introduced, non-native fishes infiltrating newly created/restored habitats designated for native fishes (e.g., Minckley 1983, Taylor et al. 1984, Tyus

1990, Minckley and Deacon 1991, Minckley et al. 1991, Johnson et al. 1993, Mueller and Marsh 2002).

For example, Mueller and Marsh (2002) report that when bonytail are stocked into ponds by themselves, they do extremely well, with multitudes of young being produced. However, if stocked in the presence of other species, young native fishes (e.g., bonytail) quickly fall prey to non-native predators. Furthermore, Mueller et al. (2003) documents that small predators (crayfish and tadpoles) can decimate native fish at both the egg and fry stages, with nearly 100% of razorback sucker eggs and fry being consumed by tadpoles and crayfish in laboratory experiments. Similarly, the largest known population of razorback sucker from Lake Mohave now shows virtually no wild recruitment, a condition likely attributable to non-native predators (Minckley et al. 1991). Marsh and Langhorst (1988) report that larval razorback sucker in Lake Mohave survived and grew better in the absence of predators. Several researchers have documented the apparent lack of escape and defense mechanisms displayed by young razorback sucker subject to predation (Loudermilk 1985, Johnson et al. 1993). Wesp and Gibb (2003) tested the hypothesis that larval razorback sucker had poor escape performance compared with non-native, introduced rainbow trout (*Oncorhyncus mykiss*). These researchers found that razorback sucker had similar (in some cases razorback sucker larvae showed greater performance) swimming velocities and acceleration rates. It was concluded that, while razorback sucker appear to perform as well (if not better) than rainbow trout in terms of swimming performance at temperatures of 12 and 18 degrees C, razorback sucker larvae did not grow as quickly, thereby increasing the duration of time under which young razorback sucker were vulnerable to predation. This phenomenon is likely to influence recruitment success and may also be applicable to populations of bonytail. In contrast, Johnson et al. (1993) show that larval razorback sucker recognize and avoid predators with less response than northern hog suckers (*Hypentelium nigricans*) of the same age, suggesting that differences in predatory avoidance in fishes can be linked to the historical fish assemblage and ecological settings that a fish has evolved in. Razorback sucker apparently evolved in a predator-poor community, while it is suggested that northern hog sucker evolved in a predator-rich environment, with both fish species experiencing differences in evolutionary pressures brought on by unique biological and physical community structures. Such information specific to flannelmouth sucker and bonytail is unfortunately not currently available, but ecological and evolutionary processes similar to those that sculpted the susceptibility of razorback sucker young to predatory impacts most likely helped to mold predator-avoidance mechanisms of the other endemic Colorado River fish species.

Potentially more applicable to bonytail, but to the two sucker species as well, is research on humpback chub that has identified that brown trout (*Salmo trutta*), channel catfish *(Ictalurus punctatus*), black bullhead (*Ameiurus melas*), and rainbow trout as the principal predators of wild humpback chub populations in the Grand Canyon. It has been reasoned that in some cases, entire year classes of humpback chub may be decimated by non-native predatory impacts (Marsh and Douglas 1997, Valdez and Ryel 1997, USFWS 2002). It has also been suggested that red shiner (*Cyprinella lutrensis*), a widespread, non-native species that often is captured occupying nursery habitats used by young native fishes may also be an important predatory threat during early and susceptible life stages. This is particularly important given that young of all three native species tend to inhabit backwater and shoreline habitat types in riverine settings (Ruppert et al. 1993, Holden 1999).

The effects of competition and predation between native and non-native fish species have proven difficult to separate and identify. Predation is typically easier to quantify (i.e., gut content analysis), while interspecific competitive interactions remain largely understudied. For example, small bodied non-natives such as red shiner and fathead minnow (*Pimephales promelas*) display aggressive behaviors towards native catostomids (Karp and Tyus 1990, Sabo et al. 1996). This also appears to be the case for young bonytail (Marsh and Mueller 2002, Mueller et al. 2003). Furthermore, competition for food may also be an important factor determining native fish recruitment (Papoulias and Minckley 1990, Mueller et al. 2003). However, Marsh and Langhorst (1988) suggest that food is unlikely a limiting factor and indicate that predation by introduced fish species appears to be a significant cause of larval mortality in Lake Mohave. In any case, it appears as though complex habitats provide an overall dampening effect on non-native fish predation and competition. This has been hypothesized in Lake Mead where vegetative cover and turbidity are correlated with areas where razorback sucker have recruited in the face of heavy predator loads (Holden et al. 1998).

As demonstrated by Mueller and Wydoski (2004), flannelmouth sucker, while highly understudied, appear to be able to cope with the effects of habitat alterations, including an apparent ability to persist in the presence of competitive and predatory pressures exerted by non-native species. This ability likely stems from the species rather obligatory riverine habitat use and apparently proportionately small use of predator-infested backwater habitats (Holden 1999, Ryden 2000, Mueller and Wydoski 2004). Perhaps through continued support of flannelmouth sucker research, applicable information pertaining to the unique species abilities of this unique fish may be used to retrospectively bolster populations of other native Colorado River fishes, including the potential application of pertinent findings to populations of razorback sucker and bonytail, particularly if more natural, ephemeral, and in-channel slackwater habitats are constructed in future years.

Finally, as with the majority of ecological studies, the effects of predation and competition, coupled with dramatic physical and environmental changes, are most likely interconnected and should not be treated as mutually exclusive. While direct effects (such as predation) are evident, indirect and less-visible impacts (such as inter- and intraspecific competition) may have substantial ramifications concerning habitat developments and restoration/recovery efforts. Future research efforts will likely allow for further conclusions to be drawn concerning complex interactions between native fish recruitment and predation, competition, and on-going habitat modifications. This is particularly true with bonytail research, given that available information is very limited and the applicability of general ecological truths needs to be further tested in order to apply correct management actions conducive to promoting survival and persistence of all life stages of bonytail and native fishes in general. Research in the UCRB has identified methodologies allowing managers to give native fishes opportunity to exist despite non-native fish presence in key backwater habitats by manipulating flow regimes and conducting annual drainings of nursery habitats (e.g., Modde 2005). Furthermore, opportunity may exist to create/modify in-channel habitats (in combination with manipulating flow regimes) that enable highly adapted native fish assemblages to express advantageous, evolutionary-honed traits, thereby promoting a potential shift in competitive advantage towards these unique, native species (e.g., Holden 1999).

Conservation and Management

Effects of Human Activity on Habitat

Minckley and Deacon (1991) describe the native fish fauna of the Colorado River Basin as evolving in a unique mix of riverine, floodplain, and lacustrine habitats. Historically, the Colorado River was known for its extreme variability in flow regimes and associated hydrogeographical components (Minckley et al. 1991). Bestgen (1990) attributes the decline of razorback sucker to many specific causes, with a detailed account of more generalized anthropogenic habitat and ecological-based alterations. For example, he indicates that habitat alterations may have restricted large, free-ranging native fish populations to small subsets of populations occupying small amounts of existing habitat. Dam and reservoir construction likely enhanced populations of non-native species and prevented immigration of native fishes to historic areas, while simultaneously changing temperature and flow regimes. Dams have also acted as buffering agents, reducing the inherent variation in the aforementioned attributes and contributing to a more stable aquatic environment. These conditions are conducive to non-native fauna but are likely beyond the evolutionary scope of the highly adapted native community.

Furthermore, it is likely changes in the chemical nature of available habitat have occurred in association with the human population increase. Increased levels of contaminants such as selenium, organochlorines, polycyclic aromatic hydrocarbons, and phenols have been documented and are regarded to be increasing due to heightened irrigation and diversion activities (e.g., Holden 1980, Tyus 1990, Hamilton and Waddell 1992, Buhl 1997, Hamilton et al. 2000). For more detailed information pertaining to selenium, its effects, and other toxicants of concern, please see Marr and Velasco (2005).

In summary, native Colorado River fishes have been expelled from approximately three-fourths of their original range (USFWS 1998), presumably due to physical habitat modifications coupled with the impacts of introduced fishes such as predation, competition, hybridization, and unintentional introductions of disease.

While all of the above-mentioned factors have affected and likely continue to affect populations of native fishes in the Colorado River in unknown and undesirable ways, one of the most interesting findings is that sizable numbers of virtually all life stages of flannelmouth sucker continue to be found, suggesting interesting and unique abilities of the species to persist despite physical and biological environmental change. This unique ability is best described by Mueller and Wydoski (2004), and demonstrates that, at least for flannelmouth sucker (although the concepts may apply to other native fishes), certain suites of characteristics can combine to allow species' persistence, particularly within the main river channel, an interesting finding especially when applied to razorback and bonytail. As discussed above, LCRB razorback sucker and bonytail are largely considered to utilize various backwater and marsh-like habitats. However, in the more natural UCRB system, these species have been shown to utilize the entire extent of riverine habitats, depending upon time of year and life stage. This suggests the possibility that certain in-channel alterations geared towards bolstering flannelmouth sucker habitat in the LCRB may in fact simultaneously increase habitat for razorback sucker (in particular) and perhaps even bonytail populations as they are introduced in greater numbers.

Degradation of Backwater Habitat

Various authors describe how a reduction in channel width could increase the average velocity in the main channel, resulting in a decrease in the number of low-velocity habitat types (e.g., Wick 1982; USFWS 1998, 2002; Holden 1999). Siltation and subsequent vegetative growth over time may also lead to successional backwater habitat reductions, as well as impact water quality in backwaters (Wick 1982, USFWS 1998). Furthermore, it is possible that dams (in addition to direct reductions in possible movement patterns) can impede sediment transport, resulting in an overall lowering of the main channel and the resultant desiccation of wetlands and other off-channel lowlands through decreased groundwater connections and lack of connection to the main channel (USFWS 1998). Flow regulation can also influence water quality, temperature, and chemical characteristics of an associated regulated river, resulting in substantial potential impacts to spawning cues and hatching success, and contributing to an alteration of competitive and predatory interactions.

As described above, some researchers have utilized flow, connectivity, and annual draining techniques (conversely flushing would likely also be of use) as management tools for coping with non-native species and their impacts. Razorback sucker survival through the initial year of life was vastly improved by these annual resettings of important physical and biological characteristics of pond-like backwaters in the UCRB (e.g., Modde 2005). It has been estimated that the LCRB alone has at least 15 impoundments that may contribute to, and influence life cycles of razorback sucker through, direct and indirect impacts on the physical and biological environments (USFWS 1998).

With Reclamation expending substantial effort to create, maintain, and promote razorback sucker recovery in managed backwater habitats (which are typically infiltrated both naturally, accidentally, and intentionally with non-native species), it seems both logical and useful to study concepts such as periodic draining (or flushing) associated with reset-type backwaters (Tyus and Karp 1989, Bestgen 1990, Bradford and Gurtin 2000, Osmundson 2001, Modde 2005), and research the importance of and issues concerning connectivity and in-channel slackwater habitat types (backwaters for native fish species, consisting of main-channel improvements). These concepts were derived from: Holden 1973, Holden and Stalnaker 1975, Behnke and Benson 1980, Minckley 1983, Mueller et al. 2003, Wydoski 2005). As demonstrated by Mueller and Wydoski (2004), flannelmouth sucker appear to do well in highly altered riverine environments despite the presence of non-native predation and habitat alteration, which provides an interesting opportunity to study this unique species and apply subsequent findings to the other species covered by MSCP goals.

Furthermore, as flannelmouth sucker have historically been collected in a wide array of habitat types, with particularly high densities of spawning- and non-spawning-aged fish collected in relatively slow-moving, in-river slackwaters, secondary channels, eddies, and complex shoreline habitat types within main river channels (Gaufin et al. 1960, Holden 1973, Cross 1975, Holden and Stalnaker 1975, McAda 1977, Gido et al. 1997) it seems logical that future flannelmouth sucker habitat restoration efforts would likely be best spent improving and constructing these in-channel features. Indirect evidence suggests that, while adult flannelmouth sucker apparently persist in tailwater-like habitats (at least during the spawning period) and appear to do well below large impoundments (Chart and Bergersen 1992, Weiss et al. 1998, McKinney et al. 1999, Mueller and Wydoski 2004), early life stages are typically collected from smaller streams or in slow-moving slackwater habitats of larger rivers (McAda and Wydoski 1985, Douglas and Marsh 1998).

Providing tailwater spawning habitats for adult fish is unlikely to become a problem (more than 15 impoundments exist in the LCRB). However, active provision of main-channel, slackwater habitats, as well as other off-channel characteristics that function similarly to smaller streams or other off channel slackwater features, remains relatively uncommon in the LCRB. Also, as demonstrated by Ryden (2000), in-channel habitats in the more functional areas of the UCRB are utilized and selected by razorback sucker during much of the year. Alteration and creation of in-channel habitat types would, therefore, not only benefit flannelmouth sucker populations, but also riverine populations of razorback sucker (and potentially bonytail) in the LCRB. Historical records indicate these species have similar in-channel habitat preferences, in which case habitat designed (by accomplishing MSCP habitat creation/restoration goals) to bolster the two sucker populations would likely benefit bonytail as well.

Habitat Criterion and Research Needs for Ranking the Utility and Future Lower Colorado River Basin Backwaters

Minckley et al. (1991) suggest that future efforts directed at benefitting razorback suckers would best be spent by improving the survival of reintroduced fish and increasing the exchange of information regarding razorback sucker management. This suggestion has largely been followed by manipulating the physical habitat available to razorback sucker for recovery efforts in the LCRB. This profile (and future project-related efforts) will continue this effort, placing specific emphasis on the importance of backwaters and their ecological underpinnings related to their construction and monitoring, as well as attempting to define backwater-specific characteristics important to native fish species recovery. Information presented herein provides an initial outline of known habitat requirements and species-specific information, identifies successful approaches used throughout the UCRB and LCRB to bolster native fish populations, and guides future model criteria development specifically tailored to ranking backwaters according to habitat preferences of native fishes, which is a useful approach for ranking and prioritizing management actions directed at restoring and maintaining crucial habitats.

Through the compilation of this report, it was apparent that various items were understudied or absent from the literature. Issues pertaining to native fish recruitment and survival through the juvenile life-history stage were particularly understudied. Many authors attributed this lack of knowledge to the scarcity of capture data available for this particular life stage. Specifically, few authors documented observation of wild juvenile razorback sucker, while fewer still related hypotheses or data for juvenile occurrence in particular areas (i.e., why razorback sucker recruited in that location), while in other areas young fish were absent. This lack of information was most evident for flannelmouth sucker and bonytail.

Another essential research effort that should be completed prior to the construction of ranking criteria for future model development is the compilation of native fish stocking information (razorback sucker in particular) in LCRB backwaters. A comprehensive, historical, and current record of numbers of fish stocked into specific backwaters would provide valuable insights pertaining to native fish survival and backwaters conducive to this. Specific information regarding the physical habitats of backwaters conducive to fish survival and information describing the physical components of undesirable backwaters should be incorporated in this effort. For example, although various LCRB backwaters are mentioned and identified in various reports (e.g., Holden et al. 1986, Gurtin and Bradford 2000, Slaughter et al. 2002, and others), a single, all-encompassing document describing native fish stockings, successes and failures of stockings, and specific

descriptions of water quality and the physical components (e.g., size, depth, shoreline development, substrate, vegetation) of LCRB backwaters has not been compiled (to the best of our knowledge, at least in published form). Therefore, items that may contribute to the successful survival of native fishes in some backwaters remains unknown. By completing this effort, detailed, specific characteristics (e.g., water quality, depth, substrates, size) that denote good LCRB native fish-rearing facilities could be delineated. If necessary, field efforts could be incorporated to fill in potential information gaps.

Once the LCRB backwater stocking compilation is complete, the applicability of reset backwaters as described by Modde (2005) to the LCRB should be investigated to achieve self-sustaining populations, particularly given that razorback sucker apparently demonstrate strong affinities for backwater habitats throughout all stages of their life history, especially during certain times of the year. This research may be more easily and cost effectively accomplished in certain areas within the LCRB; this may become more evident when the compilation document is complete.

Another research effort could identify areas of the LCRB where main-channel slackwater habitat types may be feasible to construct/maintain. This effort could also establish the relative importance of these in-channel habitat types to various life-history stages in the lower Colorado River, particularly as these habitat types have been and are purportedly crucial to historical and present populations of native fish species (particularly flannelmouth sucker and razorback sucker) in the UCRB and LCRB. Future experimental designs that review current, existing, backwater-nursery habitats and growth/survival through the initial year of life similar to those described by Modde (2005) would likely serve as an initial step towards evaluating the effectiveness of reset backwaters and in-channel features in the LCRB.

Given that (1) flannelmouth sucker appear to utilize main and secondary in-channel habitats more extensively than razorback sucker and display affinity for a more lotic environment overall, and (2) knowledge specific to bonytail is highly lacking (but suggestive of both backwater and in-channel habitat use during the various life stages), it is plausible that backwaters designed to promote razorback recruitment and survival may also become suitable habitat for bonytail, particularly bonytail in early life stages. Likewise, as demonstrated above, bonytail appear to utilize main and secondary in-channel habitats during late life stages (similar to flannelmouth sucker and later life stages and certain annual-seasonal habitat use) but also likely utilize lentic habitats, particularly during early life stages. This is suggested by the species' adaptability to pond and reservoir habitats (e.g. High Levee Pond, Lake Mohave, and Lake Havasu). Hence, there appears to be an opportunity to construct and maintain key habitat types within the main channel (i.e., those conducive to flannelmouth sucker) and provide certain types of backwaters (as described in the two sucker and bonytail profiles) that may be conducive to all three species. For example, since young razorback sucker, flannelmouth sucker, and bonytail apparently have an affinity for slackwater habitat types (during certain life stages and during certain periods of the year), work accomplished in the main channel, efforts spent constructing secondary channels or more seasonally available ephemeral features that function as secondary channels and backwater areas, and research on potentially implementing reset backwaters may benefit all three species and maximize the results of habitat improvement and overall recovery efforts while minimizing cost.

Although some work has been directed towards defining movements and season-specific habitat use of native fishes in the LCRB, further work would be useful. It may elucidate the importance of in-channel slackwater areas to a fuller extent. For example, it remains unknown if razorback sucker spawn in LCRB backwaters consistently, and if so, to what degree hatching and subsequent larval survival rates are indicative of habitat quality. The riverine habitat use of other native species when subjected to present, in-river habitat availability, is also unknown.

Finally, while much information has been collected pertaining to general razorback sucker specific physical affinities, tolerances, and requirements, this information is not complete and readily available for individual life stages. Research directed towards this end, with particular focus on the physical and biological components that delineate "quality" backwaters according to each native fish species (i.e., if a backwater holds a native species, investigate the uniqueness of specific backwater) would be beneficial. This is especially significant since backwaters are widely deemed as important habitats of native Colorado River fishes and are wildly used in razorback sucker hatchery and repatriation efforts. This effort could be combined with the comprehensive review of LCRB backwaters described above, thereby providing a means for a more detailed understanding of what makes certain backwaters in the LCRB desirable to native fishes.

Another area that may require bold actions, and research activities that have not typically been undertaken with endangered and endemic Colorado River fishes, is initiating research pertaining to imprinting and establishing self-sustaining razorback sucker and other native fish populations. As we begin to more fully understand the conduciveness of certain habitat types, and as these areas are identified and mapped, it becomes logistically important for researchers to develop experimental stocking protocols that could potentially increase native species use of specific habitat types. For example, how can we encourage razorback sucker to imprint on areas conducive to recruitment in a natural setting prior to predatory impacts? What about bonytail or flannelmouth sucker? What can be done to promote the return of adult native fishes to these critical habitat areas? Could this be an effective tool in arriving at recruitment and self-sustaining populations? By addressing and focusing research on these rather unknown questions, other recovery methods may be identified. These methods could then be implemented for native fish survival and recruitment.

In addition to the recommendations outlined above, monitoring of existing populations of native fishes needs to continue to further glean information about them (e.g., to gather long-term data vs. the "snapshot" sampling common to ecological studies or continue to gather long-term data on specific populations, such as those that were inhabiting Cibola High Levee Pond, and even create new "Cibolas" if necessary). Long-term data compilation will help elucidate trends in recruitment, growth, habitat use, and survival, and provide an overall, more complete, holistic understanding of population dynamics. Such information will become valuable and can be applied to future establishments of native fish populations.

In summary, potential criteria for ranking the utility of existing and future backwaters for native fish recovery efforts will likely only be developed in one of three ways. First, if the goal is to simply create the normal refugia-type backwaters typical of the LCRB, then it becomes necessary to fully describe existing backwaters that have resulted in both successful and unsuccessful stockings, and to gather and record specific characteristics of these backwaters in both a biological and physically descriptive manner. Once this information is obtained, specific criteria conducive to native fish

survival can be extracted and applied as criteria to develop an index for ranking refugia-type backwaters and their utility to native fishes (in light of, and in conjunction with, information supplied in this report). Second, conversely, and potentially more simple in comparison with the previous option, is to identify refugia type backwaters known to be conducive to harboring native fishes. Future backwaters can be constructed, modeled, and tailored according to the characteristics of the refugia utilized by native fishes. However, specific reasons regarding the importance of these areas will likely remain unclear. Third, and in conjunction with the first option, would be to conduct an evaluation of options for creating/maintaining both in-channel habitats and certain backwaters using the ranking criteria developed under option one. This option would require not only an assessment of off-channel backwaters conducive to native fish survival but also surveys of existing, in-river habitats and their ability to hold native fishes.

These habitat profiles, while useful and informative, by no means have allowed for solid criteria identification. Although specific, certain requirements have been identified for all species (to varying degrees, depending upon species) via literature review and could be extracted in tabular form (e.g., see southwestern willow flycatcher section), it is apparent that fish habitat use, and backwater use more specifically, is very broad and reflective of the overall abilities of the species to adapt and utilize the best, most suitable habitats available at a given time, season, life stage, and region. For example, while items could likely be extracted to establish criteria for ranking existing and future refugia-type backwaters (particularly for razorback sucker) based on documented species needs and current use patterns of presently available habitat (and once compiled through future efforts, see above paragraph), the criteria established would be intrinsically biased to those habitats currently available. Therefore, those criteria might not necessarily reflect the actual preferred and historical habitats utilized by these unique species. An adaptive management plan that not only replicates the currently used refugia-type backwaters, but also includes options for augmenting, creating, and modifying habitats within the main channel would more likely achieve conditions that promote all aspects of the life histories of native fishes. To date, this only occurs in limited sections of the UCRB, and something that will most likely be dictated by the overall direction and goals we choose to follow in the LCRB regarding native fish recovery. While initially this review was geared towards summarizing native fish habitat requirements, it quickly became clear that in order to fully attain the spirit of the MSCP and its associated goals, coupled with information often overlooked from the UCRB, that habitat improvement efforts in the LCRB would likely be biased (and therefore likely limited) to creating habitats that may or may not be truly preferred by native fishes. As such, this review provides a unique perspective of habitat use by native fishes occupying a more natural riverine environment and will assist in developing detailed criteria for ranking and prioritizing riverine habitats as future work is accomplished, overall benefitting native fishes.

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CHAPTER 2: AVIAN SPECIES

Introduction

Marsh communities along the lower Colorado River are generally classified into seven types, which are described by Anderson and Ohmart (1976). Each marsh type is based primarily on the percent cover of cattail (*Typha* spp.), bulrush (*Scirpus* spp.), common reed (*Phragmites australis*), and open water. Historically, marshes along the lower Colorado River were commonly found around oxbow lakes and backwater areas. Today, however, marshes are more frequently found in association with dams, reservoirs, and backwater areas. Marshes of the lower Colorado River frequently consist of cattail, bulrush, giant reed (*Arundo donax*), common reed, saltcedar (*Tamarix* spp.), and willow (*Salix* spp.).

Frequently, marsh communities will interface with woody riparian areas consisting of cottonwood (*Populus* spp.), willow, saltcedar, and mesquite (*Prosopis* spp.). Riparian community structure, as defined by Anderson and Ohmart (1976), is divided into six types (types I through VI) that are based on the density of foliage present in each of three vertical layers. A structural type of I has well developed foliage in all three layers, with the upper canopy dominating.

Marshes and riparian communities are frequently found in association with backwater areas along the lower Colorado River. Backwaters, to some degree, represent the open water elements of the pre-dam Colorado River channel and associated floodplain. Under existing conditions, backwaters include oxbow lakes, abandoned river channel pools, floodplain ponds and lakes, secondary river channel pools, and hydrologically isolated coves on reservoirs. Backwaters may be permanent features or temporary in nature, drying completely during some seasons or years. Some backwaters may be connected or partially connected to the river, while others may be completely disconnected. Generally, backwaters can vary in size, ranging from less than one acre to more than 100 acres. Typically, the interface between backwater areas and marsh communities consists of cattails, bulrush, and common reed. Cattails usually occur in shallow water up to 3 feet deep and are found on stable, sloping substrates. Bulrush is frequently found in association with cattails but in deeper water. Bulrushes can be found in water as deep as 5 feet and can extend as high as 10 feet above the water surface. Thick stands of bulrush usually occur on unmodified banks. Common reed can also grow in dense stands along the backwater/marsh interface.

As required by the MSCP, Reclamation must restore more than 8,000 acres of marsh, riparian, and backwater habitat for 26 covered species. As such, clear and thorough knowledge of habitat requirements must be gained to successfully implement any restoration plan. Of the 26 MSCP covered species, habitat requirements for the California black rail, least bittern, Yuma clapper rail, southwestern willow flycatcher, western yellow-billed cuckoo, and Colorado River cotton rat are discussed in the following species profiles.

Although limited in distribution, the California black rail (*Laterallus jamaicensis coturniculus*) was historically not uncommon (Conway et al. 2002). Most populations, however, are now threatened with extinction, and local populations are thought to be declining because of loss or degradation of suitable habitat (Evens et al. 1991, Eddleman et al. 1994). Currently, the lower Colorado River population of California black rail is thought to be the only stable inland population in the western

United States (Rosenberg et al. 1991). Because of habitat loss and declines in population size, California black rail is listed as endangered in Arizona and considered threatened in California (Eddleman et al. 1994). Because of its apparent decline in population size across the country, California black rail has been added to the National Audubon Society's Watch List, and it was previously listed as a Category 1 candidate species by the USFWS (Ehrlich et al. 1992, Conway et al. 2002). Consequently, California black rail are considered a priority species in the Lower Colorado River Multi-Species Conservation Plan (Conway et al. 2002).

Currently, California black rail is thought to be restricted to two isolated regions: northern California and the lower Colorado River Valley (Conway et al. 2002). The largest California black rail populations documented within the lower Colorado River Valley were found below Imperial Dam and at Mittry Lake (Repking and Ohmart 1977, Evens et al. 1991). Smaller populations have been found in areas that include the Salton Sea, West Pond, and along the All-American Canal (Gould 1975, Rosenberg et al. 1991, Conway et al. 2002).

Generally, California black rail have been shown to prefer dense, emergent vegetation in salt marshes, shallow freshwater marshes, wet meadows, and flooded grasslands (Eddleman et al. 1988, 1994). Along the lower Colorado River, California black rail tends to prefer transition zones between upland and emergent wetland communities (Repking and Ohmart 1977, Eddleman et al. 1994).

As the smallest North American raillid, the California black rail is approximately 10-15 cm long and weighs between 29-35 g. Overall, size and color among sexes are similar. Adult California black rail have darker caps than surrounding plumage, with a rufous-colored nape and a pale gray chin, throat, and breast. The remaining dorsum is relatively black (sometimes washed with brown or chestnut) with scattered white spots. Juveniles are relatively similar to adults but have duller plumage with fewer and smaller white spots. The California black rail is smaller and brighter in color than its eastern counterpart, the eastern blackrail (*Laterallus jamaicensis jamaicensis*) (Eddleman et al. 1994).

Although not considered shy, the least bittern (*Ixobrychus exilis*) is one of the most inconspicuous of all marsh bird species (Weller 1961). Because of the species' secretive nature, current Breeding Bird Survey data are considered insufficient for estimating trends in their population size and distribution (Sauer et al. 2004). Bird watchers from across the United States, however, have reported substantial declines in least bittern populations range wide and suggested that the species may have been extirpated in some areas (Gibbs et al. 1992). Habitat destruction and degradation are thought to be the principal threat to the least bittern range wide (Gibbs et al. 1992).

American Birds, the National Audubon Society's ornithological field journal, began publishing the Blue List in 1971 to provide early warning of reductions in the range or population size of North American avian species. The Blue List and was designed to supplement the Federal Threatened and Endangered Species List (Ehrlich et al. 1992). The least bittern has been Blue Listed by the National Audubon Society since 1979 (Tate 1986, Ehrlich et al. 1992) and was considered a Non-game Bird of Management Concern by the USFWS in 1982 and 1987 (Gibbs et al. 1992). By 1982 least bittern populations appeared stable, although at reduced numbers. By 1986 the species was either completely removed from some areas or in severe decline over a wide portion of its range

(Ehrlich et al. 1992). Current least bittern trend data appear inadequate for conducting reliable trend assessments because of small sample sizes (Sauer et al. 2004), although birdwatchers from across the United States have reported that the species has declined over much of its range and may be extirpated in some areas (Gibbs et al. 1992).

Although least bittern are usually found in a variety of habitats throughout their range, occupied habitat always includes dense emergent vegetation (Hancock and Kushlan 1984). Within the lower Colorado River Valley, the largest concentrations of least bittern are found in cattail or bulrush habitats (Andersen and Ohmart 1984, Rosenberg et al. 1991), which usually correspond to marsh structural types 1, 2, 3, and 5 (Sterling 2005).

Although frequently referred to as a solitary nester (Hancock and Kushlan 1984), the least bittern has been found in surprisingly high densities (Gibbs et al. 1992) where as many as 26 nests per hectare (ha) have been documented (Palmer 1962).

The smallest member of the heron family, the least bittern is approximately 33 cm in length and weighs nearly 80g. Overall, size is similar among sexes; however, plumage is dimorphic. Males have a greenish black crown, back, and tail, while females exhibit a purple-chestnut colored crown and back. Both sexes, however, have white and brown necks, flanks, and underparts with chestnut wings and pale buffy coverts (Gibbs et al. 1992). A rare dark color variant known as the Cory's least bittern exhibits chestnut colored coverts in contrast with the typical plumage (Hancock and Kushlan 1984, Gibbs et al. 1992).

Until relatively recently, the southwestern willow flycatcher (*Empidonax traillii extimus*) was thought to have been extirpated from the lower Colorado River Valley (Rosenberg et al. 1991). In 1995, however, biologists at Havasu National Wildlife Refuge (NWR) observed two fledgling southwestern willow flycatcher, which prompted Reclamation to initiate comprehensive surveys in 1996 (Reclamation 1999).

Since 1996, nesting southwestern willow flycatcher have been observed from the Grand Canyon to the Limitrophe, south of Yuma, Arizona (McKernan 1997, McKernan and Braden 2001, Koronkiewicz et al. 2004, McLeod et al. 2005).

The southwestern willow flycatcher is a widely distributed summer resident of much of the United States and southern Canada (Brown 1988). Currently, four subspecies of southwestern willow flycatcher are recognized in North America; they are distinguished by subtle differences in color, morphology, and breeding range (Phillips 1948, Aldrich 1953, Unitt 1987, Browning 1993). One subspecies breeds east of the Rocky Mountains, *E. t. traillii*. Three breed west of the Rocky Mountains, *E. t. brewsteri*, *E. t. adastus*, and *E. t. extimus* (Unitt 1987). Browning (1993) recognizes a fifth subspecies (*E. t. campestris*) that is said to occur in the central portion of the United States.

The southwestern willow flycatcher was Blue Listed by the National Audubon Society from 1980 to 1982 because of declines nearly range wide. By 1982 populations were stable or improving in at least some parts of the eastern United States. By 1986, however, populations in the northern Great Plains and along the middle Pacific Coast were reported as declining, and population sizes along the southern Pacific Coast were reported as seriously diminished (Ehrlich et al. 1992). In 1992 USFWS

was petitioned to list *E. t. extimus* as an endangered species under the Endangered Species Act. Subsequently, the USFWS published a proposal in 1993 to list the subspecies as endangered with critical habitat (58 FR 10694). The USFWS could have a published final determination of critical habitat by 2005.

Historically, southwestern willow flycatcher was likely an abundant summer resident along the lower Colorado River due to the prevalence of riparian willow thickets (McKernan 1997, McKernan and Braden 2001). However, habitat reduction and degradation have significantly reduced distribution and numbers of this species (USFWS 1997). Generally, the southwestern willow flycatcher is thought of as an obligate riparian subspecies that requires dense willow, thickets (Sedgwick and Knopf 1992), or other available low-growing woody vegetation (Stoleson and Finch 1999) with greater canopy cover (Sedgwick and Knopf 1992) along rivers, streams, marshes, or open water areas (USFWS 1997).

The southwestern willow flycatcher measures approximately 15 cm in length and weighs only about 12 g. Overall, it is roughly the size of a small sparrow. Both sexes look alike. The southwestern willow flycatcher is overall greenish or brownish gray above, with a white throat that contrasts with a pale olive breast. The belly is pale yellow. Two white wing bars are visible, but the eye ring is faint or absent. The upper mandible is dark, and the lower mandible light.

The western yellow-billed cuckoo (*Coccyzus americanus*) is a neotropical migrant that historically occupied deciduous woodlands throughout the United States (Laymon and Halterman 1989, Halterman 2004). In the western United States, the western yellow-billed cuckoo has declined significantly over the last 100 years (Gaines and Laymon 1984, Halterman 2004). The primary reason for its decline in western states is habitat loss, especially where riparian woodlands have been severely disturbed and reduced.

The decline of western yellow-billed cuckoo populations has garnered considerable preservation interest from state and Federal agencies and other private organizations. The western yellow-billed cuckoo was Blue-Listed by the National Audubon Society from 1972 to 1981 and was downlisted to species of special concern in 1982 based on temporary population increases identified during 1982 bird censuses (Ehrlich et al. 1992). In 1986 the USFWS listed western populations as category 2 (candidate for listing). Even though western yellow-billed cuckoo populations had been experiencing continued declines, in 1990 the USFWS downlisted the western populations to category 3B (former candidate for listing) because it was no longer considered a valid subspecies (Ehrlich et al. 1992).

Drought and scarcity of prey, especially the loss of sphinx-moth caterpillars (*Eumorpha fasciata*) to pesticides, appear to play a role in the western yellow-billed cuckoo decline, even where suitable nesting habitat remains (Ehrlich et al. 1992). In 2001 the USFWS published a 12-month finding for a petition to list the western yellow-billed cuckoo in the western United States and found that listing the western yellow-billed cuckoo distinct population segment (DPS) as threatened is warranted but precluded by higher-priority listing actions (Halterman 2004). The USFWS concluded that the overall magnitude of threats to the western yellow-billed cuckoo DPS are high but not imminent and assigned the species a listing priority number of 6 (Federal Register 2001). In Nevada and California, the western yellow-billed cuckoo is listed as endangered, and the Arizona Game and Fish

Department listed the western yellow-billed cuckoo as a species of special concern (Halterman 2004).

Nesting habitat throughout much of the western yellow-billed cuckoo range consists of trees or shrubs in open woodlands with dense undergrowth, extensive riparian woodlands, thickets, or parks (Ehrlich et al. 1992). Along the lower Colorado River, mature stands of cottonwood and willow provide the primary habitat. Although once thought to be fairly common throughout the lower Colorado River Valley, the largest remaining western yellow-billed cuckoo population occurs in the Bill Williams Delta. Populations are apparently decreasing elsewhere in the valley (Anderson and Ohmart 1984, Rosenberg et al. 1991).

The Yuma clapper rail (*Rallus longirostris yumanensis*) is one of three western rail races protected under the endangered species act (Conway et al. 1993) and is the only Federally listed avian species restricted primarily to the lower Colorado River Valley (Rosenberg et al. 1991). Federally, the Yuma clapper rail has been designated an endangered species (32 FR 4001, March 11 1967) and is protected under the Endangered Species Act of 1973. In California the species is recognized as rare (California Administrative Code, Title 14, Section 670.5), and Nevada recognizes the rail as endangered (Nevada Administrative Code, Sections 503.015 to 503.080, December 21, 1981).

Prior to the construction of dams on the Colorado River, Yuma clapper rail was thought to have been restricted primarily to the Colorado River Delta of Mexico. It is widely believed that the northward expansion of Yuma clapper rail distribution is directly related to an increased availability of suitable habitat resulting from dam construction and subsequent river management and control (Ohmart and Smith 1973, Tomlinson and Todd 1973, Ohmart et al. 1975). Generally, Yuma clapper rail are associated with dense vegetation, although they are frequently found in cattail-bulrush communities with only moderate foliage densities (Anderson and Ohmart 1985).

Yuma clapper rail in general are large, light gray to dull cinnamon in color with a slender, slightly decurved bill. Their bodies are compressed laterally and legs and toes are long relative to their body size. Yuma clapper rail are second in size to the king rail (*Rallus elegans*) of the gulf and east coasts of North America. Adults measure between 32 and 41 cm in length and weigh between 160 and 400 g. Males and females are alike in plumage, but males average 20 % larger in size. On males, the base and sides of the bill are bright flesh to bright orange in color; females bills are duller. Ventrally, Yuma clapper rail are uniform in color; they are darker dorsally. Back feathers have blackish or dusky centers of varying width, and flanks are marked with alternating white and dusky to dull black bars. Juvenile Yuma clapper rail have paler bills than adults, and their plumage varies from very dull gray to dark gray, with nearly black flanks and sides. Streaking on the flanks and under tail coverts is indistinct. Downy young are black and difficult to distinguish from young king rail or Virginia rail (*Rallus limicola*) (Eddleman and Conway 1998).

California Black Rail

Distribution

The California black rail is likely a permanent resident found within coastal California, northwestern Baja California, the lower Imperial Valley, and the lower Colorado River of Arizona and California

(Eddleman et al. 1994). However, researchers (Evens et al. 1991, Eddleman et al. 1994) believe that most California black rail are restricted to the tidal marshes of the northern portion of San Francisco Bay estuary with smaller, fragmented subpopulations in southeast California and western Arizona.

The California black rail population found in the lower Colorado River Valley is thought to be the only stable inland population in the western United States (Rosenberg et al. 1991). The highest densities documented within the valley have been found below Imperial Dam and at Mittry Lake (Repking and Ohmart 1977, Evens et al. 1991). Smaller populations have been found around the Salton Sea, West Pond, along the All-American Canal, at the southern end of the Coachella Canal, around the Finney/Ramer Lakes, and within the Bill Williams Delta (Gould 1975, Rosenberg et al. 1991, Conway et al. 2002). Distribution and habitat preferences, however, may change with water fluctuations (Repking and Ohmart 1977, Evens et al. 1991). In general, rallid distributions (including the California black rail), are likely determined by flood duration and water depth during the breeding season (Eddleman et al. 1988).

Migration

Eastern black rail likely migrate to wintering sites; however, some individuals may remain in their breeding range year-round (Eddleman et al. 1994). Adult California black rail are thought to remain within their breeding range throughout the year. However, juveniles do disperse erratically, although no long-distance movements have been documented (Rosenberg et al. 1991, Evens et al. 1991, Eddleman et al. 1994).

Habitat

California black rail inhabit dense, emergent vegetation in salt marshes, shallow freshwater marshes, wet meadows, and flooded grasslands (Eddleman et al. 1988, Eddleman et al. 1994). Common plants within these habitats include: saltmarsh cordgrass (*Spartina alterniflora*), saltmeadow grass (*Spartina patens*), pickleweed (*Salicornia* spp.), and small bulrush (*Scirpus americanus*; *S. olneyi*) (Eddleman et al. 1988).

The California black rail selects areas near marsh edges with shallow standing water (< 2.5 cm deep) dominated by bulrushes and cattails (Andersen and Ohmart 1984, Flores 1991, Rosenberg et al. 1991, Flores and Eddleman 1995). Juvenile California black rail use more shoreline vegetation (i.e., salt grasses and shrubs) than adults, which may be because the young have a tendency to explore new areas during the post-breeding season. Structure and morphology appear to be more important than species composition in determining optimum California black rail habitat (Flores and Eddleman 1995). Typically, California black rail will select areas with high stem densities and high canopy coverage in shallow water located close to upland vegetation (Flores and Eddleman 1995). California black rail will also select areas with gradually sloping shorelines that exhibit minimal water level fluctuations (Repking and Ohmart 1977, Rosenberg et al. 1991).

Nesting

Nest sites are usually located within a mixture of dead or decaying vegetation and young growth. Nests are generally placed on top of moist soil or above shallow water areas and usually constructed

on sites with higher elevation than the surrounding habitat (Eddleman et al. 1994). Water depth at five nest sites in Arizona was approximately 1.2 cm, and nests were placed approximately 6.4 cm above the substrate (Flores and Eddleman 1993). Substrate immediately around nest sites is typically interspersed with moist soil and small pools of water (Eddleman et al. 1994). Nests are usually placed at the center of emergent vegetation, fashioned into well-defined bowls with a ramp, and placed under a canopy of vegetation (Bent 1926, Eddleman et al. 1994). Nests are usually constructed within plants such as needlerush (*Juncus* spp.), cattails, saltgrass (*Distichlis spicata*), bulrushes, and cordgrass (*Spartina* spp.) (Eddleman et al. 1994). Five nests in Arizona were constructed with California bulrush (*Schoenoplectus californicus*), three-square bulrush (*Schoenoplectus pungens*), and cattails (Rosenberg et al. 1991).

Food Habits

Although California black rail foraging behavior is poorly understood, they are thought to forage near substrate edges such as those found at the edge of emergent vegetation. Considering their generalized bill shape, California black rail probably glean prey items from the soil or vegetation and likely rely on sight to locate food (Eddleman et al. 1994). California black rail feed on spiders, beetles, ants, leafhoppers, and snails, as well as cattail and bulrush seeds. The California black rail's diet likely changes considerably during the winter when invertebrates are less abundant (Rosenberg et al. 1991, Ehrlich et al. 1992).

Conservation and Management

Effects of Human Activity

Most populations of California black rail are now threatened with extinction (Conway et al. 2002), and local populations are thought to be declining due to loss or degradation of suitable habitat (Repking and Ohmart 1977, Eddleman et al. 1988, Evens et al. 1991, Eddleman et al. 1994). Historically, the Colorado River was known for its extreme variability in flow regimes and associated hydrogeographical components (Reclamation 1999). Unlike the Yuma clapper rail, the construction of dams and subsequent water regulation has not promoted the development of suitable California black rail habitat. Since California black rail usually occupy transition zones between upland and emergent vegetation, they are thought of as being considerably more vulnerable to water-level fluctuations than other marsh birds (Flores and Eddleman 1995). Other threats to the species include marsh subsidence caused by groundwater removal and wildfires (Eddleman et al. 1988).

Degradation of Backwater Habitat

Flow-control structures, such as dams, have greatly altered connected streamside communities (Patten 1998) and destroyed many of the backwater communities once present along the lower Colorado River. Reductions in sediment loads, narrowing of channels, reduction in the recruitment of vegetation, and increase of flow velocities are known to contribute to the loss of riparian-wetland, marsh, and backwater communities (Brown and Johnson 1985, Collier et al. 1996, Auble and Scott 1998, Friedman et al. 1998). Marshlands on the lower Colorado River are threatened by encroachment of saltcedar, *Phragmites*, and other exotics. Salt cedar generally increases the likelihood of catastrophic fire (Bock and Bock 1990, Smith et al. 1998) and may desiccate water courses (Vitousek 1990, DiTomaso 1998). Removal of groundwater, coupled with regulated flows, has contributed significantly to the loss of riparian communities and promoted erosion through the loss of stream bank stabilization by plant roots (Groeneveld and Griepentrog 1985).

Habitat Criterion and Research Needs for Assessing the Value of Habitat

As indicated in Table 1, California black rail habitat can be generally thought of as consisting of high stem density, high foliage density, low water depth, gradual slopes, and high or dry ground. Additionally, California black rails generally nest above moist soils or pools of shallow water, occupy the transition zone between marsh and upland communities, and apparently do not tolerate fluctuations in water levels very well. Although there is a very generalized understanding of California black rail habitat requirements along the lower Colorado River, there remains much to be learned prior to the implementation of a habitat restoration plan that targets California black rail habitat requirements. Studies aimed at determining the species' patch or habitat size requirement and the maximum amount of daily fluctuation in water levels that can be tolerated should be conducted. Additional studies regarding water should target the implications of seasonal fluctuations in water levels and whether or not California black rails are more sensitive to water fluctuations during specific seasons. Finally, water level studies should also include whether or not slow changes in water levels would be tolerated more than rapid changes and whether or not water velocity influences whether or not California black rails will select specific marsh communities.

Table 1. Wetland/marsh generalized habitat requirements for California black rail.

CALIFORNIA BLACK RAIL GENERALIZED HABITAT REQUIREMENTS	
Marsh structural type[a]	1, 5
Patch size	Unknown
Stem density	High
Foliage density	High
Water depth[b]	Low
Shoreline slope	Gradual
High/dry ground	Yes
Water velocity	Unknown
Substrate beneath nest	Moist soil or water
Proximity to uplands	Transition zone
Daily water level fluctuation	Low
Nest site elevation	Higher or equal to surroundings
Nesting period[c]	Early, mid-season, late

[a] Marsh structural type as defined by Anderson and Ohmart (1976).
[b] Low = < 3.0 centimeters, medium = 3.0-6.0 centimeters, high = > 6.0 centimeters.
[c] Early = February - April, mid-season = May - June, late = July - September.

Eddleman et al. (1988) suggest that habitat created for the Yuma clapper rail would likely benefit the California black rail although black rail studies should be conducted to confirm similarities in habitat requirements. In areas where California black rail are encountered most frequently, management efforts should be implemented immediately to prevent further declines in existing populations, although potential management practices should be studied and evaluated prior to implementation (Conway et al. 2002). Because California black rail are year-round residents across their range, care should be taken when considering any potential management practices. Activities

such as dredging, prescribed burning, and water-level manipulations should be carefully considered (Flores 1991).

Current projects in progress along the lower Colorado River are assessing whether or not fire can be used to restore California black rail habitat. Study areas are located within the Imperial National Wildlife Refuge (NWR), at Mittry Lake, Havasu NWR, and at the Salton Sea. Although it is generally believed that fire can be successfully used in the restoration of California black rail habitat by removing old, decadent vegetation, insufficient data has been collected to formulate any conclusions (C.J. Conway 2005 pers. comm.).

Yuma Clapper Rail

Distribution

Distribution of the Yuma clapper rail was first described by Dickey (1923) as the freshwater marshes along the lower Colorado River above Yuma and adjacent irrigation canals near Laguna Dam. Moffitt (1932) and Abbott (1940) expanded the Yuma clapper rail's distribution to include the eastern and southern portions of the Salton Sea in California. The Yuma clapper rail's distribution along the lower Colorado River was further refined by Phillips et al. (1964) to include the area from Laguna Dam northward to the Bill Williams Delta above Parker Dam. In 1966 Yuma clapper rail were documented at Beal Lake in Topock Marsh and a backwater marsh in Topock Gorge (Tomlinson and Todd 1973).

Based on a lack of responses to recorded calls during the winter, Yuma clapper rail were considered a migratory species that breeds in freshwater marshes of the lower Colorado River (Ohmart and Smith 1973, Tomlinson and Todd 1973, Banks and Tomlinson 1974, Gould 1975, Anderson and Ohmart 1985); however; more recent telemetry studies have suggested that the majority (more than 70 %) of the breeding population remains along the lower Colorado River during the winter months (Eddleman 1989, Conway et al. 1993).

In 2003, while conducting southwestern willow flycatcher surveys along the Virgin and lower Colorado rivers, Koronkiewicz et al. (2004) documented Yuma clapper rail near Mesquite, Nevada; Topock Marsh, Arizona; and Mittry Lake in Arizona and California. Based on surveys conducted in Arizona and California, the USFWS (2003) recorded the largest Yuma clapper rail populations since 1994. Increases were noted at the Salton Sea NWR, Havasu NWR, and in the Laguna Division of the lower Colorado River. In 2005 Yuma clapper rail were documented at Topock Marsh and Cibola and Imperial NWRs (McLeod et al. 2005).

Areas of highest Yuma clapper rail density during the breeding season include Mittry Lake, West Pond, Imperial NWR, the Bill Williams Delta, Topock Gorge, and Topock Marsh. These areas are considered among the largest of marshes along the lower Colorado River. Smaller Yuma clapper rail populations occur where moderately extensive emergent vegetation is present (Tomlinson and Todd 1973, Gould 1975, Rosenberg et al. 1991). In 2003 the USFWS reported 291 Yuma clapper rails at Imperial NWR, 154 at Salton Sea NWR, 66 at Topock Gorge, 61 at Cibola NWR, and 46 at Imperial NWR (USFWS 2003).

Prior to the use of radio telemetry, Yuma clapper rail were known during winter only by their responses to recorded calls and thought to be largely restricted to the area between Imperial and Laguna dams (Tomlinson and Todd 1973, Rosenberg et al. 1991). Radio telemetry studies, however, have suggested that most radio-marked Yuma clapper rail did not respond to recorded calls, and that in general exhibited a tendency towards seasonal shifts in habitat use. During the winter months, rail showed an increase in daily movements, within-day movements, and home ranges. It is possible that by seasonally adjusting characteristics of preferred habitat and overall distribution, experience increased foraging opportunities while experiencing less risk of predation (Eddleman 1989, Conway et al. 1993).

Along the lower Colorado River, Yuma clapper rail have exhibited narrower habitat breadth during the winter months then at any other time of year (Anderson and Ohmart 1985, Conway et al. 1993).

Prior to the construction of dams on the Colorado River, Yuma clapper rail were thought to be restricted primarily to the Colorado River Delta of Mexico. It is widely believed that the northward expansion of the Yuma clapper rail's distribution is directly related to an increased availability of suitable habitat resulting from dam construction and subsequent river management and control (Ohmart and Smith 1973, Tomlinson and Todd 1973, Ohmart et al. 1975). The introduction of crayfish (primary source of food) along the lower Colorado River may have also contributed to the northward expansion of Yuma clapper rail populations (Ohmart and Tomlinson 1977).

Migration

Based on responses to taped calls, the Yuma clapper rail was thought to be a migratory species that occurred along the lower Colorado River from April 21 to late June (Ohmart and Smith 1973, Bennett and Ohmart 1978, Anderson and Ohmart 1984) and wintered at the Colorado River Delta in Mexico. Phillips et al. (1964), however, questioned the migratory status of the species. Radio telemetry studies indicated that more than 70 % of the breeding population winters along the lower Colorado River (Eddleman 1989, Conway 1990).

Habitat

Along the lower Colorado River, primary Yuma clapper rail habitat is best described as palustrine persistent emergent wetlands. This community type consists of non-tidal wetlands dominated by persistent emergent plants, most commonly cattail and bulrush. Although these wetlands can be found adjacent to river channels, they are more frequently found in large impoundments such as behind dams and in backwater areas where water is slow moving and the channel is shallow (Cowardin et al. 1979).

Yuma clapper rail are generally associated with dense vegetation; however, they regularly reach highest densities in cattail-bulrush communities with only moderate foliage densities and have been documented in areas consisting of dense reed (*Phragmites* spp.) and sparse cattail-bulrush (Anderson and Ohmart 1985).

Specific characteristics described as critical components of year-round Yuma clapper rail habitat include: low emergent stem densities and low presence of residual vegetation (Anderson and Ohmart

1985, Conway et al. 1993); marsh vegetation that averages greater than 2 m heigh (Anderson and Ohmart 1985, Eddleman 1989); the presence of dry ground or isolated islands between water and banks to allow for foraging and walking (Gould 1975, Anderson and Ohmart 1985, Conway et al. 1993); the presence of flowing water through many small channels that measure from 0.5-3 m in width or small bodies of open water measuring 0.02-0.2 ha in size (Tomlinson and Todd 1973, Gould 1975); extensive areas of water where overall depth does not exceed 0.3 m with little or no fluctuation in daily water levels (Tomlinson and Todd 1973, Gould 1975).

Ideally, good Yuma clapper rail habitat should consist of a mosaic of emergent vegetation in stands of differing ages interspersed with shallow pools of open water (Eddleman and Conway 1998). Habitat quality is greatly improved when dry ground is present and appears unaffected when dry ground is present as very narrow strips (Anderson and Ohmart 1985). High or dry ground in relation to rail habitat use typically refers to sites only slightly higher than the water level or prevailing substrate depth (Eddleman 1989).

Generally, Yuma clapper rail along the lower Colorado River encounter frequent fluctuations in water levels. Yuma clapper rail are capable of tolerating frequent water-level fluctuations if marsh vegetation remains undisturbed, water is consistently present, changes in substrate depth occur gradually, and slightly higher sites or upland edge are nearby. The primary issues of concern may likely be related to the timing and speed at which water levels rise (Eddleman 1989). If water rises quickly during the breeding season, nests may be lost, and young, newly hatched Yuma clapper rail with downy plumage may drown (Smith 1975). During other seasons, rapid increases in water level may force Yuma clapper rail into upland areas that make them more susceptible to predation. If implemented early in the breeding season, a rapid drop in water level will expose additional habitat and may provide additional nesting or foraging sites (Eddleman 1989). Bennett and Ohmart (1978) suggest, however, that prolonged periods of low water may be detrimental to wetlands and could result in nest abandonment by Yuma clapper rail.

Although marsh size has been cited as a critical component of Yuma clapper rail habitat (Smith 1975, Gould 1975), Anderson and Ohmart (1985) found that marsh size is independent of Yuma clapper rail density per unit area. Yuma clapper rail are known to exhibit changes in seasonal use of available habitat (Anderson and Ohmart 1985, Eddleman 1989, Conway et al. 1993). Eddleman (1989) found that male Yuma clapper rail differed from females in the size of home ranges used in all seasons. Females had smaller home ranges than males during late breeding, incubation, and early winter, and larger home ranges than males during post breeding. Although movement by Yuma clapper rail varied by season, movement occurred most commonly from late June through October after nesting. At Mittry Lake observed Yuma clapper rail shifted their home ranges or moved more than 500 m from June to October. Eddleman (1989) also found that Yuma clapper rail have a tendency to shift habitats used in response to increases in water level, and have been found along mesquite-marsh edges during periods of high water across all seasons. It is possible that seasonal changes may also serve to reduce predation and improve foraging efficiency (Smith 1975, Todd 1986, Eddleman 1989).

Smith (1975) found that both Yuma clapper rail and crayfish were most abundant in moderately dense cattail and bulrush communities at Topock Marsh. However, Bennett and Ohmart (1978) found that crayfish and Yuma clapper rail were most commonly found in the densest stands of

cattails at the Salton Sea and that bulrush communities were not significantly used. Differences in Yuma clapper rail habitat use between the Salton Sea and Topock Marsh may be a reflection of differences in habitats where crayfish are most abundant (Anderson and Ohmart 1985). Along the lower Colorado River, an annual peak in crayfish populations has been shown to correspond with Yuma clapper rail hatching and brood-rearing stages, and crayfish abundance is lowest during the winter, which corresponds with seasonal shifts in use of available habitats by Yuma clapper rail (Bennett and Ohmart 1978, Eddleman 1989, Conway et al. 1993).

Seasonal shifts in use of available habitat include the following characteristics (Eddleman 1989, Conway et al. 1993): in early winter (November-December) Yuma clapper rail have shown a preference for lower emergent stem density and ground coverage, shorter distances to water, dense overhead coverage by vegetation, greater distances to adjacent uplands and vegetative edges, increased water depth and water coverage, and taller emergent plants. Late winter (January-February) preferences included low overhead coverage by vegetation, shorter distances to adjacent uplands, and low stem densities. Yuma clapper rail exhibit narrower habitat breadth during all winter months than at any other time of year (Anderson and Ohmart 1985, Conway et al. 1993). High rail densities during the winter are more strongly correlated with dense cattail-bulrush communities than during other seasons (Anderson and Ohmart 1985). Habitat discrimination by Yuma clapper rail appears greatest during the early winter months when features of occupied habitat showed more differences as compared to available habitat than other seasons (Conway et al. 1993).

During the early breeding season (March-April) Yuma clapper rail exhibited a preference for habitat that was close to open water, and characteristics of preferred habitat during the late breeding season (May-July) included short distances to vegetative edges, greater distances to adjacent uplands, shorter distances to vegetative edge, and short distances to dry ground. During the post-breeding season (August-October), Yuma clapper rail were found to prefer areas with low overhead coverage by vegetation, short distances to vegetative edges, greater distances to adjacent uplands, decreased water depth, and decreased basal coverage by emergent vegetation (Conway et al. 1993).

During the spring Yuma clapper rail are usually associated with dense stands of cattails and bulrush; however, unexpectedly high Yuma clapper rail densities have been documented in dense stands of reed and in moderately dense cattail stands interspersed with open water.

High Yuma clapper rail populations during the summer are less highly correlated with dense cattail-bulrush marshes than at other times of the year and positively correlated with sparse cattail-bulrush marshes that measure less than 1 m tall. Rail densities are highly correlated with dense cattail-bulrush communities during the late summer and fall; however, Yuma clapper rail densities are negatively correlated with those marshes being invaded by trees and grasses. Breadth of habitat use during late summer and fall declines when compared with summer but is broader than habitat found during winter and spring (Anderson and Ohmart 1985).

Although Eddleman (1989) found that seasonal variation in the selection of emergent cover types by Yuma clapper rail was consistent with previous studies (Smith 1975, Bennett and Ohmart 1978, Anderson and Ohmart 1985, Todd 1986), several exceptions were reported. Yuma clapper rail utilized willow and willow-salt cedar communities in response to rising water levels during the late-breeding and post-breeding seasons. Yuma clapper rail were also reported to have utilized upland

edges and arroweed communities during the late breeding season, which was also in response to rising water levels.

Generally, the species or density of emergent vegetation does not appear very important to Yuma clapper rail so long as vegetation provides cover and nest sites, and proper water levels are maintained (Anderson and Ohmart 1985, Eddleman 1989). While the presence of residual vegetation as a component of habitat is important for feeding and movement during periods when water levels are high, it probably impedes movement in shallow areas (Eddleman 1989). Water depth and the amount of mud flats present may be the most significant criterion used in the selection of nesting sites in the Imperial Valley (Bennett and Ohmart 1978). Other factors thought to influence nest site selection include: water level consistency throughout the nesting period, the presence of high ground in the form of dikes or high ground for loafing and rearing young, and water depths of 1-15 cm.

Nesting

In the Imperial Valley of California, nests were found in cattail stands where they were typically constructed of dry, interwoven cattail with interspersed grasses and twigs. Located between 35-54 cm above the water surface, nests usually included a ramp composed of fallen cattail leading from the water surface, unless the nests were constructed above relatively dry ground. Ramps were known as a means by which to enter and exit the nest. When located above relatively dry ground, nests were constructed on fallen cattails between 5-16 cm above the ground surface (Bennett and Ohmart 1978). Eddleman (1989) reports that 50 % of nests studied along the lower Colorado River lacked ramps, which may be more common in areas where water levels are high or frequently fluctuate.

Brooding platforms or loafing nests constructed using mats of dead emergent vegetation are frequently found at the base of cattails on mud flats and are thought to serve as a place on which to loaf and cast pellets after feeding. These non-breeding nests differ from other nests in that they are capable of floating when water levels are high (Eddleman and Conway 1998).

Food Habits

Based on the analysis of stomach contents from 10 Yuma clapper rail collected along the lower Colorado River, Topock Marsh south to Imperial Reservoir, they were found to consume mostly introduced exotic crayfish; however; clams (*Corbicula* sp.), isopods, water beetles, and small fish were also found (Ohmart and Tomlinson 1977).

Along the lower Colorado River, Yuma clapper rail are generally found foraging in sites represented by high water coverage, relatively low stem densities (compared with other marsh areas) and water depth of approximately 7.5 cm (Eddleman 1989, Conway et al. 1993). Increases in daily movements, within-day movements, and home ranges during winter months may be a reflection of decreased crayfish abundance where such movements may improve foraging opportunities (Conway et al. 1993).

Conservation and Management

Effects of Human Activity

Historically the Colorado River was known for its extreme variability in flow regimes and associated hydrogeographical components (Reclamation 1999). The construction of dams and subsequent control of river flows promoted the development of suitable Yuma clapper rail habitat (Ohmart and Smith 1973), which resulted from the deposition of sediments and permitted Yuma clapper rail to expand their range northward from the Colorado River Delta in Mexico (Ohmart and Smith 1973, Anderson and Ohmart 1985, Eddleman 1989). Despite increases in the amount of available habitat in the United States, the loss of marshes caused by dredging, riprapping of stream banks, and high water flows remain the primary threats to the species (Todd 1986).

The effects of most pesticides and contaminants on rail are poorly understood, although clapper rail have shown a high tolerance to DDT and DDD (Eddleman et al. 1988). The presence of selenium in Yuma clapper rail, their eggs, and their food (crayfish) has been documented. Selenium occurred in concentrations that are known to have effects on the reproduction of mallards (*Anas platyrhynchos*). The presence of selenium in the lower Colorado River has been linked to areas above Davis Dam and may have resulted from natural processes, burning coal for generation of electricity, uranium and coal mining, or agricultural practices that include irrigation. The effects of selenium on Yuma clapper rail are largely unknown and may potentially impact reproduction (Eddleman 1989).

Predation by raptors is thought to be the primary factor contributing to mortality of Yuma clapper rail on the lower Colorado River. Eddleman (1989) suggests that Yuma clapper rail are most susceptible to predation during the late summer through early fall when they begin to move over larger home ranges. Increased Yuma clapper rail movements coincide with the arrival of wintering raptors during December and January, and predation by raptors is thought to be related to releases of water and subsequent increase in water levels. It is possible that by decreasing the rate at which water levels rise and changing the timing of releases to coincide with the departure of wintering raptors, predation may be minimized.

Ohmart and Smith (1973) suggest that availability of suitable habitat and food are the two primary factors responsible for controlling Yuma clapper rail populations along the lower Colorado River. As long as marsh or backwater habitat for both the rail and crayfish is maintained, the Yuma clapper rail is likely to persist along the lower Colorado River (USFWS 1983).

Degradation of Backwater Habitat

Water regimes along the lower Colorado River range from nominally stable at sites disconnected from the river to seasonally unstable and unpredictable on wetlands connected to the channel. Rapidly rising water levels during the nesting season may flood connected wetlands or backwater habitat and result in the loss of nests and young. Decreased flows may temporarily increase the amount of available foraging habitat, but if low flows persist over extended periods of time, wetlands would likely desiccate, resulting in Yuma clapper rail nest abandonment (Eddleman 1989). Additionally, flow-control structures, such as dams, have greatly altered connected streamside communities (Patten 1998). Reductions in sediment loads, narrowing of channels, reduction in the recruitment of vegetation, and increase of flow velocities are known to contribute to the loss of

riparian-wetland, marsh, and backwater communities (Brown and Johnson 1985, Collier et al. 1996, Friedman et al. 1998, Auble and Scott 1998).

Habitat Criterion and Research Needs for Assessing the Value of Habitat

As indicated in Table 2, Yuma clapper rails generally require patch sizes of more than 8 hectares in size, moderately high to high stem density, high foliage density, low water depth, gradual shoreline slopes, and slow water velocities. Nest sites usually include the presence of either moist soils or shallow water. Habitat structure appears to differ seasonally where Yuma clapper rails are known to occupy marsh communities and other areas that are in closer proximity to upland areas. Although Yuma clapper rails are thought to tolerate fluctuations in water levels, fluctuations should occur slowly and gradually. Nest sites are generally located on substrate that is either equal to or slightly higher in elevation than surrounding habitat.

Table 2. Wetland/marsh generalized habitat requirements for Yuma clapper rail.

YUMA CLAPPER RAIL GENERALIZED HABITAT REQUIREMENTS	
Marsh structural type[a]	1, 3, 5, 7
Patch size	> 8 hectare
Stem density	High/Moderately high
Foliage density	High
Water depth[b]	Low
Shoreline slope	Gradual
High/dry ground	Yes
Water velocity	Slow
Substrate beneath nest	Moist soil or water
Proximity to uplands	Marsh, transition zone
Daily water level fluctuation	Low - gradual change
Nest site elevation	Higher or equal to surroundings
Nesting period[c]	Early, mid-season, late

[a] Marsh structural type as defined by Anderson and Ohmart (1976).
[b] Low = < 3.0 centimeters, medium = 3.0-6.0 centimeters, high = > 6.0 centimeters.
[c] Early = February - April, mid-season = May - June, late = July - September.

A variety of studies of clapper rail habitat, such as those conducted by Tomlinson and Todd (1973), and surveys of *yumanensis* habitat (Gould 1975, Eddleman 1989) have led to differing opinions of which criterion actually define suitable Yuma clapper rail habitat (Conway et al. 1993). Until relatively recently, *yumanensis* was thought to be migratory due to a lack of responses to taped calls during winter surveys (Tomlinson and Todd 1973, Smith 1975, Bennett and Ohmart 1978, Todd 1986). However, more recent studies (Eddleman 1989, Conway et al. 1993) revealed, through the use of radio telemetry, that more than 70 % of the Yuma clapper rail population remains along the

lower Colorado River during the winter. As such, our understanding of seasonal habitat requirements and Yuma clapper rail responses to seasonal variation in water levels is somewhat limited.

Specific characteristics described as critical components of year-round Yuma clapper rail habitat include: low emergent stem densities and low presence of residual vegetation (Anderson and Ohmart 1985, Conway et al. 1993); marsh vegetation that averages greater than 2 m in height (Anderson and Ohmart 1985, Eddleman 1989), the presence of dry ground or isolated islands between water and banks to allow for foraging and walking (Gould 1975, Anderson and Ohmart 1985, Conway et al. 1993), the presence of flowing water through many small channels that measure from 0.5-3 m in width or small bodies of open water measuring 0.02-0.2 ha in size (Tomlinson and Todd 1973, Gould 1975), and extensive areas of water where overall depth does not exceed 0.3 m with little or no fluctuation in daily water levels (Tomlinson and Todd 1973, Gould 1975). Optimal year-round habitat appears as a mosaic of emergent vegetation in stands of differing ages interspersed with shallow pools of open water (Eddleman and Conway 1998), and consists of cattail and bulrush (Tomlinson and Todd 1973, Gould 1975, Bennett and Ohmart 1978, Todd 1986). Generally, Yuma clapper rail select marshes consisting of emergent cover types (Todd 1986), although cover types have frequently been described as insignificant so long as occupied communities are either partially flooded or covered in shallow water (Ohmart and Smith 1973, Anderson and Ohmart 1985, Eddleman 1989).

During the breeding season, preferred Yuma clapper rail microhabitat consists of shallow water measuring less than 30 cm in depth (Gould 1975, Bennett and Ohmart 1978, Todd 1986) and vegetation usually measuring more than 40 cm tall (Todd 1986). Marshes will have an interface between upland and marsh communities or will have higher ground as a critical component (Tomlinson and Todd 1973, Gould 1975, Smith 1975, Bennett and Ohmart 1978). Eddleman (1989) defines high ground as sites only slightly higher than the water level or the prevailing substrate. Examples of high ground in marshes include gradually sloping shorelines and areas that have resulted from deposition at the sides of existing and filled channels. Microhabitat during the breeding season should also consist of a mat of residual vegetation that covers at least 5 % of the area (Ohmart and Smith 1973, Bennett and Ohmart 1978, Todd 1986). Stable water levels during the breeding season are also thought to be important (Gould 1975, Bennett and Ohmart 1978). During the early breeding season (March-April) Yuma clapper rail prefer habitat that is close to open water. During the late breeding season (May-July) Yuma clapper rail prefer areas in close proximity to vegetative edges, greater distances to adjacent uplands, shorter distances to vegetative edge, and short distances to dry ground. During the post-breeding season (August-October), rail will occupy areas with low overhead coverage by vegetation, short distances to vegetative edges, greater distances to adjacent uplands, decreased water depth, and decreased basal coverage by emergent vegetation (Conway et al. 1993). Characteristics of microhabitat during the summer include sparse cattail-bulrush marshes that measure less than 1 m tall. Microhabitat used during the late summer and fall includes dense cattails and bulrush, and trees and grasses are absent from occupied areas (Conway et al. 1993). In early winter (November-December), clapper rail prefer low emergent stem density and ground coverage, shorter distances to water, dense overhead coverage by vegetation, greater distances to adjacent uplands and vegetative edges, increased water depth and water coverage, and taller emergent plants. Late winter (January-February) preferences included low overhead coverage by vegetation, shorter distances to adjacent uplands, and low stem densities.

Eddleman (1989) suggests that management of existing Yuma clapper rail habitat along the lower Colorado River is not necessary, as long as appropriate water flows are not disrupted. Instead, emphasis should be placed on the creation of new marshes and backwater habitat. Since Yuma clapper rail appear to be highly dependent upon crayfish as their primary source of food, studying crayfish to examine seasonal movements, population dynamics, and habitat requirements should be considered. Understanding crayfish biology on the lower Colorado River should help to further refine seasonal habitat requirements and implications of existing management practices for both the Yuma clapper rail and crayfish. Densely vegetated marshes with shallow water appear to be relatively common along the lower Colorado River and provide adequate nesting habitat for Yuma clapper rail. Interior marshlands with low basal cover and deep water, however, are uncommon and provide few wintering and possibly foraging sites (Conway et al. 1993). Therefore, emphasis should be placed on the creation of interior marshlands rather than breeding habitat. Newly created marshes should be as large as possible to allow for increased foraging opportunities across all seasons. These marshes would provide variation in topography to minimize effects of water-level fluctuations. Larger habitat blocks in breeding areas would provide additional nesting sites and escape cover (Eddleman 1989). Although Yuma clapper rail habitat requirements in terms of plant species, water depth, vegetation height, etc. are reasonably well described, our understanding of vegetation measurements in habitat across all seasons is lacking. Variables such as emergent coverage, residual mat coverage, water depth and coverage, and emergent stem density need to be more clearly understood in order to more effectively create suitable habitat.

Current projects being conducted along the lower Colorado River are assessing whether or not fire can be used to restore Yuma Clapper Rail habitat. Study areas are located within the Imperial National Wildlife Refuge (NWR), at Mittry Lake, Havasu NWR, and at the Salton Sea. Although it is generally believed that fire can be successfully used in the restoration of Yuma Clapper Rail habitat by removing old, decadent vegetation, insufficient data has been collected to formulate any conclusions (C. Conway, University of Arizona, Personal Communication).

Least Bittern

Distribution

Within the lower Colorado River Valley, least bittern are considered relatively common breeders from April to September in expansive marsh communities that include Topock Marsh and near Imperial Dam (Andersen and Ohmart 1984, Rosenberg et al. 1991). A maximum density of 41 individuals was recorded at Imperial NWR on 13 July 1978; however, densities are lower in all other portions of the valley (Rosenberg et al. 1991).

Migration

Least bittern migrants typically summer in temperate areas but winter in both temperate and subtropical marsh habitats (Gibbs et al. 1992). Spring migrants usually arrive on their breeding grounds from early to mid March through mid May (Bent 1926). Adult winter migration usually begins in either September or October, while juvenile least bittern may occasionally remain at nesting areas through October after adults have departed (Palmer 1962). Most least bittern populations in the United States are migratory, however, it is currently unclear if individuals found

along portions of the lower Colorado River are migratory or year-round residents. Generally, least bitterns that are migratory will arrive in the lower Colorado River Valley in April and remain through September (Anderson and Ohmart 1984, Rosenberg et al. 1991). Typically, least bittern that nest west of the Rocky Mountains migrate through western Mexico to winter in Costa Rica, while individuals nesting east of the Rocky Mountains migrate to northern Columbia (Hancock and Kushlan 1984).

Habitat

Although least bittern are usually found in a variety of habitats throughout their range (Bent 1926), occupied habitat always includes dense emergent vegetation (Hancock and Kushlan 1984). Typically, least bittern habitat is characterized by both fresh and brackish marshes within tall, dense stands of aquatic and/or semiaquatic vegetation including *Typha* spp., *Carex* spp., *Scirpus* spp., *Sagittaria* spp., or *Myriscus* spp. (Palmer 1962, Hancock and Kushlan 1984, Gibbs et al. 1992). They have also been shown to prefer territories that include interspersed woody vegetation with deep open water (Weller 1961, Palmer 1962). Characteristically, least bittern prefer to forage near open deeper water utilizing emergent vegetation as perches to cling from while attempting to forage for prey on the open-water side (Eastwood 1932, Sutton 1936, Weller 1961).

Within the lower Colorado River Valley, the largest concentrations of least bittern are found in cattail or bulrush habitats within Topock Marsh and near Imperial Dam (Andersen and Ohmart 1984, Rosenberg et al. 1991). Scattered and less-numerous concentrations of least bittern can be found in smaller marsh habitats throughout the Colorado River Valley, including ponds and agricultural canals (Rosenberg et al. 1991). For marsh land cover types defined by Anderson and Ohmart (1976), suitable least bittern habitat usually corresponds to marsh structural types 1, 2, 3, and 5 (Sterling 2005).

Nesting

Least bittern typically choose nest sites among tall, dense stands of emergent vegetation usually consisting of *Typha* spp., *Carex* spp., *Scirpus* spp., but they are occasionally found within *Sagittaria* spp., *Phragmites* spp., *Salix* spp., *Cephalanthus* spp., or *Rhizophora* spp. Nest sites are usually located in clumps of emergent or woody vegetation that measure at least 2 m high (Gibbs et al. 1992). Nests are most frequently constructed directly on top of previous years' vegetation, which serves as a foundation for nest construction. Occasionally, old nests built by other avian species may be used as the nest foundation (Weller 1961).

During initial nest construction, adults will first bend down old or new vegetation to the water surface and then begin piling on plant matter to create a nesting platform (Weller 1961). Platform consistency coincides with available vegetation within that specific breeding territory, but usually the platform vegetation will be of the same species as the nest foundation (Weller 1961). However, Bent (1926) noted two instances where nest platforms were constructed with different material. Bent (1926) described a nest within "rushes" with a platform consisting of "sticks," while a second nest was located within *Typha* spp. but the platform was constructed with *Salix* sticks. Nests are typically constructed 15-76 cm above water and are 8-96 cm in depth (Gibbs et al. 1992).

Least bittern are thought to be non-colonial nesters, but data have shown that under specific conditions, least bittern will breed in concentrated areas (Kushlan 1973). Kushlan (1973) found that least bittern might nest colonially when feeding areas are localized, and Hancock and Kushlan (1984) suggested that colonial nesting behavior exhibited by least bittern, especially when population densities are high, may be a response to an over-abundance of food.

Food Habits

Least bittern found along the lower Colorado River will forage primarily for small fish and amphibians but also for rodents, reptiles, and insects (Andersen and Ohmart 1984).

Conservation and Management

Effects of Human Activity

Habitat destruction and degradation are thought to be the principal range-wide threats to the least bittern (Gibbs et al. 1992). Tiner (1984) estimated that less than half of the wetlands present in the United States at the time of European settlement remain today. Many remaining wetlands have been degraded from their original condition by sedimentation, eutrophication, and chemical contamination. Gibbs et al. (1992) suggest that changes in water quality could adversely affect the least bittern's prey base and increase the potential for impacts from parasites such as nematodes (*Eustrongilides* spp.). Alterations to the hydrology of wetlands, such as drainage or channelization, may reduce least bittern breeding success by drying or flooding potential nest sites (Monfils 2003).

Degradation of Backwater Habitat

Marshes and backwaters that historically occurred along the lower Colorado River have been destroyed or severely degraded because of agricultural conversion, construction of reservoirs, river channelization, and shoreline stabilization. Given the regulated nature of the Colorado River, natural formation of new marshes and backwaters resulting from channel movement and periodic flooding is now unlikely. However, flow regulation and shifts in the timing of flows because of water diversion have resulted in the development of large marsh and backwater complexes where riparian vegetation historically occurred. Marsh complexes developed behind Imperial Dam and Parker Dam at the Bill Williams Delta and Topock Marsh. The construction of training structures also created areas of more expansive and permanent backwaters and marshes than had occurred historically on the lower Colorado River (MSCP 2004).

Habitat Criterion and Research Needs for Assessing the Value of Habitat

Although broadly defined habitat requirements are known (Table 3), detailed information on least bittern reproduction, population structure, home-range sizes, etc. is lacking. As such, establishing habitat requirements and other criterion to effectively manage and stabilize least bittern populations would be very difficult at best (Gibbs et al. 1992). Generally, however, least bittern have exhibited preferences for dense wetland communities with deep water where vegetation is characterized by species of the following genre: *Typha* spp., *Carex* spp., *Scirpus* spp., *Sagittaria* spp., or *Myriscus* spp. (Palmer 1962, Hancock and Kushlan 1984, Gibbs et al. 1992). In North America, however, least bittern are most frequently found in cattail- and sedge-dominated marshes (Hancock and Kushlan 1984). Preferred habitat may also consist of woody vegetation interspersed throughout

Table 3. Wetland/marsh generalized habitat requirements for least bittern.

LEAST BITTERN GENERALIZED HABITAT REQUIREMENTS	
Marsh structural type[a]	1, 2, 3, 5
Patch size	Unknown
Stem density	High
Foliage density	Unknown
Water depth[b]	High
Shoreline slope	Unknown
High/dry ground	Unknown
Water velocity	Unknown
Substrate beneath nest	Water
Proximity to uplands	Marsh
Daily water level fluctuation	Unknown
Nest site elevation	Unknown
Nesting period[c]	Early, mid-season, late

[a] Marsh structural type as defined by Anderson and Ohmart (1976).
[b] Low = < 3.0 cm, medium = 3.0-6.0 cm, high = > 6.0 cm.
[c] Early = February - April, mid-season = May - June, late = July - September.

(Weller 1961, Palmer 1962). Typically, emergent vegetation measures at least 2 m high (Gibbs et al. 1992) and is used for cover, movement over deep water for foraging, and nesting (Eastwood 1932, Sutton 1936, Weller 1961, Palmer 1962, Gibbs et al. 1992). Water depth usually ranges from 10 to 50 cm (Gibbs et al. 1992).

Within the lower Colorado River Valley, the largest concentrations of least bittern are found in cattail or bulrush habitats within Topock Marsh and near Imperial Dam (Andersen and Ohmart 1984, Rosenberg et al. 1991). Scattered and less-numerous concentrations of least bittern can be found in smaller marsh habitats throughout the valley, including ponds and agricultural canals (Rosenberg et al. 1991). Sterling (2005) suggests that least bittern habitat along the lower Colorado River is best described by marsh cover types as defined by Anderson and Ohmart (1976). Marsh cover type 1 (MA-1) consists of nearly 100 % cattails and bulrush with small amounts of *Phragmites* spp. and open water. Marsh cover type 2 (MA-2) is defined as consisting of nearly 75 % cattails and bulrush with many trees and grasses interspersed. Approximately 25 to 50 % of marsh cover type 3 (MA-3) consists of cattails and bulrush with some *Phragmites* spp., open water, trees and grasses. Marsh cover type 7 (MA-7) is defined as open marsh, where approximately 75 % of the total area consists of water adjacent to sparse emergent vegetation and includes sandbars and mudflats when water levels are low. Density of emergent vegetation is thought to, at least in part, dictate the number of least bittern nests found in an area. As such, interspersion of water and cover may be an important characteristic of breeding habitat (Gibbs et al. 1992). Weller and Spatcher (1965), for example, found a greater number of least bittern nests at two Iowa marshes during years when ratios of emergent vegetative cover to open water were equal (the “hemi-marsh” condition), which may

represent optimal least bittern habitat characteristics. In Wisconsin, Manci and Rusch (1988) found that least bittern were restricted to deep-water marshes dominated by cattails and sedges, and avoided these areas when dry.

Studies designed to target least bittern habitat requirements should seek to determine minimum patch size requirements, determine whether or not high or dry ground should be present, and what the implications of high water velocities are on least bittern habitat suitability. Additionally, studies should investigate how tolerant least bitterns are to daily fluctuations in water levels and how the rate at which water levels fluctuate might influence habitat suitability. Studies should also investigate nesting habitat structure and how water level fluctuations might decrease nesting habitat quality. Seasonal tolerance to fluctuations in water levels should also be determined. As previously mentioned, detailed information on least bittern demography, habitat requirements, home range sizes, and breeding requirements is lacking and should be studied extensively before establishing habitat requirements for the species. Along portions of the lower Colorado River, least bittern are thought to be year-round residents. As such, studies that help define these variables are extremely important in determining population distributions seasonally and assessing trends in and responses to seasonal habitat features.

Southwestern Willow Flycatcher

Distribution

The southwestern willow flycatcher is a widely distributed summer resident of much of the United States and southern Canada (Brown 1988). The current breeding range of the southwestern willow flycatcher (*E. t. extimus*) includes Arizona, southern California, New Mexico, southern Nevada, southern Utah, and southwestern Colorado (Unitt 1987, Koronkiewicz et al. 2004). Breeding records and museum collections suggest that the southwestern willow flycatcher may have historically been found along the extreme southern reaches of the lower Colorado River region. Prior to surveys conducted by McKernan in 1996, no breeding confirmation had been documented in more than 50 years (McKernan 1997), although surveys conducted by Arizona Partners in Flight in 1993 did document four or five territorial southwestern willow flycatcher along the lower Colorado River near Yuma, Arizona (McKernan 1997).

During surveys conducted in 1996, McKernan (1997) documented eight willow flycatcher nests at sites near the Lake Mead Delta and at Topock Marsh. During surveys conducted by McKernan and Braden (2001), nests were documented at the Virgin River study site, Mormon Mesa North and South, Mesquite West, and Topock Marsh. Southwestern willow flycatcher were also documented at Pahranagat NWR and Overton Wildlife Management Area (WMA) along the Muddy River (McKernan and Braden 2001).

In 2003 Koronkiewicz et al. (2004) conducted southwestern willow flycatcher presence/absence surveys at 95 pre-selected sites along the Virgin and lower Colorado Rivers. Flycatcher were detected on at least one occasion at 54 of the study sites. Resident breeding southwestern willow flycatcher were detected at Pahranagat NWR, Mesquite, Mormon Mesa and Topock Marsh.

McLeod (2005) also found resident southwestern willow flycatcher at survey locations identified in 2003, in addition to several other locations (Littlefield, Grand Canyon, Overton WMA, and Bill Williams NWR). Although flycatcher were recorded at Overton WMA and Bill Williams NWR, no breeding activity was documented.

Migration

Phillips et al. (1964) suggested that *E. t. extimus* was the first to arrive and has been documented in southern Arizona from May 3 to September 10. Occasionally, individuals have been found away from their breeding areas during early June and from July 21 to early September. Rosenberg et al. (1991) suggest that low-elevation breeding populations, all thought to be *E. t. extimus*, migrated early, arriving on their breeding grounds in late April and early May, whereas montane breeding populations, such as *E. t. brewsteri* and *E. t. adastus*, arrived in mid May and continued to pass through the lower Colorado River Valley through mid-June.

Koronkiewicz et al. (2004) documented southwestern willow flycatcher along the Virgin and lower Colorado Rivers and tributaries between 15 May and 25 July. Most southwestern willow flycatchers south of the Bill Williams River were detected before 18 June, with a single detection recorded on 2 July. As the result of monitoring subsequent to initial sightings at sites south of the Bill Williams River, all individuals recorded were most likely north-bound migrants (Koronkiewicz et al. 2004).

Arrival and departure times of southwestern willow flycatcher vary in relation to sex and age (Sogge et al. 1997, Yong and Finch 1997). Bent (1942) observed that male southwestern willow flycatcher arrived on breeding grounds ahead of females, which may be due to competition among males for high-quality territories, ability to tolerate harsher weather than females, and differences between males and females in travel distances to breeding and wintering areas (Yong and Finch 1997). Adults can start migrating earlier in the fall than young flycatcher because they delay molting until they reach their wintering grounds in Central and South America. Unitt (1987) suggests that young southwestern willow flycatcher molt into their first winter plumage prior to migration, which adds to the length of stay on their breeding grounds.

Habitat

In general, southwestern willow flycatcher prefer territories with high shrub density, open canopies, and moderate ground vegetation cover (Whitmore 1977). They require dense, low-growing woody vegetation (i.e., tree and/or shrub) (Bent 1942) and occur in both mesic and xeric conditions (Barlow and McGillivray 1983, Sedgwick and Knopf 1992).

The southwestern willow flycatcher is an obligate riparian subspecies that requires dense willow thickets (Sedgwick and Knopf 1992) or other available low-growing woody vegetation (Stoleson and Finch 1999) with greater canopy cover (Sedgwick and Knopf 1992) along rivers, streams, marshes, or open water areas (USFWS 1997). Consequently, habitat selection may be influenced by floodplain morphology. That is, southwestern willow flycatcher may select territories within larger floodplains due to the presence of young, dense riparian vegetation communities influenced by fluvial-geomorphic processes (Hatten and Paradzick 2003).

Historically, southwestern willow flycatcher were likely abundant summer residents along the lower Colorado River due to the prevalence of riparian willow thickets (McKernan 1997, McKernan and Braden 2001). However, habitat reduction and degradation have significantly reduced distribution and numbers of this species (USFWS 1997). The invasion of salt cedar was originally thought to have contributed to habitat degradation (Whitmore 1977). However, recent studies (Brown and Trosset 1989, McKernan 1997, Sogge et al. 1997, McKernan and Braden 2001, Koronkiewicz et al. 2004, McLeod et al. 2005) have found that southwestern willow flycatcher do in fact utilize tamarisk within breeding territories. Optimal structural components found within available habitat likely influence habitat selection rather than the presence or absence of native plant communities (Brown and Trosset 1989). Additionally, such structural components (i.e., patch and canopy morphology) vary widely throughout the range of the southwestern willow flycatcher and, therefore, requirements can be broadly defined (Sogge et al. 1997). Specifically, Sogge et al. (1997) split potential southwestern willow flycatcher habitat into four general habitat communities including: monotypic high-elevation willow, monotypic exotic, native broadleaf dominated, and mixed native/exotic.

Monotypic high-elevation willow: Nearly monotypic, dense stands of willow (often *Salix exigua* or *S. geyeriana*) above 2,300 m in Arizona, 3-7 m in height with no distinct overstory layer; often associated with sedges (*Carex* sp.), rushes (*Juncus* sp.), nettles (*Urtica* sp.), and other herbaceous wetland plants; usually very dense structure in lower 2 m; live foliage is high from the ground to the canopy.

Monotypic exotic: Nearly monotypic, dense stands of exotics such as salt cedar or Russian olive (*Elaeagnus angustifolia*), 4-10 m in height forming a nearly continuous, closed canopy (with no distinct overstory layer); lower 2 m often very difficult to penetrate due to dense branches; however, live foliage density may be relatively low, 1-2 m above ground, but increases higher in the canopy; canopy density is uniformly high**.**

Native broadleaf dominated: Composed of a single willow species, often Goodding's willow (*Salix gooddingii*), or a mixture of native broadleaf trees and shrubs including cottonwood, willows, boxelder (*Acer negundo*), ash (*Fraxinus* spp.), alder (*Acnus* spp.), and buttonbush (*Cephalanthus occidentalis*), height from 3-15 m; characterized by trees of different size classes; often a distinct overstory of cottonwood, willow, or other broadleaf tree with recognizable subcanopy layers and a dense understory of mixed species; exotic or introduced species may be a rare component, particularly in the understory.

Mixed native/exotic: Dense mixtures of native broadleaf trees and shrubs mixed with exotic or introduced species such as salt cedar or Russian olive; exotics are often primarily in the understory but may be a component of the overstory; the native and exotic components may be dispersed throughout the habitat or concentrated as a distinct patch within a larger matrix of habitat; overall, a particular site may be dominated primarily by natives or exotics, or be a more or less equal mixture.

Riparian patches used by breeding southwestern willow flycatcher vary in size and shape, and may be a relatively dense, linear, contiguous stand or an irregularly shaped mosaic of dense vegetation with open areas. Southwestern willow flycatcher have nested in patches as small as 0.8 ha and as large as several hundred hectares. Southwestern willow flycatcher have not, however, been found

nesting in narrow, linear riparian habitats measuring less than 10 m wide, although they will use these areas during migration (Sogge et al. 1997, Sogge and Marshall 2000).

Telemetry studies in Arizona suggest that male southwestern willow flycatcher habitat use differed significantly from the amount of available habitat. In total, 53 % of recorded southwestern willow flycatcher locations occurred in mixed mature riparian habitat, which represented only 28 % of the available habitat (Cardinal and Paxton 2005).

Southwestern willow flycatcher breeding habitats usually include or are near open water or saturated soils. As a general rule, southwestern willow flycatcher nests are rarely more than a few dozen meters away from water or saturated soils (Sogge and Marshall 2000).

Results of studies conducted along the lower Colorado River by McKernan and Braden (2001) suggest that vegetation species composition varied equally for both occupied and unoccupied study areas and provided no clear distinction, based on perennial species composition, as to what constitutes suitable southwestern willow flycatcher habitat. Foliage height profiles of occupied study areas provided no clear demarcation of the understory, subcanopy, or canopy structure. Only slight differences in canopy heights between occupied and unoccupied areas were detected, which suggests that canopy height may probably not be a good indicator of suitable habitat. Based on foliage density, patchiness of occupied and unoccupied habitats were for the most part equivalent, suggesting that patchiness may not serve as an indicator of occupancy, although it may represent the quality of occupied habitat.

Koronkiewicz et al. (2004) reported that southwestern willow flycatcher breed in a wide variety of riparian habitat types throughout the Virgin and lower Colorado River regions. Although occupied southwestern willow flycatcher habitat at each of the study areas consisted of relatively homogeneous, contiguous stands of riparian vegetation, the sites differed from each other both in structure and composition. Vegetation and habitat characteristics at four study sites were measured during the 2003 study. On plots at Pahranagat NWR (n=25), average canopy height was 15.3 m (SE=1.6), percent total canopy closure was 90.8 % (SE=2.57), percent woody ground cover was 13.8 % (SE=3.4), percent of plot centers within 30 m of standing water or saturated soil was 24 %, distance to nearest canopy gap was 5.9 m (SE=0.8), percent of plot centers within 30 m of a broadleaf tree was 100 %, the number of shrubs or sapling stems within a 5 m radius of the plot center was 10.6 (SE=5.9), and the number of tree stems within an 11.3 m radius of the plot center was 11.2 (SE=2.3). The one characteristic reported by Koronkiewicz et al. (2004) and common to all habitat types where southwestern willow flycatcher were found, regardless of plant species composition, height, and canopy closure was that foliage density was always greatest from 2-4 m above the ground (at and immediately above mean nest height). (For vegetation and habitat characteristics at the Mesquite, Mormon Mesa, and Topock study sites, refer to Table 6.1 in Koronkiewicz et al. 2004). Generally, McLeod (2005) found that canopy closure was greater and there were more tree stems at nest sites than at non-nest sites.

Along the Gila and San Pedro rivers in Arizona, nesting southwestern willow flycatcher selected dense habitat patches dominated by young tamarisk and willow trees located near moist soils or standing water and within a larger complex of riparian forest. Occupied patches had greater basal area and total foliage volume than unoccupied patches, and foliage density was greater at all height

intervals. Canopy cover was greater and less variable in occupied patches. Maximum canopy did not differ between occupied and unoccupied patches, although there was less variation in height in occupied patches. Average canopy cover was 88 %, while minimum canopy cover was 71 % (Paradzick 2005).

In Arizona, Allison et al. (2003) report that areas within southwestern willow flycatcher nesting habitat exhibited greater canopy closure than non-nesting areas, and foliage density was highest at nest height. Greater canopy closure, taller canopy height, and dense foliage at nest height might provide a more favorable microclimate at nests. The presence of water may also be a factor in providing a suitable microclimate (Sogge 2000). Based on microclimate data collected in 2003, Koronkiewicz et al. (2004) suggest that microclimate (in a complex interaction with habitat type, vegetative structure, and possibly other factors) may potentially limit nesting habitat suitability. Because standing water or saturated soils were present at all nesting sites, Koronkiewicz et al. (2004) suggest that measuring the presence of water early in the breeding season may serve as a good indicator of preferred southwestern willow flycatcher breeding habitat.

Microclimate studies were conducted by McLeod et al. (2005) to document temperature, relative humidity, and soil moisture at southwestern willow flycatcher nests. Comparisons were made between nest sites, within territory sites, and with unoccupied riparian habitat outside the southwestern willow flycatcher's territory. Results suggested that southwestern willow flycatcher prefer to establish territories and build their nests at sites with significantly cooler, more humid, and wetter microclimates.

Nesting

In southern California and Arizona, nest building begins in mid to late May. Females usually select the site, collect nesting material, and build the nest while males perch nearby. Nest construction usually lasts between 4 and 7 days (Sedgwick 2000, Sogge 2000).

McKernan and Braden reported that 40.9 % (27 of 66) of the known nests substrates during the 2000 surveys consisted of tamarisk and nest success was significantly greater for nests placed in tamarisk than for nests placed in willow species (McKernan and Braden 2001). It was also reported that the highest southwestern willow flycatcher productivity occurred in monotypic tamarisk habitats while some of the lowest flycatcher productivity occurred at willow dominated sites. Koronkiewicz et al. (2004) reported that 57 % of all nests found were placed in tamarisk, 24 % were found in Goodding's willow, and 18 % in coyote willow (*Salix exigua*). In Arizona, tamarisk and Goodding's willow were the primary nesting substrates, although mesquite, common buttonbush, and Arizona ash (*Fraxinus velutina*) were also used (Munzer et al. 2005).

Along the Virgin, lower Colorado, and Bill Williams Rivers, McKernan (1997) reported that the mean height of nest placement for all nests found was 2.4 m (s.d. ± 0.35) while Koronkiewicz et al. (2004) reported a mean nest height of 2.9 m (SE=0.19) and McLeod (2005) suggested a mean nest height of 3.2 m. In Arizona, mean nest height at two different study areas was 4.73 and 3.58 m (Munzer et al. 2005).

Food Habits

The southwestern willow flycatcher is primarily insectivorous and capable of catching flying insects on the wing, although they may also glean insects from leaves by hovering. Flycatchers forage within and above the canopy, along the patch edge, in openings within their territory, and above surface water (Sogge 2000). In Ontario, flycatchers were reported as having spent 5 % of their time foraging and 63 % sitting. As such, they have been described as "time minimizers" because they are able to simultaneously engage in foraging, territorial advertising, vigilance, and resting (Prescott and Middleton 1988). In southeastern Washington, Frakes and Johnson (1982) reported that willow flycatchers exhibited flexibility in behavior by changing their foraging behavior such that it was optimized for a particular habitat in response to interspecies competition.

Changes in native vegetation are thought to alter available food resources and subsequently influence avian population abundance, distribution, and behavior (Kleintjes and Dahlsten 1994). Salt cedar is thought to negatively impact flycatcher reproduction and spatial distribution by altering insect fauna (Sedgwick 2000). Durst (2004) reported significant differences between arthropod communities found in native, mixed, and exotic riparian habitats as well as differences in flycatcher diets in each of these habitats. Based on the flycatcher's ability to exploit a diverse array of arthropod taxa, there is little evidence to suggest that changes in potential prey base associated with salt cedar have an impact on southwestern flycatcher habitat quality (Durst 2004).

Based on the analysis of fecal samples collected from southwestern willow flycatcher at Roosevelt Lake in central Arizona, Durst (2004) identified 1,316 individual prey items across eight taxonomic groups, which represent between 94 and 96 % of the southwestern willow flycatcher's prey. The taxonomic groups identified included *Araneae*, *Coleoptera*, *Diptera*, *Formicidae*, *Hemiptera*, *Homoptera*, flying *Hymenoptera*, and *Lepidoptera*. Significant differences in southwestern willow flycatcher diets were reported by habitat. More *Homoptera* were consumed in exotic habitat than in mixed or native habitats, more *Lepidoptera* were consumed in native habitat than in mixed or exotic habitats, and more *Araneae* were consumed in native habitats than in exotic ones (Durst 2004).

Based on the stomach contents of 135 willow and alder flycatcher that were collected from May through September from 17 states, Beal (1912) reports that 96.05 % of food consumed consisted of animal matter, and 3.95 % consisted of vegetable matter. *Coleoptera* (beetles) of all types made up approximately 17.89 % of the animal matter consumed and represented a large portion of southwestern willow flycatcher diet during all months except September, when vegetable matter was the most frequently consumed. *Hymenoptera* (bees, wasps, and ants) represented the largest item of animal food consumed by southwestern willow flycatcher (41.37 %) in every month and were represented mostly by bees and wasps. *Diptera* (flies) accounted for 14.20 % of the southwestern willow flycatcher's diet and were consumed regularly every month except for September when it only accounted for 8.05 % of the southwestern willow flycatcher's diet. The southwestern willow flycatcher's diet also consisted of *Hemiptera* (true bugs), and *Lepidoptera* (butterflies and moths). Of the vegetable food consumed, primarily during the month of September, raspberries and blackberries (*Rubus* spp.) were among the most common.

In Ontario nestling diets consisted primarily of *Diptera* and *Hemiptera* but also included *Mollusca* (mollusks), *Arachnida* (spiders), *Isopoda* (primitive crustaceans), *Orthoptera* (grasshoppers and crickets), *Coleoptera*, *Lepidoptera*, and *Hymenoptera* (Prescott and Middleton 1988).

Conservation and Management

Effects of Human Activity

The decline of the southwestern willow flycatcher has likely occurred due to the destruction and degradation of riparian woodlands in combination with heavy brood parasitism by the brown-headed cowbird (*Molothrus ater*) (Ehrlich et al. 1992). However, along the lower Colorado River, dramatic changes in flow regime due to human manipulation have facilitated habitat degradation and destruction. That is, the restriction of seasonal natural flood events has enabled aggressive exotic plant species to invade and out-compete native plants restricting regeneration and repopulation, which has subsequently influenced flycatcher habitat. Livestock grazing may also directly influence flycatcher populations. In Arizona, riparian woodlands comprise less than 0.5 % of the landscape (Strong and Bock 1990) where cattle grazing is the major cause of decline (Ehrlich et al. 1992, Sedgwick 2000). Damage by livestock includes soil compaction and gullying, grazing of willows, and changes in willow foliage height and volume (Harris et al. 1987). However, cattle grazing is not a significant concern influencing southwestern willow flycatchers along the lower Colorado River.

Salt cedar have invaded riparian areas throughout the west, particularly the southwest, and have replaced some riparian communities completely. For example, salt cedar replaces the preferred multi-layered shrub community with a monotypic stand that has only one shrub layer, decreases plant and insect diversity, and can increase the frequency and intensity of fire. Dams, flood control, and highly saline irrigation water also give salt cedar a competitive edge over native vegetation (Sogge et al. 1997). More than 40 % of all avian species depend, at least in part, on the presence of riparian communities in the southwestern United States (Hunter et al. 1988). Loss of native riparian habitat and subsequent spread of salt cedar has been shown to negatively influence the population sizes of many riparian bird species (Anderson et al. 1977) and studies along the Colorado River suggest the bird species richness and total density is lower in salt cedar communities than in native riparian habitat (Anderson and Ohmart 1984). However, recent work has shown southwestern willow flycatchers will often nest within salt cedar when specific habitat components are present (Koronkiewicz et al. 2004). Specifically, microclimate, which is influenced by habitat structure, type, and presence of water, may be a limiting factor in nesting suitability (Koronkiewicz et al. 2004) regardless of the presence of a native or an exotic plant community (McKernan and Braden 2001).

The southwestern willow flycatcher is a common host for brood parasitism by the brown-headed cowbird (Bent 1942, King 1955, Walkinshaw 1966, Sedgwick and Knopf 1988). Brood parasitism may pose a significant threat, particularly in its western range where habitat is limited and fragmented, and where livestock are often present in meadows and riparian habitats. There is, however, evidence of adaptive behavior toward brown-headed cowbird parasitism in some populations. Parasitized nests may be abandoned or dismantled, and renesting may occur in some cases (Sedgwick and Knopf 1988), although fewer eggs may be laid (Holcomb 1974). Southwestern willow flycatcher nesting success and productivity are thought to be in some cases reduced as a

result of brown-headed cowbird parasitism (Sedgwick and Knopf 1988, Larison et al. 1998, Whitfield and Sogge 1999, Sedgwick and Iko 1999, Finch et al. 2000). McLeod et al. (2005) reports that nest parasitism across all study sites along the lower Colorado River ranged from 0-47 % and averaged 26 %. McKernan and Braden (2001) suggest nest parasitism by brown-headed cowbird along the lower Colorado River has little consequences, if any, on the daily or overall probability of nest survival for the southwestern willow flycatcher.

Degradation of Backwater Habitat

Along the Colorado River, riparian habitats have been modified, reduced, or lost downstream of dams as a result of changes in flood frequency and duration (Marshall and Stoleson 2000). In 1981 24 months of continual high flows from water releases at Alamo Dam resulted in the loss of 120 ha of cottonwood-willows at the confluence of the Bill Williams and Colorado Rivers (Hunter et al. 1987). Flood-control projects generally shorten, straighten, and narrow river channels such that main channels are cutoff from side channels and adjacent floodplains, thereby reducing meander patterns, slowing stream velocity, and dampening the effects of flooding. Channelization alters stream banks by elevating them well above groundwater, which essentially restricts access by phreatophytic vegetation to groundwater and reduces the overall width of wooded riparian habitat (Marshall and Stoleson 2000).

Habitat Criterion and Research Needs for Assessing the Value of Habitat

Along the lower Colorado River, significant differences in vegetation measurements were identified between southwestern willow flycatcher nesting habitat and non-use areas (Table 4). Canopy height in occupied habitat ranged from 5.2-15.8 m, percent canopy closure ranged from 89.6-98.4 %, and nest substrate height ranged from 2-21.8 m. Vertical foliage density in native habitats ranged from 2-4 m, while foliage density in mixed native/exotic and monotypic exotic habitat was confined to the upper strata. Nesting substrate usually consists of salt cedar, Goodding's willow, Fremont cottonwood (*Populus fremontii*), mesquite, coyote willow, and snags (Koronkiewicz et al. 2004, McLeod et al. 2005, Munzer et al. 2005). Linear southwestern willow flycatcher habitat measures at least 10 m wide for nesting purposes. Although habitat less than 10 m wide will not be used for breeding, they will be used as stopover areas during migration (Sogge et al. 1997, Sogge and Marshall 2000).

Microclimate data collected along the Virgin and lower Colorado Rivers by Koronkiewicz et al. (2004) and McLeod et al. (2005) suggest that southwestern willow flycatcher select nesting habitat by proximity to water or saturated soils, diurnal temperature, and relative humidity. In native habitat where at least 90 % of the community is dominated by native vegetation, the average distance to water or saturated soils is 38.7 m. In exotic habitat where 90 % of the community consists of exotic vegetation, the average distance to water or saturated soils is 28.3 m. In mixed habitat where 50 to 90 % of the vegetation is dominated by either native or exotic plants, the average distance to water or saturated soils is 28.4 m. Mean diurnal temperature in native habitat is 28.7 degrees C while diurnal temperature in exotic habitat is 31.3 degrees C. Diurnal temperatures in mixed habitat averages 32.2 degrees C. Diurnal relative humidity averages 42.2 % in native habitat, 55 % in exotic habitat, and 49 % in mixed habitat.

The southwestern willow flycatcher has, as a result of its decline and subsequent listing as an endangered species, been relatively well studied. As such, scientists and land managers throughout the southwestern United States have identified a variety of planned or needed research.

Table 4. Generalized habitat requirements for southwestern willow flycatcher.

SOUTHWESTERN WILLOW FLYCATCHER GENERALIZED HABITAT REQUIREMENTS	
Community type[a]	Mixed, SC
Community structural type[b]	III, IV
Patch size	≥ 0.8 hectares
Linear habitat width / length	Unknown length / ≥ 10 meter width
Understory density	High
Foliage density	High
Foliage density height	2-4 meters
Canopy cover	High
Proximity to water	≤ 24 meters
Nesting period[c]	Mid-season, late
Nesting habitat selection based on microclimate[d]	Yes

[a] Although cottonwood and willow dominate yellow-billed cuckoo habitat, saltcedar is frequently found in the understory; mixed = native/non-native, CW = cottonwood willow; SC = monotypic saltcedar, W = monotypic willow.
[b] Community structural types as defined by Anderson and Ohmart (1976).
[c] Early = February - April, mid-season = May - June, late = July - September.
[d] Although not well understood, southwestern will flycatcher nest site selection is thought to be influenced, at least in part, by microclimate where humidity is high and temperature is cooler (Koronkiewicz et al. 2004, McLeod et al. 2005).

Riparian areas in the southwest are critically important to a variety of vertebrate taxa (Ellis et al. 1997, Auble and Scott 1998) and are possible only where a river or other water source provides a moisture substrate (Patten 1998). Although water management practices have been implicated in reducing the health of riparian communities (Busch and Smith 1995, Friedman et al. 1998), much remains to be learned about how such practices affect riparian vegetation and how water management can be altered to benefit riparian habitat and southwestern willow flycatcher. The effects of groundwater withdrawal or pumping, surface water movement, and dredging and channel bed alterations on riparian communities should be studied further (Stoleson et al. 2000).

Habitat characteristics at southwestern willow flycatcher breeding sites should continue to be quantified in terms of patch area and shape, proximity to water (Stoleson et al. 2000, Paradzick 2005), stand age and successional status, vertical structure, plant species composition, landscape matrix, climate and microclimate, and topography (Stoleson et al. 2000, Munzer et al. 2005). The relative importance of different habitat types to southwestern willow flycatcher may vary both seasonally and annually among sites, affecting observed home ranges and movement patterns. As such, telemetry studies involving fledgling and female southwestern willow flycatcher should be conducted to better understand family movements and fledgling dispersal patterns, and habitat use over time (Cardinal and Paxton 2005).

Features of habitat structure associated with nest parasitism may be related to nest searching behavior of brown-headed cowbird and should be investigated in future research. Community-wide studies should be investigated, since habitat features that decrease parasitism in some species may increase parasitism in others (Larison et al. 1998). Because parasitism rates vary with site, year,

patch size, and southwestern willow flycatcher and brown-headed cowbird population sizes, sites should be monitored over multiple seasons (Finch et al. 2000). Although brown-headed cowbird parasitism has been found to negatively impact southwestern willow flycatcher populations, habitat destruction and modification are the primary causes of decline in southwestern willow flycatcher populations; brown-headed cowbird parasitism is merely a symptom. Therefore, habitat acquisition, improvement, and restoration must be given high priority and implemented in conjunction with brown-headed cowbird monitoring and, if necessary, control measures (Whitfield and Sogge 1999).

Although Durst (2004) was able to describe southwestern willow flycatcher diets as they pertain specifically to sites in Arizona, such data have yet to be collected along the lower Colorado River. To further quantify the effects of encroachment by exotic vegetation on southwestern willow flycatcher habitat, invertebrate sampling in occupied and unoccupied areas should be conducted. Furthermore, differences in the abundance and diversity of invertebrates between occupied and unoccupied habitat may help to further define southwestern willow flycatcher-specific habitat requirements.

Western Yellow-billed Cuckoo

Distribution

The summer distribution of the western yellow-billed cuckoo ranges throughout much of the United States, southeastern Canada, Greater Antilles, and Mexico, but range boundaries have been confused by recurrent observations of non-breeding individuals away from their breeding sites. Vagrants are not unusual on the Atlantic coast and Canadian prairies, and they are occasionally found as far away as Alaska and Western Europe (Bent 1940, Ehrlich et al. 1992, Hughes 1999).

Although once thought to be fairly common throughout the lower Colorado River Valley, the largest remaining western yellow-billed cuckoo population occurs in the Bill Williams Delta. Populations are apparently decreasing elsewhere in the valley (Anderson and Ohmart 1984, Rosenberg et al. 1991). Halterman (1998) suggests that the Bill Williams NWR is one of the few remaining places in the western United States were an apparently viable population of western yellow-billed cuckoo exists. Gaines (1974) suggests that the area above Laguna Dam on the Colorado River may be the last stronghold for western yellow-billed cuckoo in California.

In 2003, while conducting southwestern willow flycatcher surveys along the Virgin and lower Colorado rivers, Koronkiewicz et al. (2004) documented western yellow-billed cuckoo at Pahranagat NWR, Mormon Mesa, Topock Marsh, Bill Williams NWR, Imperial NWR and Yuma. In 2004, McLeod et al. (2005) documented cuckoos at Pahranagat NWR, Topock Marsh, Bill Williams NWR, and Yuma.

The western yellow-billed cuckoo's winter range extends from northern South America to eastern Peru, Bolivia, and northern Argentina (Ehrlich et al. 1992).

Migration

The western yellow-billed cuckoo is one of the few species whose normal spring migration period extends well into mid June (Robbins et al. 1986). Along the lower Colorado River, the western yellow-billed cuckoo was reported as a fairly common breeder from June through August and a rare breeder in late May and early September (Phillips et al. 1964, Anderson and Ohmart 1984, Rosenberg et al. 1991). Halterman (1998) reports that most breeding pairs were not encountered at the Bill Williams NWR until early July, and a few pairs were not encountered until late July.

Habitat

Nesting habitat throughout much of the western yellow-billed cuckoo's range consists of trees or shrubs in open woodlands with dense undergrowth, extensive riparian woodlands, thickets, or parks (Ehrlich et al. 1992).

Mature cottonwood-willow stands provide the primary habitat in the lower Colorado River Valley. Willows or isolated cottonwoods mixed with tall mesquite (*Prosopis glandulosa* and *P. pubescens*) are also used but to a lesser extent. As a mid-summer breeder in the lower Colorado River Valley, the western yellow-billed cuckoo must be a nest-site specialist for cooling eggs. Mature cottonwoods, with willows forming a subcanopy layer, provide the best shading of any riparian habitat. Standing water found in many cottonwood-willow-dominated areas may also help reduce air temperature by evaporative cooling (Rosenberg et al. 1991).

On a 20-ha study plot along the Bill Williams River, Rosenberg et al. (1982) report that western yellow-billed cuckoo were most commonly found in stands of cottonwood and Goodding's willow with a patchy understory of salt cedar (*Tamarix chinensis*), cattails, and bulrush. Another study on the Bill Williams NWR documented that western yellow-billed cuckoo utilize extensive areas of willow, salt cedar, and cattails (Halterman 1998).

Western yellow-billed cuckoo in Arizona are most frequently found in cottonwood-willow and dense mesquite associations throughout the state (Phillips et al. 1964). In California Gaines (1974) only observed western yellow-billed cuckoo in areas where the extent of riparian vegetation exceeded 300 m in length and 100 m in width. No observations were made in narrow riparian strips 20-100 m wide. All observations were made within 100 m of water, and areas in which western yellow-billed cuckoo were found most frequently consisted of extensive riparian vegetation interspersed with lakes, sloughs, and/or marshy areas. Other reported characteristics of western yellow-billed cuckoo habitat included dense understory vegetation and willow thickets. Gaines also suggests that as much as 10 ha of river bottom vegetation per breeding pair may be required for nesting.

The relationship between habitat patch size and the proportion of patches that are occupied by western yellow-billed cuckoo has been found to be of considerable importance in defining nesting habitat in California (Laymon and Halterman 1989). Only 9.5 % of sites measuring between 20 and 40 ha were found occupied, while 100 % of sites measuring more than 80 ha in size were occupied. Along the lower Colorado River, 46.2 % of sites that were between 20 and 40 ha were occupied. One study site more than 80 ha in size was found to be occupied (Laymon and Halterman 1989).

Nesting

Nesting microhabitat is generally described as consisting of broad-leafed deciduous hardwoods with thick bushes, vines, or hedgerows providing dense foliage within 10 m of the ground (Laymon 1980, Hughes 1999). In arid regions nesting sites are usually restricted to river bottoms, ponds, marshes, and damp thickets with relatively high humidity (Gaines and Laymon 1984). Nests in the western United States are usually associated with willow, Fremont cottonwood, mesquite, hackberry (*Celtis* spp.), soapberry (*Sapindus saponaria*), alder, and cultivated fruit trees (Hanna 1937, Laymon 1980, Hughes 1999). At the Bill Williams NWR, 79 % of western yellow-billed cuckoo monitored in 1998 were found in cottonwood-willow-dominated habitat and 94 of 95 nests were found in willows specifically (Halterman 1998).

Based on four nests found in California, Laymon (1980) describes nest site characteristics as: height from ground ranging from 2.5-4.3 m, distance from trunk ranging from 4.0-4.5 m, distance from end of branch ranging from 0.3-1.0 m, percent foliage cover in area ranging from 70-85 %, foliage cover above nest ranging from 80-98 %, and distance of nest to foliage above ranging from 0.15-90 m. Nests were generally well concealed by surrounding foliage, especially from above.

At Bill Williams NWR, vegetation surveys determined that the average nest height was 5.8 m and ranged from 1.8 to 17.0 m. The average nest tree measured 9.2 m tall and 19.0 cm in diameter at breast height (DBH). The average canopy closure in nesting habitat was 78.7 %, and the amount of bare ground ranged from 65.8-18.6 %. Areas with low ground cover consisted primarily of grasses. Willows were the dominant tree species at nest sites (67.3%), followed by cottonwoods (18.5%) and salt cedar (14.3%). The shrub layer surrounding nests consisted mainly of salt cedar, although seep willow (*Baccharis glutinosa; B. salicifolia*) were also present. The percent canopy closure decreased as distance from the nest increased and measured 95 % directly beneath the nest. The maximum average tree foliage (3-5 m) was at nest height, suggesting that western yellow-billed cuckoo prefer to nest in trees where foliage density is highest (Halterman 1998).

Food Habits

The western yellow-billed cuckoo's diet consists mostly of insects, especially caterpillars (*Lepidoptera* spp.). It is also known to consume tree frogs, occasional bird eggs, berries, and other fruit (Ehrlich et al. 1992). Of six birds examined from the lower Colorado River, large insects made up the largest portion of western yellow-billed cuckoo diet. Cicadas (*Diceroprocta apache*) were apparently the western yellow-billed cuckoo's most important food item. Other food items included mantids (*Orthoptera* spp.), grasshoppers (*Orthoptera* spp.), and caterpillars (*Lepidoptera* spp.). Western yellow-billed cuckoo are also known to catch lizards and tree frogs (Rosenberg et al. 1991). On the Bill Williams River, grasshoppers were also considered an important food item. Rosenberg et al. (1982) and Halterman (1998) report that cicadas and katydids were the principal food items at the Bill Williams NWR. Young western yellow-billed cuckoo were fed cicadas, katydids, caterpillars, sphinx moth larvae, and grasshoppers. Adult males showed a preference for green Orthopterans while females preferred cicadas (Halterman 1998).

Rosenberg et al. (1991) report that of 48 observed foraging attempts, two-thirds occurred in willows and the remainder in cottonwoods. Nearly all attempts apparently occurred at more than 6 m above

the ground. Based on telemetry studies of two western yellow-billed cuckoos at the Bill Williams NWR, it was determined that, although western yellow-billed cuckoo nest predominantly in willows, they prefer to forage in cottonwoods (Halterman 1998).

The onset of breeding appears to be correlated with an abundance of local food (Hamilton and Orians 1967, Rosenberg et al. 1982, 1991, Ehrlich et al. 1992). Once initiated, the breeding cycle occurs extremely rapidly and requires only 17 days from egg-laying to fledgling of young (Hamilton and Orians 1967, Hughes 1999). Large clutch sizes are thought to be a response by western yellow-billed cuckoo to the presence of excess food sources (Nolan and Thompson 1975, Fleischer et al. 1985).

Conservation and Management

Effects of Human Activity

Habitat studies along the lower Colorado River in Arizona suggested that 31 % (1,068 ha) of cottonwood-willow habitat was lost from 1976 to 1986 and resulted in a 16 % decrease in western yellow-billed cuckoo populations. Water releases from Alamo Dam in the 1970s and 1980s destroyed approximately 99 % of remaining cottonwoods and 64 % of willows. The estimated number of western yellow-billed cuckoo before flooding was approximately 13 birds per 40 ha. Post-flood populations were estimated at three western yellow-billed cuckoo per 40 ha (Groschupf 1987, Hughes 1999). Remnant fragments of riparian habitat are also being threatened by the encroachment of salt cedar. Laymon and Halterman (1987) report that cottonwood-willow communities present in 1977 are now monotypic stands of salt cedar that remain unused by western yellow-billed cuckoo. Historically, riparian communities were destroyed by agricultural conversion, submersion below reservoirs, and channelization for flood control (Halterman 2002). Pesticides are thought to affect western yellow-billed cuckoo behavior or cause death by direct contact. Pesticide spraying in orchards has exposed nests and young to sublethal poisoning (Laymon 1980). Pesticides also contaminate the cuckoo's preferred prey, especially *Lepidopteran* larva. Other prey species, such as frogs, occur in pesticide-contaminated runoff adjoining agricultural land. Shell fragments taken from three nests in California averaged 19 % thinner than shells collected from pre-DDT periods (Laymon and Halterman 1987). In Florida, Grocki and Johnston (1974) found that 15 of 16 individual birds collected during the spring and fall had DDT or its metabolites in interfurcular adipose tissue. Pesticide concentrations ranged from 0.01-3.34 parts per millions lipid weight. Average DDT concentrations were found to be higher in cuckoo collected during the fall than those collected during the spring, suggesting that pesticides may be excreted during migration or when over wintering. Pesticides may also be translocated to brain or muscle cells.

Degradation of Backwater Habitat

Flow-control structures, such as dams, have greatly altered connected streamside communities (Patten 1998) and destroyed much of the riparian communities once present along the lower Colorado River (Halterman 1998). Reductions in sediment loads, narrowing of channels, reduction in the recruitment of vegetation, and increase of flow velocities are known to contribute to the loss of riparian-wetland, marsh, and backwater communities (Brown and Johnson 1985, Collier et al. 1996, Auble and Scott 1998, Friedman et al. 1998). Remaining remnant riparian habitat fragments are being threatened by encroachment of exotic vegetation such as salt cedar. Monotypic stands of

salt cedar generally increase the likelihood of catastrophic fire (Bock and Bock 1990, Smith et al. 1998) and may desiccate water courses (Vitousek 1990, DiTomaso 1998). Removal of groundwater, coupled with regulated flows, has contributed significantly to the loss of riparian communities and promoted erosion through the loss of stream bank stabilization by plant roots (Groeneveld and Griepentrog 1985).

Habitat Criterion and Research Needs for Assessing the Value of Habitat

Vegetation measurements taken at nest sites in 1998 determined that the average nest height was 5.8 m and ranged from 1.8-17.0 m (Table 5). The average nest tree measured 9.2 m tall and 19.0 cm DBH. The average canopy closure in nesting habitat was 78.7 %, and the amount of bare ground ranged from 65.8-18.6 %. Areas with low ground cover consisted primarily of grasses. Willows were the dominant tree species at nest sites (67.3 %), followed by cottonwoods (18.5 %) and salt cedar (14.3 %). The shrub layer surrounding nests consisted mainly of salt cedar, although seep willow was present also. The percent canopy closure decreased as distance from the nest increased and measured 95 % directly above the nest. The maximum average tree foliage (3-5 m) was at nest height, suggesting that western yellow-billed cuckoo prefer to nest in trees where foliage density is highest (Halterman 1998). During summer 2001 Halterman (2002) reported measurements that differed considerably from those reported in 1998. Average nest height was 7.3 m, average nest tree height was 10.3 m, and the average tree height was 9.7 m. Total canopy cover at the nest was 89.5 %, and all but one nest were found in Goodding's willow (Halterman 2002). Laymon and Halterman (1989) suggest narrow sites dominated by salt cedar should be considered poor habitat; however, western yellow-billed cuckoo found using these areas have been shown to nest successfully. Habitat use by western yellow-billed cuckoo may be too variable to allow the use of a single set of evaluation criteria. Such variability in habitat use suggests a need for continued long-term studies to evaluate habitat variables under varying conditions. Additionally, habitat preferences and requirements are not fully understood and should be studied further (Halterman 2002).

Based on multi-year surveys throughout California and along the Colorado River, Laymon and Halterman (1989) suggest that western yellow-billed cuckoo habitat requirements be redefined to include characteristics of microhabitat. Rosenberg et al. (1991) also indicate that microclimate may be a significant factor in selection of nesting habitat, since western yellow-billed cuckoo are mid-summer breeders and may be required to cool their eggs. That is, western yellow-billed cuckoos may select cottonwoods and willows for nesting habitat to shade from harmful temperatures and select cottonwood/willow groves with standing water to benefit from the evaporative cooling process (Rosenberg et al. 1991). Although such studies have been conducted since 1989 (Halterman 1998, Halterman 2002, Halterman 2004), much remains to be learned. Determining how and why western yellow-billed cuckoo are selecting habitat will improve the likelihood of successfully creating or restoring suitable habitat.

Even though widespread loss of riparian habitat has been implicated as the primary reason for decline of the western yellow-billed cuckoo (Laymon and Halterman 1987, Ehrlich et al. 1992), recent declines along the lower Colorado River, at the Bill Williams NWR, and in California do not appear highly correlated with large-scale losses of nesting habitat. A secondary factor, or combination of factors, is probably contributing to recent declines (Halterman 2002). Continued research should be conducted to identify secondary causes of declines in western yellow-billed

Table 5. Generalized habitat requirements for western yellow-billed cuckoo.

WESTERN YELLOW-BILLED CUCKOO GENERALIZED HABITAT REQUIREMENTS	
Community type[a]	CW, W
Community structural type[b]	I, III
Patch size	20-40 hectares
Linear habitat length/width	300 meters / 100 meters
Understory density	High
Foliage density	High
Foliage density height	≤ 10 meters
Canopy cover	High
Proximity to water	≤ 100 meters
Nesting period[c]	Mid-season, late
Nesting habitat selection based on microclimate	Unknown

[a] Although cottonwood and willow dominate yellow-billed cuckoo habitat, saltcedar is frequently found in the understory; mixed = native/non-native, CW = cottonwood willow; SC = monotypic saltcedar, W = monotypic willow.
[b] Community structural types as defined by Anderson and Ohmart (1976).
[c] Early = February - April, mid-season = May - June, late = July - September.

cuckoo populations. Laymon and Halterman (1987) suggested that pesticides may be at least partially responsible for these declines. Since western yellow-billed cuckoo are known to consume insects, tree frogs, and occasionally bird eggs (Ehrlich et al. 1992), the prevalence of pesticides in the western yellow-billed cuckoo's prey should be evaluated as well.

Although western yellow-billed cuckoo are known to successfully nest in salt cedar, changes from native vegetation to salt cedar and other exotic species are thought to alter available food resources and subsequently influence avian population abundance, distribution, and behavior (Kleintjes and Dahlsten 1994). In Arizona Durst (2004) reports significant differences between arthropod communities found in native, mixed, and exotic riparian habitats, as well as differences in southwestern willow flycatcher diets in each of these habitats. Implications associated with these types of changes with regards to the western yellow-billed cuckoo should be investigated to determine how to better manage exotic communities in an attempt to create native riparian communities along the lower Colorado River.

Laymon (1980) suggests that the possibility of captive breeding and reintroduction of western yellow-billed cuckoo to naturally regenerated or reforested riparian habitat should be investigated. Western yellow-billed cuckoo may recolonize portions of their former range if suitable habitat has been restored. Western yellow-billed cuckoo were found foraging on 11 ha of replanted sites in southern California in the second year and nesting during the third year following replanting. Western yellow-billed cuckoo recolonized areas only when cottonwood and willows grew to optimum height and foliage volume (Anderson and Laymon 1989).

Yellow-billed cuckoo nests occasionally are parasitized by the brown-headed cowbird. Clotfelter and Brush (1995) report the use of the western yellow-billed cuckoo as hosts by the bronzed cowbird (*Molothrus aeneus*) in south Texas and suggest that bronzed cowbird parasitism and egg puncturing of western yellow-billed cuckoo nests and eggs increases the frequency of nest abandonment, which may have contributed to the decline of the species. However, little information is known of the frequency of parasitism of the western yellow-billed cuckoo along the Bill Williams and lower Colorado Rivers. Therefore, additional research is needed to investigate the frequency and impact of parasitism on the western yellow-billed cuckoo within the lower Colorado River Valley. Community-wide studies should be pursued since habitat features that decrease parasitism in some species may increase parasitism in others (Larison et al. 1998).

It has been suggested that yellow-billed cuckoos, because they are late arrivals to the lower Colorado River Valley, may select nesting habitat based at least in part on nest site microclimate to keep nests and eggs cool. Prior to implementation of yellow-billed cuckoo habitat restoration plan, studies should specifically target microclimate and investigate the significance of nest microclimate on habitat suitability.

Conclusions

As mentioned in the preceding avian profiles, the primary threat to all backwater, marsh, and riparian obligate species described is loss or degradation of habitat. As a highly managed and regulated system, the lower Colorado River is not the dynamic and highly variable system it once was. The decrease of periodic flooding and sediment deposition, coupled with increased flow velocities and other factors, has contributed to the degradation or loss of stream side communities. Furthermore, such conditions have promoted the encroachment of non-native vegetation such as salt cedar and common cane to name a few.

Changes to stream-side communities have wide ranging implications to all bird species. Their responses to such changes can be highly variable, since they tend to utilize habitat and other available resources differently. For example, sediment deposition resulting from the construction of dams has created suitable habitat for the Yuma clapper rail, which is thought to have allowed it to extend its range northward into areas where it historically wasn't found. Additionally, Yuma clapper rail have been shown to change habitat requirements seasonally; however; shallow water and emergent vegetation interspersed with high ground are considered vital components of suitable Yuma clapper rail habitat regardless of season. Although the California black rail is also considered a marsh obligate species, it appears to prefer the transition zone between marsh and upland communities. Also, it is not known to alter habitat requirements seasonally. As such, California black rail appear to be considerably more vulnerable to fluctuations in water levels than Yuma clapper rail and other marsh species. It has been suggested that creation of habitat suitable for Yuma clapper rail may also benefit the California black rail. Indeed, such a possibility may be true during certain seasons and should be investigated further. However, structure of habitat and fluctuation of water levels are thought to be the limiting factors in determining whether or not habitat will be utilized. California black rail have been shown to utilize dense emergent vegetation almost exclusively. Yuma clapper rail, on the other hand, have been shown to utilize both dense and moderately dense communities and alter habitat requirements seasonally. Additionally, Yuma clapper rail have been shown to tolerate high water levels by moving to high ground. However, such

responses by California black rail have not been documented. Given these differences, newly created habitat targeted specifically for the Yuma clapper rail may not necessarily benefit the Yuma black rail.

Although different species may require similar habitat, it is clear that structure is important in dictating whether or not these birds will utilize available habitat. Studies designed to quantify such differences may provide information on how to best create suitable habitat for more than a single species. Although marsh size has not been shown to be important to species such as the Yuma clapper rail, creation of larger marshes would likely promote structural diversity, as opposed to small habitat patches where a single structural type would prevail.

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CHAPTER 3: MAMMALS

Currently, the Colorado River cotton rat (*Sigmodon arizonae plenus*) is believed to be disjunct from other populations of *S. arizonae*, which occur in a variety of habitats throughout Arizona and California (Hoffmeister 1986, Blood 1998). As such, the species is limited in distribution to a narrow band of mesic habitat along the lower Colorado River, although the distributional limits of *S. a. plenus* along the Colorado River have yet to be established, and the southern extent of its range are still unknown (Blood 1998). Federally, the Colorado River cotton rat is listed as a Candidate 2 species and a species of special concern in California (Blood 1998).

Generally, *S. a. plenus* is described as a pallid species that most closely resembles *Sigmodon hispidus eremicus,* although much larger in size. Underparts of *S. a. plenus* are pale buff in color, and the rump is rusty. The ears and feet are dull white or gray, and the tail is light brown above and dull gray beneath. *S. a. plenus* measures approximately 294 mm TL, the body measures 164.3 mm long, and the tail is approximately 129.7 mm long (Hoffmeister 1986).

Based on the loss of previously occupied habitat and the lack of trapping success in southern Nevada, isolated cotton rat populations are thought to have gone extinct (Bradley 1966).

Colorado River Cotton Rat

Distribution

The Arizona cotton rat (*Sigmodon arizonae*), as a species, is distributed from northern Yuma County near Ft. Whipple and Camp Verde south and southeast to southern Gila and southwestern Graham counties, and all but the northeastern corner of Cochise County. The Colorado River cotton subspecies, however, is thought to be limited to northwestern Yuma County along the Colorado River from Parker to Ehrenberg (Hoffmeister 1986).

In Nevada, Hall (1946) describes the Colorado River cotton rat's range as restricted to the bottomlands of the Colorado River at the extreme southern tip of the state. Colorado River cotton populations once occurring along the river in Nevada are now thought to be extinct (Hall 1946, Bradley 1966).

Grinnell (1914) reports having collected Colorado River cotton rat from along the California side of the Colorado River. Specimens were collected from an area located a several kilometers south of Palo Verde, 8 km northeast of Yuma, and from an area near Pilot Knob. Although described by Grinnell (1914) as *S. hispidus eremicus*, Hoffmeister (1986) believed that, based on measurements reported by Grinnell, the species was most likely *S. arizonae* and probably *S. a. plenus.* Goldman (1928) reports Colorado River cotton rats from only three locations in California: Needles; Colorado River opposite Parker, Arizona; and 24 km southwest of Ehrenberg, Arizona. The distribution was apparently spotty, rather than continuous.

Currently, the distributional limits of Colorado River cotton rat along the Colorado River have yet to be established, and the southern extent of its range are still unknown (Blood 1998). Generally, Colorado River cotton rat are thought to be restricted in distribution to the mesic habitats of the

lower Colorado River Valley and believed to avoid the xeric conditions of the surrounding desert (Grinnell 1914, Goldman 1928, Hoffmeister 1986).

Systematics

Four subspecies of *S. arizonae* have been described and include *arizonae, plenus, cienegae,* and *jacksoni* (Hoffmeister 1986). Although initially described by Goldman (1928) as *S. hispidus plenus*, Zimmerman (1970) showed that *S. hispidus* and *S. arizonae* differed greatly in chromosome number and structure, and were distinguishable in skeletal structure.

Habitat

Arizona cotton rats generally occupy arid desert habitat composed of mesquite tumbleweed (*Salsola* spp.), and sparse grasses (Hoffmeister 1986). The Colorado River cotton rat, however, is generally thought to be restricted to isolated sections of alluvial bottom along the Colorado River (Goldman 1928) and has demonstrated a strong distributional association with patterns of irrigation canals (Hoffmeister 1986).

Grinnell (1914) reports that Colorado River cotton rats were found in a tule (*Scirpus acutus*) patch at the edge of a slough adjacent to a dense stand of seedling willow. Northeast of Yuma, Grinnell found Colorado River cotton rats in wiregrass (*Cynodon dactylon*) bordering a backwater near a stand of young willow. Near Pilot Knob, Colorado River cotton rats were reported in cane (*Phragmites* spp.) surrounded by dense arroweed (*Pluchea sericea*). Generally, Grinnell found that Colorado River cotton rats occupied cottonwood willow communities where the dominant vegetation consisted of willow, Fremont cottonwood, baccharis (*Baccharis glutinos; B. salicifolia*), and common reed. Grinnell also noted that Colorado River cotton rats were common in irrigated agricultural fields.

In Nevada Colorado River cotton rats were found on a tract of less than 0.4 ha in size with flowing water and cattail. The surrounding upland communities consisted of bermudagrass (*Cynodon dactylon*) and mesquite. The Colorado River cotton rats were found living just above water level and created runways that passed through 2.5 cm or more of water. Runways measured approximately 8 cm in width and were made by either cutting grass near its base or by running over grass and pushing it down (Hall 1946).

Blood (1998) suggests that Colorado River cotton rat along the Colorado River may have expanded their range by occupying agricultural fields. He also reports (Blood 1990) capturing Colorado River cotton rat in disturbed and open areas. Trapping efforts in grassy habitats along the Colorado River resulted in no captures. Zimmerman (1970) reports capturing *S. a. plenus* in stands of common reed near Parker, Arizona.

Nesting

Nesting habitat is largely unknown, although Hall (1946) reported that the Colorado River cotton rat were found living just above the water line in Nevada. Similar species, including *S. hispidus eremicus* (Goldman 1928), are thought to be restricted in distribution to the lower Colorado River

Valley and have been taken from areas dominated by *Pluchea*, *Typha*, and *Phragmites* (Hoffmeister 1986). *S. h. eremics* is known to nest beneath rocks, logs, and in burrows.

Food Habits

Specific details on the feeding habits of the Colorado River cotton rat are unknown; however, similar species, such as *S. hispidus*, are largely herbivorous and diets usually consist of stems, foliage, seeds, insects, and small vertebrates (Martin et al. 1961).

Conservation and Management

Effects of Human Activity

Flow-control structures, such as dams, have greatly altered connected streamside communities (Patten 1998) and destroyed many of the backwater communities once present along the lower Colorado River. Reductions in sediment loads, narrowing of channels, reduction in the recruitment of vegetation, and increase of flow velocities are known to contribute to the loss of riparian-wetland, marsh, and backwater communities (Brown and Johnson 1985, Collier et al. 1996, Auble and Scott 1998, Friedman et al. 1998). Marshlands on the lower Colorado River are threatened by encroachment of salt cedar, *Phragmites*, and other exotics. Salt cedar generally increases the likelihood of catastrophic fire (Bock and Bock 1990, Smith et al. 1998) and may desiccate water courses (Vitousek 1990, DiTomaso 1998). Removal of groundwater, coupled with regulated flows, has contributed significantly to the loss of riparian communities and promoted erosion through the loss of stream bank stabilization by plant roots (Groeneveld and Griepentrog 1985). In contrast, human activity may potentially benefit the Colorado River cotton rat. Along the Colorado River, Blood (1990) captured individuals only in disturbed agricultural fields, while Grinnell (1914) noted that cotton rats were common in irrigated fields.

Degradation of Backwater Habitat

Marsh and backwaters that historically occurred along the lower Colorado River have been destroyed or severely degraded by agricultural conversion, construction of reservoirs, river channelization, and shoreline stabilization. Given the regulated nature of the Colorado River, natural formation of new marshes and backwaters resulting from channel movement and periodic flooding is now unlikely. However, flow regulation and shifts in the timing of flows for water diversion have resulted in the development of large marsh and backwater complexes where riparian vegetation historically occurred. Marsh complexes developed behind Imperial Dam and Parker Dam at the Bill Williams Delta and Topock Marsh. The construction of training structures also created areas of more expansive and permanent backwaters and marshes than had occurred historically on the lower Colorado River (MSCP 2004).

Habitat Criterion and Research Needs for Assessing the Value of Habitat

The Colorado River cotton rat is thought to require habitat dominated by emergent vegetation including common reed, bulrush, and cattails (Table 6). Other species include willow, cottonwood, baccharis, bermudagrass, and arroweed. However, specific habitat requirements (e.g., stem density, vegetation height, water depth) for the Colorado River cotton rat are unclear. Studies are needed to better discern optimal structural components needed by the cotton rat along the Colorado River. In addition, Colorado River cotton rats are known to occupy – and some research indicates they prefer – disturbed agricultural fields and exhibit a strong distributional association with irrigation

Table 6. Generalized habitat requirements for Colorado River cotton rat.

COLORADO RIVER COTTON RAT GENERALIZED HABITAT REQUIREMENTS	
Community type	Agriculture/cottonwood-willow
Structural type	Unknown
Proximity to water	Close
Patch size	Unknown
Water depth	Unknown
Water fluctuation	Unknown
Seasonal requirements	Unknown
Nesting requirements	Unknown

canals. Additional research is needed to discern why such areas are beneficial over native habitat. Specifically, studies should attempt to define Colorado River cotton rat distributions and determine if distribution is seasonally variable. If distribution is seasonal, studies should attempt to identify seasonal differences in habitat type and structure. Studies should also attempt to identify soil types associated with suitable habitat and how soil types might influence nesting and burrowing. Patch or habitat size requirements are largely unknown and should be the target of future studies along with implications of daily fluctuations in water on habitat suitability. Studies should also attempt to address nesting habits and their relationship with water velocity and fluctuation. Finally, studies should seek to determine where suitable habitat is located relative to upland areas, marsh and riparian communities.

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